

# **DRAFT FINAL REPORT**

## **SEDIMENT INVESTIGATIONS AT ISLAIS CREEK AND MISSION CREEK 1998 – 1999 – 2000**



**Prepared for:  
CITY AND COUNTY OF SAN FRANCISCO  
PUBLIC UTILITIES COMMISSION, SPARC**

**Submitted to:  
CALIFORNIA STATE WATER RESOURCES CONTROL BOARD**



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**Prepared by:**

**Battelle Memorial Institute  
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Carlsbad, CA 92008**

**November, 2002**

## **PREFACE**

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In early 1998, the City and County of San Francisco, Public Utilities Commission (SFPUC) began intensive investigations into the sediment conditions at Islais and Mission Creeks. These investigations, which were conducted over the course of three years, required the combined efforts of San Francisco Bay Regional Board staff, program managers and scientists from the City and County of San Francisco, and scientists and technicians from several environmental consulting firms. Although the project team largely remained intact over the course of these investigations, many of the team members changed firms and/or affiliations before completing this report. Ms. Leslie Lundgren consistently served as Program Manager for SFPUC and facilitated interactions between the Regional Board and the City and County of San Francisco. She was the primary contact for the two Regional Board staff Project Managers that oversaw the investigations: Mr. David Leland from 1998-1999 and Mr. Brad Job from 1999-2001. Ms. Cynda Maxon functioned as the Scientific Project Manager, while with Arthur D. Little (ADL) from 1998 to 2001, and with Battelle Memorial Institute (Battelle) from 2001 through 2002. Ms. Maxon was the primary author of this report. Field sampling was overseen by Ms. Arleen Navarett of SFPUC with support from Olivia Chen Consultants and SCA Environmental. Ms. Navarett also oversaw sediment toxicity tests conducted at the SFPUC Oceanside Biology Laboratory. Organic chemicals were analyzed at the former ADL Environmental Laboratory in Cambridge, MA. Inorganic analyses were performed at the SFPUC Water Quality Bureau Laboratory and overseen by Mr. Jim Salerno. Mr. Frederick Newton of Battelle oversaw the bioaccumulation testing performed by EVS Environmental Consultants. Mr. Newton also assembled and managed the project database, and performed all statistical tests.

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## ACRONYMS AND ABBREVIATIONS

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<b><u>Abbreviation</u></b>	<b><u>Definition</u></b>
%	percent
µg	microgram
µg g <sup>-1</sup>	micrograms per gram (same as ppm)
µg L <sup>-1</sup>	micrograms per Liter (same as ppb)
<	less than
>	greater than
µm	micron
‰	parts per thousand
AAF	atomic absorption flame
AAGF	atomic absorption graphite furnace
AAH	atomic absorption hydride
AAS	atomic absorption spectrometry
ADL	Arthur D. Little, Inc.
Al	aluminum
ANOVA	analysis of variance
ARAR	applicable or relevant and appropriate requirement
As	arsenic
ASTM	American Society of Testing Methodology
AVS	acid volatile sulfides
Ba	barium
Battelle	Battelle Memorial Institute
BPTCP	Bay Protection and Toxic Cleanup Plan
BSAF	biota-sediment accumulation factor
C	centigrade
Cd	cadmium
CERCLA	comprehensive environmental response, compensation and liability act
cm	centimeters
COC	chemical of concern
COPC	chemical of potential concern
CPI	carbon preference index
Cr	chromium
CSO	City-operated combined sewer outfall
CTD	conductivity, temperature and depth
Cu	copper
CV	coefficient of variation
CVAA	cold vapor atomic absorption
CVAAS	cold vapor atomic absorption spectrophotometry
DDT	dichlorodiphenyltrichloroethane
DGPS	Differential Global Positioning System

DHS	Department of Health Services
DMMO	Dredge Materials Management Organization
EBS	Environmental Baseline Study
EIA	Environmental Impact Assessment
EPA	Environmental Protection Agency
ERL	effects range-low
ERM	effects range-median
FAAS	flame atomic absorption spectrophotometry
FDA	United States Food and Drug Administration
Fe	iron
FED	functional equivalent document
g	gram
GC	gas chromatograph
GC/FID	gas chromatography/flame ionization detection
GC/MS	gas chromatography/mass spectrometry
GPS	Global Positioning System
H <sub>2</sub> S	hydrogen sulfide
H <sub>2</sub> SO <sub>4</sub>	sulfuric acid
HCl	hydrochloric acid
HF-HNO <sub>3</sub> -HClO <sub>4</sub>	hydrofluoric-nitric-perchloric acid
Hg	mercury
HMW	high molecular weight
HNO <sub>3</sub>	nitric acid
ICP	inductively coupled plasma spectrometry
ICP/MS	inductively-coupled plasma/mass spectrometry (ICP)
kg · m <sup>-3</sup>	kilogram per cubic meter
kg	kilogram
km	kilometer
L	liter
LAB	linear alkylbenzene
LMW	low molecular weight
m	meter
m <sup>2</sup>	square meter
MDL	method detection limit
mg	milligram
mg?kg <sup>-1</sup>	milligrams per kilogram
MGD	million gallons per day
MGP	manufactured gas plant
mL	milliliter
mm	millimeter
N/P	napthalenes to phenanthrenes ratio
NAS	National Academy of Science
ng?g <sup>-1</sup>	nanograms per gram (same as ppb)
NH <sub>3</sub>	ammonia

NIST	National Institute of Standards and Technology
nm	nanometer
NWAS	Northwest Aquatic Sciences
OC	organic carbon
OCC	Olivia Chen Consultants
OEHHA	Office of Environmental Health Hazard Assessment
p	probability
PAH	polycyclic aromatic hydrocarbons
PER	Pacific EcoRisk
Pb	lead
PCA	principal component analysis
PCB	polychlorinated biphenyl
ppb	parts per billion (same as $\text{ng g}^{-1}$ )
ppm	parts per million (same as $\text{? g g}^{-1}$ )
PRMP	pilot regional monitoring program
r	correlation coefficient
$r^2$	correlation coefficient squared
RBI	relative benthic index
RI/FS	remedial investigation and feasibility study
RMP	regional monitoring program
RPD	relative percent difference
RSD	relative standard deviation
RTU	relative turbidity units
RWQCB	San Francisco Bay Regional Water Quality Control Board
SAP	Sampling and Analysis Plan
SCA	SCA Environmental
SEWPCP	South East Water Pollution Control Plant
SFPUC	San Francisco Public Utilities Commission
SHC	saturated hydrocarbons
SIM	selected ion monitoring
SOP	standard operating procedure
SRM	standard reference material
SWRCB	State Water Resources Control Board
SWTP	San Francisco wastewater treatment plant
TIE	toxicity identification evaluation
TOC	total organic carbon
TPAH	total polycyclic aromatic hydrocarbons
TPH	total petroleum hydrocarbons
TRV	toxicity reference value
UCM	unresolved complex mixture
UPL	upper predictive limit
ZGFAAS	Zeeman graphite furnace atomic absorption spectrophotometry
Zn	zinc

## 1.0 INTRODUCTION

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This report presents the results, conclusions, and recommendations of two sediment investigations conducted in San Francisco Bay at Islais and Mission Creeks on behalf of the City and County of San Francisco, Public Utilities Commission (SFPUC). Two of the investigations were conducted during wet weather in October 1998 and April 2000. A single dry weather investigation was conducted in October 1999. This report also evaluates the relevant regulatory framework and data utilized by the San Francisco Bay Regional Water Quality Control Board (RWQCB) in the Bay Protection and Toxic Cleanup Plan (BPTCP) in the designation of Islais and Mission Creeks as toxic hot spots.

Creek conditions were examined relative to clean in-bay reference sites to evaluate the spatial extent of sediment chemical contamination and toxicity. Sediment contaminant type and distribution were examined for each creek in order to estimate relative contributions from City-operated combined sewer overflows (CSOs) and other possible sources. In addition, this information may be used to evaluate possible remedial or preventative measures. The scope of the investigations followed the Sampling and Analysis Plan (SAP) first submitted to the RWQCB in October 1998 (ADL 1998), and revised for the October 1999 and May 2000 investigations (ADL 1999). The SAPs were responsive to the RWQCB's Section 13267 letter of June 1998 and subsequent letters issued in August and September 1998 that further defined the requirements for collection and analysis of sediment data in the two subject creeks. A third site, Yosemite Creek, was also addressed in the RWQCB August and September 1998 letters, and investigated concurrently with Islais and Mission Creeks. Results for the Yosemite Creek studies will be issued in a separate report. The investigations were performed by Arthur D. Little, Inc. (ADL) and SFPUC, with field support from Olivia Chen Consultants (OCC) and SCA Environmental (SCA). This report was prepared by Battelle Memorial Institute (Battelle).

### 1.1 PURPOSE OF REPORT

The primary objectives of this report are to determine the current environmental status of each of the two creeks, and to confirm or refute the toxic hot spot designations of Islais and Mission Creeks. These designations were based on state legislation, passed in 1989, which provided modifications to Division 7, Chapter 5.6 of the California Water Code, and are described as the Bay Protection and Toxic Cleanup Program (BPTCP).

Toxic hot spot designation by the RWQCB was primarily based upon significant recurrent toxicity to test organisms from sediments collected (and tested) in Fall 1994 and Spring 1997 by the RWQCB in both Islais and Mission Creeks. The designations were supported by elevated chemical concentrations in creek sediment and indications of an impacted benthic invertebrate community in 1997.

The RWQCB issued a Water Code Section 13267 letter in June 1998, requiring SFPUC to define sediment contamination in Mission and Islais Creeks and determine the extent to which combined sewer overflows at both creeks and the Quint Street outfall at Islais are presently or historically responsible (RWQCB 1998). In addition, information regarding the vertical extent of contamination was required to determine the extent of impact and for use in considering remedial options.



The comprehensive sampling program initiated by SFPUC was responsive to both RWQCB requests concerning the two creeks. Toxicity and chemistry tests were conducted on surface sediments to document the horizontal extent of impacts; and subsurface cores were analyzed to estimate the vertical extent of contamination. Bioaccumulation of selected chemicals of potential concern (COPCs) was examined in May 2000 only, using clam tissue exposed to creek sediments in standard 28-day laboratory tests. The analytical results provide an appropriate basis for interaction with the RWQCB to achieve final site designation and to develop plans for future prevention and control strategies, if warranted.

Specific objectives of these investigations were:

1. to confirm or refute the RWQCB's findings, indicating that Mission and Islais Creeks are toxic and contaminated;
2. to define the vertical extent of contamination in the two creeks;
3. to define the horizontal extent of contamination and toxicity under both wet and dry weather conditions in the two creeks;
4. if possible, to determine sources of contamination to the receiving creek beds; and,
5. to provide recommendations regarding remediation or preventative actions appropriate for each creek.

## 1.2 REPORT ORGANIZATION

The main body of this report consists of ten sections described below. Appendices A and B present laboratory data and graphics, organized by creek and study year. Detailed descriptions of laboratory, data analysis and quality control methods used in support of these investigations are presented in separate documents (ADL 1998 & 1999) that are included on the CD-ROM (*Sediment Investigations at Islais Creek and Mission Creek*), accompanying this report.

*Section 1 - Introduction* presents the purpose and objectives of the study, citing regulatory requirements and background. An overview of methods used in the BPTCP to evaluate sediment chemical and biological data in the designation of toxic hot spots is provided. Methods used by the RWQCB are contrasted with those used in other regulatory programs to examine alternative interpretations of sediment data.

*Section 2 – Site History* presents the location, geological setting, potential contaminant sources, and sediment chemical and biological results from previous investigations for each of the two creeks. Historical data used in the hot spot designations of Islais and Mission Creeks are reviewed.

*Section 3 – Decision Rules, Study Design and Methods* presents decision rules that are applied in Section 9, to determine the extent of environmental impact at each creek. These rules were agreed upon by SFPUC and RWQCB staff, and are used to determine whether the subject creeks warrant additional action. An overview of the study design applied to each creek along with sampling inventories and corresponding minor modifications made between sampling events are presented in Section 3.2. Brief

discussions of field, laboratory and analytical procedures are provided in Section 3.3. Detailed methods are presented in the project SAPs are included on the CD-ROM.

*Section 4 – Physical Characteristics of Sediment* presents results for grain size and total organic carbon in creek and reference area sediments. Effects of sediment physical characteristics on chemical and biological parameters are emphasized.

*Section 5 – Sediment Toxicity* presents results for standard 10-day acute laboratory tests, using the amphipod *Eohaustorius estuarius*. Results for creek and reference areas are described by presenting central tendencies, range and variation. Comparisons of survival between creek and reference areas also are presented for each of the two creeks. October 1998 and 1999, and May 2000 data are compared to estimate temporal variation for each creek.

*Section 6 – Sediment Chemistry* presents results for organic and metal contaminants measured in each of the two creeks, defining vertical and horizontal patterns of distribution. Results are described by presenting central tendencies, range and variation. Results for statistical comparisons between creek and reference area surface sediments also are presented, and are used to identify contaminants of potential concern (COPC) for each creek. October 1998 and 1999, and May 2000 data are compared to estimate temporal variation for each creek.

*Section 7 – Bioaccumulation in Clams* presents results for selected chemicals of concern measured in organisms exposed to sediments in standard 28-day laboratory tests. Results are described by presenting central tendencies, range and variation. Lipid-normalized chemical concentrations in tissue for each creek station are compared to tissue results from clams exposed to in-bay reference stations.

*Section 8 – Source Identification of Selected COPCs.* This section discusses potential contaminant sources for selected COPCs at each creek. Chemicals of potential concern identified in Section 6 that may have multiple sources are discussed, including metals, polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs). Knowledge of site use combined with forensic chemistry methods, including source ratio and principal components analysis (PCA), are used to identify likely contaminant sources for each creek.

*Section 9 – Application of Decision Rules and Recommendations* are presented in this section. The current status of each creek is determined by applying chemistry and toxicity results to the decision matrix presented in Section 3. Recommendations for no further action, additional analyses, remedial action or preventative measures are presented following the results from the applied decision rules. Arguments to delist all of Mission Creek and most of Islais Creek as confirmed toxic hot spots are presented, since these sediments no longer meet RWQCB criteria.

Cited literature is listed in *Section 10*.

### 1.3 REGULATORY BACKGROUND

In 1989, the California State Legislature established a Bay Protection and Toxic Cleanup Program (BPTCP). Under BPTCP legislative authority, four goals were established: 1) protection of present and future beneficial uses of California's bays and estuaries; 2) identification and characterization of contaminated areas; 3) development of plans for cleanup of contaminated sites, or other remedial or mitigation action; and 4) development of control strategies for toxic pollutants to prevent creation of new areas of contamination.

The BPTCP designated contaminated areas as "toxic hot spots," following Section 13391.5 of the Water Code, as areas in which hazardous substances have accumulated in water or sediment to levels which 1) may pose a substantial present or potential hazard to aquatic life, wildlife, fisheries, or human health; 2) may adversely affect beneficial uses of bays, estuaries, or ocean waters as defined the water quality control plans; or 3) exceed adopted water quality or sediment quality objectives. Additional details of BPTCP toxic hot spot definition are provided by SWRCB (1998).

The definition of a toxic hot spot was broadly defined in the Water Code. The legislation, therein, granted the State Water Resources Control Board (SWRCB) and RWQCB the final authority for designation of toxic hot spots in their areas of jurisdiction, providing a means for refined definitions to address specific contamination issues within the context of local background conditions. A "potential toxic hot spot" is a site where information exists indicating possible impairment but without sufficient information to be classified as a "candidate toxic hot spot" (Hunt et al. 1998a).

As set forth in the Draft Functional Equivalent Document (FED) Appendix A, Volume I (SWRCB 2000), and summarized by Hunt et al. (1998a), a site is considered a candidate toxic hot spot if any of the following conditions are met.

1. The site exceeds water or sediment quality objectives or water quality standards of the RWQCB or the U.S Environmental Protection Agency (EPA).
2. The water or sediment exhibits recurrent toxicity associated with toxic pollutants that is significantly different from the toxicity observed at reference sites. The toxicity tests must also be approved by the SWRCB and RWQCB, and toxic pollutants should be present in concentrations known to cause or contribute to the toxic response.
3. Tissue pollutant levels of organisms collected from the site exceed acceptable levels established by the U.S. Food and Drug Administration (FDA) for the protection of human health, or National Academy of Science (NAS) standards for protection of human health and wildlife. If the state Office of Environmental Health Hazard Assessment (OEHHA) or Department of Health Services (DHS) has issued a health advisory against consumption of edible resident non-migratory organisms on a site or water body, the site is a candidate toxic hot spot if the contaminant of concern is associated with sediment or water at the site.
4. Impaired environmental conditions associated with toxic pollutants, as measured by reduction in growth, reduction in reproductive capacity, abnormal development, or histopathological (tissue) abnormalities in organisms.

5. Significant degradation in biological populations and/or communities associated with the presence of elevated levels of toxic pollutants.

Once a candidate toxic hot spot was designated and approved by the SWRCB and RWQCB, it was subsequently listed in the statewide cleanup plan as a “known toxic hotspot,” including requirements for pollution prevention, control and remediation.

These definitions include a degree of subjectivity that provides a means of interpretation and site definition within the framework of local conditions. For example, there are no regulatory standards for sediment quality in the state of California (Condition 1), and interpretation of sediment toxicity results in the case of San Francisco Bay is complicated by the presence of a residual level of toxicity in virtually all bay sediments representing background conditions. Condition 2 raises the issue of chemical concentrations “known to cause, or contribute to, toxic responses.” Marine and estuarine sediments present a complex physical and chemical mixture of compounds and elements, potentially including toxic contaminants of anthropogenic origin. The biological effects of contaminants are significantly influenced by their equilibria dynamics between adsorbed and soluble states, as determined by physical and chemical conditions within the matrix of sediment particles and pore-water. Factors such as organic carbon content, grain size, reduction-oxidation potential, pH, and the presence of compounds associated with organic degradation (e.g., ammonia, hydrogen sulfide), all play a role in contaminant bioavailability and biological response, including acute or chronic toxicity, and need to be considered in the determination of impact. Because the basic physical-chemical nature of sediment varies between localities, irrespective of introduced contaminants, considerable variability has been observed in toxicity tests with respect to individual contaminant concentrations (Long & Morgan 1990; Long et al. 1995, 1998); and chemical concentrations “known to cause toxicity” cannot accurately be predicted. However, predictability of response generally increases as a function of the number of contaminants present that exceed estimated concentration thresholds (Long et al. 1998, MacDonald et al. 2000).

Significant degradation of biological communities (Condition 5) has its definitive precepts in the well-documented changes in benthic communities associated with point source discharges of contaminants, particularly where organic matter is introduced. Reduced biological diversity and the presence of high numbers of opportunistic “pollution indicator” species characterize these areas in relatively stable offshore marine habitats. Shallow estuarine sites that are subject to wider ranging vicissitudes of salinity and temperature, and food availability tend to have higher levels of variability in diversity and individual species abundance under natural conditions, and effects of biological contaminants are less discernible in the face of elevated natural variability. High levels of variability in San Francisco Bay benthic communities and difficulties in the interpretation of pollution effects are recognized by both regional regulators and benthic ecologists (Hunt et al. 1998a; Nichols 1986; Thompson et al. 1997).

### 1.3.1 Toxic Hot Spot Identification in San Francisco Bay

In the San Francisco Bay region, the phases of BPTCP toxic hot spot identification included: 1) a review of existing reports on water and sediment quality; 2) surveys of sites in a Pilot Regional Monitoring Program (PRMP), which included a method validation study along a pollution gradient; 3) a reference site study that established toxicological and statistical methods for identifying polluted sites in

comparison with reference conditions in the bay; 4) a toxicity screening study of sites throughout the region, selected on the basis of previous information and PRMP results; and 5) confirmation (follow-up) studies of sites that exhibited toxicity and/or elevated chemistry during screening (Hunt et al. 1998a).

The monitoring and screening approach to identifying sediment toxic hot spots included toxicity testing, chemical analysis for trace metals and organic compounds, and benthic community analysis. This approach, known as the sediment quality triad (Chapman et al. 1997), was modified in that benthic community structure was not included for all confirmation studies due to inherent difficulties in the interpretation of benthic data for parts of San Francisco Bay (see Methods, Hunt et al. 1998a). The primary screening tool was sediment toxicity testing, augmented by chemical and biological analyses, with definition of a candidate toxic hot spot contingent upon evidence of recurrent toxicity by confirmation testing at a subsequent date. Most sites in the bay, including Islais and Mission Creeks, were initially monitored in 1994, with follow-up confirmation in 1997.

In emphasizing this “weight-of-evidence” approach, the BPTCP upon Condition 2, defined toxic hot spot sites as those displaying recurrent sediment toxicity or impaired benthic community, supported by synoptic chemical concentrations present at levels known to be associated with toxicity.

### 1.3.1.1 Toxicity Criteria

A primary BPTCP criterion for candidate and final toxic hot spot designation is recurrent toxicity, i.e., “when at least two samples collected at different times from a station or site are determined to be significantly toxic by any of the BPTCP toxicity test protocols.” BPTCP criteria for evaluation of toxicity data in San Francisco Bay adopted an approach of statistical comparison with background bay reference sites. Sediment samples were considered toxic “if there was at least a 95% probability that the sample was as toxic or more toxic than would be expected from the worst 10% of reference samples” (Hunt et al. 1998a). This approach, which acknowledges the presence of background residual toxicity in the bay, developed reference envelope tolerance limits for sediment toxicity (Table 1-1). For example, within this framework, any toxicity test result in which amphipod (*Eohaustorius*) survival exceeded 69.5% of control survival would not indicate significant toxicity. Control sediments from an unimpacted habitat are collected with the test organisms (e.g., amphipods). Animals tested in these control sediments must display at least 90% amphipod survival for a successful test.

**Table 1-1. Reference tolerance limits for San Francisco Bay sediment toxicity tests (from Hunt et al. 1998a).**

Protocol	Tolerance Limit as percent (%) of Control
Amphipod ( <i>Eohaustorius</i> ) Survival	69.5
Sea Urchin Larval Development in porewater	94.3
Sea Urchin Larval Development at sediment water interface	86.7

The reference site standards shown in Table 1-1 indicate moderate toxicity of background sediments to the amphipod test organism and relatively little effect upon sea urchin larval development.

Five BPTCP reference sites sampled in 1994 and 1995 were utilized for establishing background conditions for the amphipod test. San Francisco Bay BPTCP reference sites and test results are

described by Hunt et al. (1998b). In addition, test results from three regional monitoring program (RMP) sites were included as part of the regional background database for the 10-day amphipod test (Dr. John Hunt, UC Santa Cruz, personal communication, 5/99). Toxicity test procedures for the RMP are described in the 1994 Annual Report (SFEI 1995).

Although conservative, this approach is reasonable to assess relative toxicity in San Francisco Bay. However, due to the transient nature of toxicity test results the approach is significantly improved if test (i.e., the site under investigation) sediment toxicity data are compared to reference toxicity data collected and analyzed within the same time frame (similar to sediment control data). This is because temporal variations in reference sediment toxicity may occur if bay-wide conditions or test organisms are stressed. Under changing conditions it is important to quantify the background or residual toxicity of uncontaminated reference sites for comparison and subsequent determination of toxic hot spots.

**Confounding Factors.** Factors other than anthropogenic contaminants can control or contribute to toxicity expressed in a sediment bioassay. These factors obscure or interfere with the relationship between chemicals of concern and the observed toxic effect and are commonly referred to as “confounding factors”. Confounding factors known to influence benthic infauna toxicity, include porewater quality, temperature, salinity, sensitivity and health of test organisms, and sediment physical characteristics. Some of the most prominent confounding factors are non-persistent chemicals such as ammonia, hydrogen sulfide, or salinity, and persistent sediment features such as total organic carbon (quantity and quality), grain size, sediment grain angularity, and water content and cohesiveness of sediment. Sensitivity of tested populations to these factors cannot be overstated, especially when evaluating studies across seasons. Laboratory quality control measures are intended to reduce effects from confounding factors; however, they are often difficult to measure and impossible to control. Not controlling or accounting for confounding factors during testing can produce effects that are not related to the chemicals of concern, leading to misinterpretation of sediment quality.

Four specific types of possible confounding factors were identified from the analysis of BPTCP toxicity test data: elevated levels of ammonia ( $\text{NH}_3$ ) and hydrogen sulfide ( $\text{H}_2\text{S}$ ), low dissolved oxygen concentrations and increased test organism sensitivity due to acclimation and holding time. Ammonia and hydrogen sulfide were “quantified to assist in the interpretation of biological analyses” in the BPTCP studies, but no attempt was made to control test concentrations. These chemicals, most often attributed to the bacterial degradation of organic compounds, occur naturally in marine and estuarine sediments. Sediments associated with quiescent, low energy environments, like those found in the upper ends of Islais and Mission Creeks, are often found to contain toxic levels of ammonia and hydrogen sulfide in the complete absence of other contaminants. Depressed dissolved oxygen concentrations are generally associated with organically enriched (high TOC) sediments and high levels of bacterial/biological activity. The San Francisco Dredge Materials Management Organization (DMMO) acknowledges the importance of these confounding factors and their influence on toxicity test results and in concordance with EPA Region IX, recommends a modification to standard static amphipod toxicity testing protocols. This important, often necessary, modification permits the sequential replacement of water overlying the tested sediments prior to the introduction of test organisms. During this procedure, overlying water is removed (approximately 80%), tested for ammonia and hydrogen sulfide, and replaced with clear fresh seawater adjusted to specified test conditions (e.g., temperature, salinity). If ammonia and hydrogen

sulfide levels are found to exceed threshold limits for a specific test organism, sediments can be equilibrated for approximately four hours followed by water replacement. This sequence is repeated until ammonia and hydrogen sulfide levels are below threshold limits. Admittedly, the replacement of overlying water also has the potential of removing chemicals of concern, but this removal is considered inconsequential since *in situ* sediments are in a continuous state of water replacement. Overlying water replacements were not performed during BPTCP amphipod testing, thus high levels of ammonia and hydrogen sulfide, and low dissolved oxygen levels could have contributed to observed toxic responses.

Recent investigations (Battelle 1999) have shown a negative relationship between toxic response and acclimation/holding time for *Eohaustorius estuarius*. Amphipods that were rapidly acclimated to test salinity conditions (rate greater than 5 parts per thousand [‰] per day) and held for less than 48-hours post acclimation had higher toxicity and inter-replicate variation than amphipods more slowly acclimated and held 48-hours or more prior to testing. Longer combined acclimation/holding times (up to 11 days) seemed to produce healthier, less stressed test organisms. *Eohaustorius estuarius* used in the BPTCP studies were collected in the very low salinity waters (~2 ‰) of Yaquina Bay, Oregon and acclimated to a salinity of 20 ‰. Once acclimated, the amphipods were held for an additional 48 hours prior to test initiation. If the salinity acclimation rate used in the BPTCP studies (rate was not defined) was greater than 5 ‰ per day, organism response (toxicity) could have been confounded by this additional stress.

High levels of hydrogen sulfide, high ammonia, low dissolved oxygen levels and possible short acclimation periods, all known factors that confound the interpretation of toxicity test results, combine and bring into question the conclusions drawn by the BPTCP for Islais and Mission Creeks.

#### 1.3.1.2 Chemical Criteria

The BPTCP adopted chemical guidelines based upon a summary evaluation of 89 studies detailing synoptically collected chemistry and toxicity data from North American coastal estuarine and marine sediments. Results of these studies, which assessed data from more than 1000 samples tested nationwide, were published by Long et al. (1995), providing chemical concentration guidelines that have been embraced as an interpretive tool by regulatory agencies. The guidelines are presented as effects range-low (ERL) and effects range-median (ERM) concentrations for individual and summary (e.g., total PCBs) compounds. Generally, adverse effects were noted in less than 10% of studies in which chemical concentrations were below the ERL guideline. The ERM represents concentrations at the middle of the observed effect range, above which significant toxic effects were observed in more than half of the tests. ERM chemical concentration guidelines from Long et al. (1995) are presented in Appendix C. The BPTCP adopted these guidelines for interpretation of San Francisco conditions, with the exception of a higher DDT criterion, which was derived from local studies in San Francisco Bay (RWQCB 1994) and normalized to organic carbon content following a method from Schwartz et al. (1994).

In recognition of compounded and synergistic effects from sediment co-occurrence of multiple contaminants, Long et al. (1998) presented the additional concept of an ERM quotient for interpretation of sediment toxicity test results. For any suite of sediment chemicals with potential contaminant effects, individual concentrations are divided by the ERM value, producing a corresponding ERM quotient.

These quotients are then totaled and divided by the number of compounds analyzed to give a mean ERM quotient. Relative to controls, Long et al. (1998) found that 71% of amphipod tests indicated a highly toxic response when mean ERM quotients exceeded 1.0, and a 32% incidence within the mean ERM quotient range of 0.11 to 1.0, also noting that the probability of significant toxicity generally increased with increasing numbers of chemicals that exceeded their ERMs. The BPTCP adopted a more conservative guideline of 0.5 for the mean ERM quotient, where exceedance of this value was used as an indication of "contributing chemical contamination."

As a caveat to the wide ranging toxicity results observed nationwide in contaminated sediments, Long et al. (1998) states that sediment quality guidelines were prepared as "...informal (non-regulatory) benchmarks to aid in the interpretation of sediment chemistry data." It was emphasized that sediment quality ERL and ERM guidelines are non-existent for many of the chemicals measured in test sediments, and toxicity test results may be complicated by the presence of high concentrations of ammonia and hydrogen sulfide that can co-vary with anthropogenic substances and contribute to observed toxicity. These compounds are organic breakdown products, occurring both naturally and from anthropogenic activities that result in organic loading of sediments (e.g., domestic sewage discharges).

BPTCP procedures for identification of toxic hot spots have adopted this caveat, utilizing chemistry data only in a supportive context to accompany candidate toxic hot spot designations that are primarily based upon observed recurrent biological impact (i.e., toxicity or impaired benthic community). BPTCP site designations in San Francisco Bay have emphasized the number of chemical concentrations that exceed ERM guidelines, the factor by which they are exceeded, and the ERM quotients. The mean ERM quotient utilized by the San Francisco BPTCP was calculated using ERM quotients of 16 individual and summary chemicals, including nine metals and seven organic compound categories (Fairey et al. 1996). Individual compounds, for each of the following two group categories are summarized in Appendix C.

1. Metals: antimony, arsenic, cadmium, chromium, copper, lead, mercury, silver and zinc.
2. Organic compounds: total DDT, total Chlordane, Dieldrin, Endrin, total PCBs, low molecular weight polycyclic aromatic hydrocarbons (LMW PAH) and high molecular weight PAH (HMW PAH).

BPTCP standards for San Francisco Bay (Hunt et al. 1998a) designated sites as having elevated sediment chemical concentrations if any of the following conditions were present:

1. the mean ERM quotient exceeded 0.5,
2. six or more chemicals were present at concentrations exceeding ERM values, and
3. one or more chemicals were present at concentrations that are known to show a likely association with biological effects, based upon existing literature or best professional judgement.

**Application of ERM Quotients.** The source of ERM values used to calculate each one of the 16 ERM quotients was cited from Long et al. (1995) by the BPTCP. ERM values for chlorinated pesticides used by the BPTCP were actually taken from Long and Morgan (1990). These guidelines were considered inappropriate and subsequently dropped from Long's 1995 listing, except for DDT, which had a different ERM in 1995. The compounds were dropped because of insufficient data, which produced extremely low



confidence in corresponding ERM values, especially for total Chlordane (E. Long, email communication).

Additionally, several individual ERM quotients were calculated incorrectly in the BPTCP, thereby overestimating the level of chemical contamination at potential toxic hot spot sites, including the creeks. Low and high molecular weight PAH concentrations were calculated using 12 compounds each instead of seven and six, respectively, as dictated by Long et al. (1995). BPTCP ERM quotients for low and high molecular weight PAH were both overestimated by an approximate factor of two (since compounds in each group nearly always co-occur). This error caused the BPTCP to incorrectly conclude that PAHs were responsible for "contributing chemical contamination" to observed toxicity using their own definition.

ERM standards are based on data collected nationwide and, therefore, do not necessarily provide the best estimates of threshold concentrations "known to cause toxicity" in San Francisco Bay sediments. For example, nickel has significantly higher background concentrations in San Francisco Bay than its corresponding ERM value; however, it is rarely associated with toxicity as recognized by the BPTCP and others. This condition exists for other substances and warrants determination of ERM values specific to sediments in San Francisco Bay or a different approach altogether to designate sites as chemically elevated (e.g., reference envelope approach).

**Organic carbon and sediment quality guidelines.** The ERM is an effects-based guideline for sediment dry-weight chemical concentrations, which does not account for common factors known to influence contaminant bioavailability and toxicity. Therefore, many sediments with contaminant concentrations much higher than the ERM(s) show no toxic effect when tested. Grain size characteristics, total organic carbon (TOC) content and acid-volatile sulfide (AVS) concentration are known to have a significant influence on sediment contaminant concentrations and associated toxicity (Di Toro 1990, 1991; Lamberson et al. 1992). Sediment contaminants are frequently associated with low-energy (depositional) environments (such as the two creeks) where fine-grained particles accumulate and organic carbon content is often enriched. These environments also are potential repositories for contaminants transported from distant sources via aerial fallout and/or bay hydrodynamics. ERMs, which are based on an average concentration of 1.2% TOC, do not address contaminant bioavailability of these sediments, which include the TOC-enriched sediments often encountered along the San Francisco Bay waterfront.

Since sediment pore-water provides the main route of toxic exposure for many organisms (USEPA 1999), the equilibrium partitioning between the soluble porewater-phase and the relatively unavailable phases associated with organic carbon are critical factors. For nonionic organic compounds (e.g., chlorinated pesticides, PAH, PCB) that have a strong binding affinity for organic carbon, higher TOC portends a reduced level of bioavailability. This equilibrium partitioning approach was first adopted by the EPA in 1996 (EPA 1996 draft), dropped, and then revised in 1999 (EPA 1999) to normalize nonionic organic chemical sediment concentrations to organic carbon content. Application of these guidelines to individual PAH compounds would triple the threshold criteria for sediments with an average TOC concentration of 3%, such as those located in the west end of Islais and Mission Creeks. Total DDT was the only BPTCP "chemical" with a sediment quality criterion based on TOC concentration (i.e., 100  $\mu\text{g/g}$  total DDT per gram organic carbon [ $100 \mu\text{g/g}^1 \text{OC}$ ]) from Schwartz et al. (1994). Use of this criterion

substantially reduces the effective concentration of DDT in sediments with high TOC, such as the creeks. For example, a sediment dry weight concentration of  $100 \text{ ng g}^{-1}$  DDT corresponds to an organic carbon normalized concentration of  $100 \text{ ng g}^{-1}$  OC DDT for a sample containing 1% TOC. However, the same sediment sample containing 2% TOC (similar to those in the west end of Islais and Mission Creeks) would halve the carbon normalized concentration of DDT (i.e.,  $50 \text{ ng g}^{-1}$  OC DDT).

In summary, if the BPTCP continues to use ERM quotients to determine the extent of chemical contamination in support of toxic hot spot designation, the analytes used in the quotients must be consistent with those on which they are based (i.e., Long et al. 1995). In addition, they should be internally consistent as well as consistent with national criteria (e.g., USEPA 1993). Consistency as well as scientific defensibility calls for use of criteria based on organic carbon content for all nonionic organic compounds, such as that used for total DDT, especially if these data are used in support of biological impacts, as they are in the BPTCP. Total organic carbon normalized criteria for nonionic chemicals of concern for the subject creeks is discussed further in Section 4.

**Recalculation of ERM Quotients.** Based on the above discussion, ERM quotients were recalculated for SFPUC 1997 data to assess the current status of chemical contamination of Islais Creek. This was done to confirm or refute BPTCP findings indicating that the subject site is a toxic hot spot, while correcting for errors in the original BPTCP determination. Based on guidelines provided in Long et al. (1995) (except Dieldrin and Endrin, which were based on Long and Morgan [1990] and total DDT based on Schwartz et al. [1994]) and the BPTCP proposed list of contaminants, the mean ERM quotient was recalculated as follows:

$$\text{Mean ERM quotient} = (\text{ArsenicQ} + \text{CadmiumQ} + \text{ChromiumQ} + \text{CopperQ} + \text{LeadQ} + \text{MercuryQ} + \text{SilverQ} + \text{ZincQ} + \text{Total DDTQ} + \text{DieldrinQ} + \text{EndrinQ} + \text{LMW PAHQ} + \text{HMW PAHQ}) / 13 \text{ (total number of chemical quotients)}$$

where:

$$\text{LMW PAHQ} = (\text{Acenaphthene} + \text{Acenaphthylene} + \text{Anthracene} + \text{Fluorene} + \text{C1 Naphthalenes (includes all structural isomers)} + \text{Naphthalene} + \text{Phenanthrene}) / 3160$$

$$\text{HMW PAHQ} = (\text{Benz(a)anthracene} + \text{Benz(a)pyrene} + \text{Chrysene} + \text{Dibenzo(a,h)anthracene} + \text{Fluoranthene} + \text{Pyrene}) / 9600$$

$$3160 = \text{ERM for LMW PAH} \text{ \& } 9600 = \text{ERM for HMW PAH from Long et al. (1995)}$$

All other quotients are the same as those used in the BPTCP.

The following quotients were not used in the recalculated mean ERM quotient for reasons previously discussed or noted: total Chlordane, antimony (not measured in this study) and total PCB. The recalculated mean ERM quotient is based on a total of 13 compounds compared to the mean BPTCP ERM quotient, which is based on 16 compounds. PCBs and Chlordane, which are generally not toxic to sediment biota, were retained as chemicals of concern in this study and evaluated for their tissue bioaccumulation potential (see Section 7).

### 1.3.1.3 Benthic Community and Other Guidelines

BPTCP benthic community guidelines were based upon a Relative Benthic Index (RBI), first developed for the San Diego Bay BPTCP report (Fairey et al. 1996), and modified for use in San Francisco Bay. The index involves a number of calculations that compare site conditions to the range of conditions in the bay. The relative abundance of known pollution-tolerant and pollution-intolerant species from the bay are factored into the calculation, as are total community abundance and the abundance and diversity of molluscs and crustaceans, representing major taxonomic groups that are relatively sensitive to pollution (described in Hunt et al. 1998a). The standardization of site conditions to the range of conditions in the bay gives an RBI ranging from 0.00 (most impacted) to 1.00 (least impacted). The BPTCP adopted an RBI value of 0.3 or lower to indicate a degraded benthic community in San Francisco Bay.

Other BPTCP environmental criteria for hot spot designation include standards for tissue bioaccumulation of contaminants (Condition 3) and impairment of biological function as evidenced by altered growth, reproduction or histopathological abnormality (Condition 4). These two conditions were not factors in the toxic hot spot designation of Islais and Mission Creeks (see RWQCB 1998).

Use of benthic community criteria are probably not appropriate for environments such as Islais and Mission Creeks, which may have impaired communities due to physical factors independent of contaminant concentrations. These physical factors include diminished water circulation (low-energy), episodic inputs of fresh water from road runoff and CSOs, and high deposition of detrital material contributing in part to anoxic conditions (e.g., low levels of dissolved oxygen).

## 2.0 SITE HISTORY

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Site descriptions and environmental histories for Islais and Mission Creeks are provided in this section. Existing data from the Bay Protection and Toxic Cleanup Program (BPTCP) along with results from other relevant studies are presented as a precursor to SFPUC's investigations. Site history and setting are reviewed to identify potential contaminant sources at each creek. Site location and the major drainage basin within San Francisco Bay are shown for each creek in Figures 2-1 and 2-2, respectively.

### 2.1 ISLAIS CREEK

The Islais Creek channel extends approximately one mile in length, running east-west on the San Francisco waterfront, near the foot of Potrero Hill and Caesar Chavez Street (Figure 2-1). The creek has been maintained in its present configuration for more than 100 years. The creek watershed that extended further to the west was land filled prior to the turn of the century. The present Islais Creek drainage basin is pictured in Figure 2-2. The creek channel is presently divided into a western and eastern segment by a narrow 30-m wide constriction underneath the 3rd Street Bridge (Figure 3-1, Section 3). The western channel, representing about one-third of the creek length, ranges from 75-90 m in width. The eastern segment expands to widths of 120-150 m. Channel depths range from approximately 2-11 m. Representative BPTCP sampling of three stations in 1994 and 1997 recorded depths ranging from 2-3 m.

A general description of Islais Creek provided by the RWQCB (1998) portrays the creek banks as being lined with concrete riprap, interspersed with small isolated patches of vegetation. Old pier pilings are common along the south shore of the western segment and much of the eastern segment lies underneath pier structures. These creosote-soaked wooden pilings may provide continuing sources of polycyclic aromatic hydrocarbons (PAH) to creek sediments. Areas adjacent to the creek are characterized by light industrial and urban development, including a sand and gravel facility, grain terminal, oil and grease rendering facility, warehouse and container cargo terminal along shore areas of the eastern segment, and auto wreckers, scrap metal recyclers and warehouses along the western segment. Freeway Interstate 280 passes directly over the upper end of the creek (Figure 3-1, Section 3). These surrounding activities may be pollutant sources to the creek from runoff and deposition from air emissions. The RWQCB considered them minor sources compared to contributions from combined sewer overflows (CSOs) and the Quint Street wastewater outfall (RWQCB 1998).

#### 2.1.1 CSOs and the Quint Street Outfall

Five CSO discharges, including the main CSO Weir, enter into Islais Creek. Combined sewer overflow pairs are located on opposite sides of the creek at 3<sup>rd</sup> Street at the western (upper) end near Shelby and Marin Streets. A main CSO weir, constructed in 1997, is located on the north shore and extends from Indiana Street to about a block west of 3rd Street. Discharges consist of domestic and industrial wastewater and stormwater runoff. They enter into the western segment of Islais Creek during periods of wet-weather flow when the Southeast Water Pollution Control Plant (SEWPCP) exceeds its treatment capacity. The SEWPCP discharges secondarily treated sewage into the bay, approximately 250 m offshore of Pier 80 at a depth of 12.5 m (40 feet), extending offshore about 70 m upcoast of the Islais Creek entrance into the bay (Figure 3-1, Section 3).

Historically, the SEWPCP has treated 67 million gallons per day (MGD) to secondary standards during dry weather flow. The plant has a treatment capacity of 250 MGD during periods of wet weather flow that collect storm water runoff. During these periods, which occur about 600 hours per year, the discharge capacity of the bay outfall (100 MGD) is exceeded and overflows of secondarily treated wastewater up to 150 MGD are routed to the Quint Street outfall, located on the south shore of Islais Creek, just west of the 3rd Street Bridge (Figure 3-1, Section 3). When the overall 250 MGD treatment discharge capacity from both outfalls is exceeded, the additional wastewaters are discharged through the five Islais Creek CSOs. These discharge events, which consist of a combination of storm water runoff with domestic and industrial sewage, presently occur at an average frequency of 10 times per year (J. Salerno, San Francisco Wastewater Treatment, personal communication 4/99). Relevant to the interpretation of recent SFPUC sampling results, are changes and improvements in wastewater discharge capacity that became operative in April 1997. At this time the SEWPCP capacity was expanded from 210-250 MGD, reducing the need for overflow discharge into the creek. Furthermore, wastewaters routed to the Quint Street outfall were upgraded from an approximate 50/50 primary and secondary treatment blend to 100% secondarily treated wastes, improving settleable and floatable waste removal by upwards of 90%. In addition, storage capacity tanks for the CSOs were expanded by 37 million gallons. Combined sewer overflow discharges declined from an average of 60 to 10 events per year.

Dredging activities exert a significant influence on horizontal sediment transport, the vertical sediment profile and release of contaminants into aqueous phase. Dredging in the area of the creek entrance into the bay last took place in 1988, when nearly 500,000 yd<sup>3</sup> were removed, to a depth of 12 m (40 feet) (T. Bruch, U.S. Army Corps of Engineers, San Francisco, personal communication, 4/99).

### 2.1.2 Historical Data

Several earlier studies have documented contaminated sediment conditions in the western creek segment, including elevated chemical concentrations (CH2M Hill 1979 and Chapman et al. 1987), sediment toxicity and degraded benthic community conditions (Chapman et al. 1987). Chapman's study included analysis of sediment coprostanol, a fecal steroid that is an indicator of mammalian waste input. Levels measured in the western creek sediment exceeded bay background concentrations by a factor of about 10, strongly indicating an impact from CSOs and the Quint Street outfall on sediment quality.

A more recent analysis of sediment toxicity conducted in 1996 (MEC Analytical Systems 1997) found significant toxicity in amphipod tests conducted with *Ampelisca* sp. at four (4) of 15 sites from three cross-creek transects in the western channel segment. Sediment pore water conditions showed high levels of ammonia and sulfide in samples from the upper end of the creek where significant toxicity was recorded. Significant toxicity was not evident at 25 sites from five cross-creek transects that were located in the creek segment east of the 3rd Street Bridge. Earlier studies of channel sites to the east of the bridge found PAHs and polychlorinated biphenyls (PCBs) that exceeded ERM levels (Long et al. 1995), and significant toxicity of sediments to urchins and mussels in comparison with reference sites (Anderson et al. 1995).

Historical data were cited by the BPTCP as supportive of the 1998 candidate toxic hot spot designation for the western segment of Islais Creek. However, data from the BPTCP 1994 screening and confirmation studies in 1997 relied on data from only three locations to form the sole evidentiary basis

for Islais Creek candidate toxic hot spot designation under provisions of the California Water Code, as discussed below.

### 2.1.3 Islais Creek Candidate Toxic Hot Spot Designation

The western segment of Islais Creek was designated as a candidate toxic hot spot by RWQCB (1998) due to impacts upon aquatic life resulting from contaminated sediment, meeting the criteria prescribed in Condition 2 of the California Water Code candidate toxic hot spot definition (SWRCB 1998). BPTCP results showed recurrent toxicity in both amphipod and sea urchin tests at Station 20011, from the reference site screening study in 1994 and again in this site vicinity in 1997. The 1997 sampling added two additional sampling stations from the end- and mid-creek gradient. Sampling stations are shown in Figure 3-1 (Section 3). A summary of BPTCP sediment toxicity and chemistry data is presented in Table 2-1.

The BPTCP indicated a high site ranking for aquatic life impact due to the incidence of recurrent sediment toxicity, with supportive evidence from sediment chemical and benthic community analysis. Significant toxicity was evident from the western creek sediments from initial and confirmation testing using amphipod survival and urchin development tests (Table 2-1).

Test sediments had high levels of hydrogen sulfide and unionized ammonia that may have contributed to the toxic response since they exceeded known toxicity threshold levels for these organisms reported by Knezovich et al. (1996) and EPA (1995). However, the BPTCP conducted a Toxicity Identification Evaluation (TIE) following the initial sampling and found significant toxicity remaining after hydrogen sulfide and ammonia removal, concluding that residual toxicity had to be due to other chemicals present. The source of the remaining toxicity was not identified, but was surmised to have possibly been due to polar organic compounds (S.R. Hansen & Associates, 1996). The San Francisco Bay BPTCP evaluates hydrogen sulfide and ammonia effects on a case-by-case basis when examining sediment toxicity data, stating that "Elevated ammonia and hydrogen sulfide may be the result of natural processes, or their concentrations may be enhanced by human activities, such as discharge of organically enriched wastewater. These compounds can therefore be considered as natural interference in toxicity tests (if of natural origin) or as pollutants in need of management (if anthropogenically enhanced)." (Hunt et al. 1998a).

Toxicity measured at Islais Creek BPTCP Station 20011 was associated with a mean ERM quotient of 1.18, which exceeded the adopted Regional BPTCP threshold standard of 0.5. Organic compounds noted as exceeding ERM thresholds were Chlordane, PCBs, Dieldrin and low molecular weight PAH. Chlordane and PCBs exceeded ERM guidelines by more than a factor of five. Screening phase (1994) toxicity testing of amphipod survival and urchin development was not conducted from mid- and end-creek gradient sites (BPTCP Stations 21303 and 213033), thus toxic hot spot designation was not possible, based upon the recurrent toxicity criterion. Tests in 1997 indicated significant, but reduced toxicity, compared with the western creek segment (Station 20011) and several chemicals exceeded ERM guidelines (Table 2-1), including the addition of mercury. Nickel was also listed, but, as noted by Long et al. (1995, 1998) and Calder et al. (1996), it is not considered to have a reliably based ERM and it has been generally excluded from regulatory concern in San Francisco Bay (see Hunt et al. 1998a).

Table 2-1. Islais Creek sediment toxicity and chemistry results from 1994 and 1997 BPTCP studies.

Station No./ID	Site Description	Date Sampled	Percent Fines <sup>1</sup>	Percent TOC <sup>2</sup>	Amphipod <sup>3</sup> % Survival	Urchin <sup>4</sup> Porewater % devel.	Urchin <sup>5</sup> SWI % devel.	ERM Quotient	Chemicals Exceeding ERM
20011/1411	Islais Creek	9/94	87.7	4.32	57 <sup>6</sup>	0 <sup>6,7</sup>	0 <sup>6,7</sup>	na	na
20011/1735	Islais Creek	4/97	38.7	3.99	0 <sup>8</sup>	8 <sup>6,7</sup>	8 <sup>6,7</sup>	1.18	Chlordane <sup>8</sup> , Dieldrin PCBs <sup>8</sup> , LMW PAH
21303/1736	Islais Creek mid-gradient	4/97	100	2.68	81	na	45 <sup>6</sup>	0.60	Mercury, Chlordane, ppDDE, PCBs
21304/1737	Islais Creek end-gradient	4/97	100	2.99	49	na	76 <sup>6</sup>	0.62	Mercury, Nickel Chlordane, PCBs

<sup>1</sup> fines = % (dry weight) of sediments smaller than 63 microns.

<sup>2</sup> TOC = total organic carbon, % sediment dry weight.

<sup>3</sup> Amphipod test organism = *Eohaustorius estuarius*

<sup>4</sup> Mean percent normal development of sea urchin larvae (*Strongylocentrotus purpuratus*) in 100% sediment porewater.

<sup>5</sup> Mean percent normal development of sea urchin larvae (*Strongylocentrotus purpuratus*) at the sediment water interface (SWI).

<sup>6</sup> Hydrogen sulfide concentrations exceeded known threshold toxicity levels reported by Knezovich et al. (1996) for amphipod (0.114 mg/L) and urchin development (0.0076 mg/L); lowest observed effects concentrations (LOEC).

<sup>7</sup> Unionized ammonia concentrations exceeded known threshold toxicity level reported by Knezovich et al. (1996) for amphipod (0.8 mg/L), lowest observed effects concentration (LOEC).

<sup>8</sup> more than 5 times guideline ERM value.

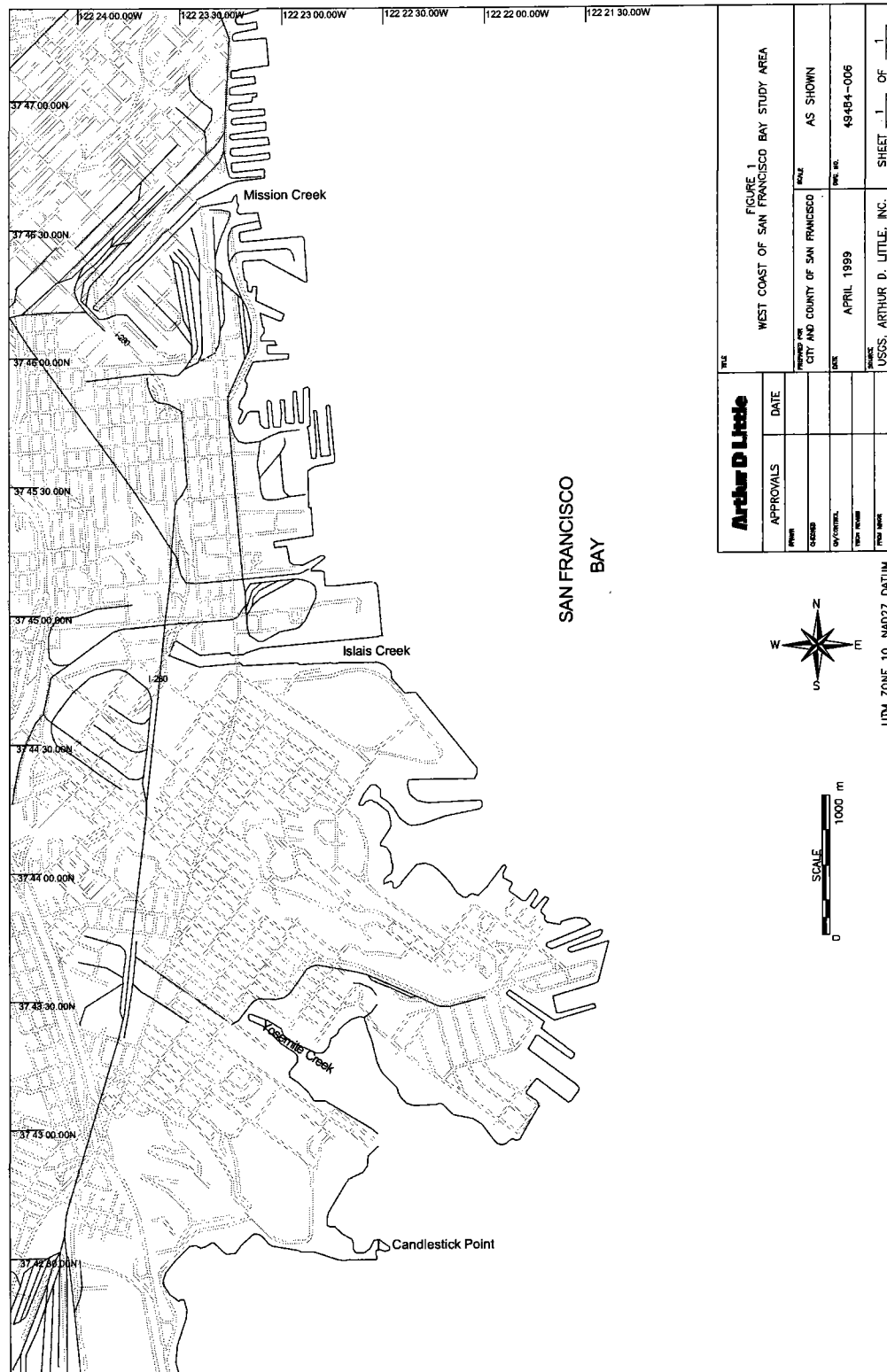


Figure 2-1. San Francisco Bay study area.



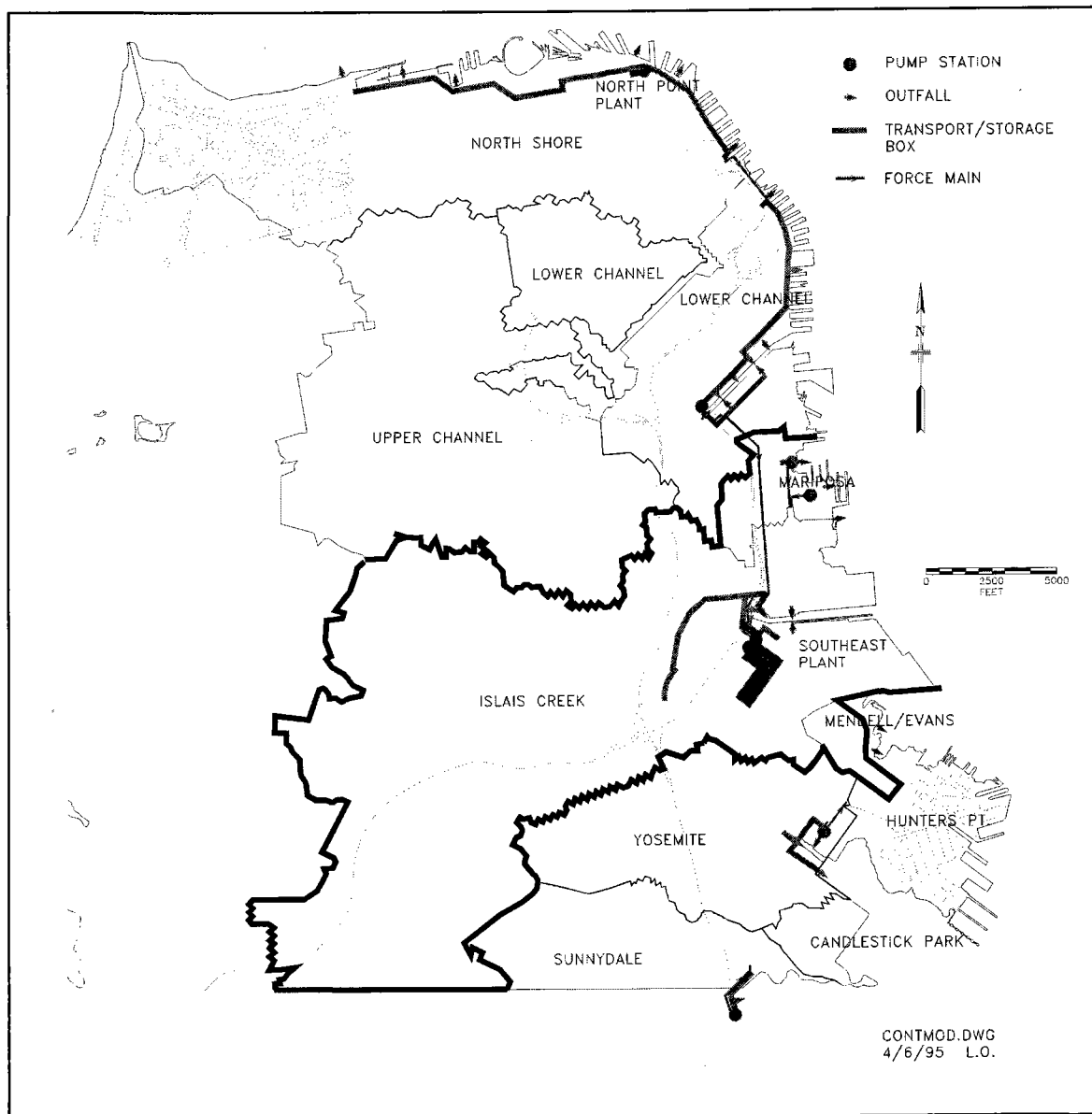


Figure 2-2. Major drainage basin for Islais and Mission Creeks.

## 2.2 MISSION CREEK

Mission Creek extends 1.2 km from its upper end, entering into the bay on the eastern San Francisco waterfront (Figure 2-1). The creek has been maintained in its present configuration for more than 100 years. Landfill of the watershed that extended further to the west took place prior to the turn of the century. The present Mission Creek drainage basin is pictured in Figure 2-2. Most creek sections are 10-60 m wide, with narrowing at the two bridges at 3rd and 4th Streets. Interstate 280 passes over the creek between 6th and 7th Streets.

A description of the present creek environment and surroundings provided by the Regional Board (RWQCB 1998), characterized the creek as follows: "...Concrete riprap and isolated vegetation line the creek banks." Houseboats with year-round residents are docked at the Mission Creek Harbor located between 5<sup>th</sup> and 6<sup>th</sup> Streets on the south shore. Light industrial and urban development surrounds the creek. Demolition debris is evident along the north shore at 2<sup>nd</sup> Street in China Basin. New retail development is planned for this area. On the south shore there are warehouse facilities, a sand-and-gravel operation near the creek mouth and a golf driving range near 6<sup>th</sup> Street.

During wet-weather storm periods, seven CSOs potentially discharge sanitary sewage water, industrial wastewater and storm water runoff into the creek. Discharge points range from 3<sup>rd</sup> Street to the upper end near 7<sup>th</sup> Street (Figure 3-2, Section 3). Two CSO pairs are located at 4<sup>th</sup> and 6<sup>th</sup> Streets, discharging at opposite shorelines. The remaining three CSOs at 3<sup>rd</sup>, 5<sup>th</sup> and 7<sup>th</sup> Streets discharge from the northern shoreline. The upper end CSO near 7<sup>th</sup> Street, often referred to as the Division Street overflow structure, is the major source of discharge, accounting for approximately 95% of the stormwater overflow entering the creek (RWQCB 1998). The construction of storage facilities in 1983 made it possible to treat storm water overflow to primary standards, removing significant amounts of wastewater settleable and floatable solids. These sewer storage boxes have acted to restrict ground water flow into the creek channel. These collection facilities are perpendicular to the groundwater flow direction, extending to a depth of nearly 6 m (20 feet) below ground surface along the length of the creek, collecting combined sewer and runoff overflows and acting as a barrier to groundwater seepage.

There are several historical sources of chemicals into soil and groundwater surrounding Mission Creek channel (Figure 2-3). ENVIRON (1998) tested soils and groundwater from a 238-acre area adjacent to the south shoreline of the creek. Groundwater was reported at 1.2-3 m below surface with seepage flow directed toward the creek channel. Tidal fluctuations influenced water levels in groundwater wells, indicating exchange with surrounding marine waters. Soils were found to contain trace levels of a few pesticides, volatile organic compounds, PAHs and other various organic chemicals, metals, asbestos and petroleum hydrocarbons. Sources include landfills from the early 1900s, spills and leaks from underground storage tanks, and releases from numerous industrial operations, including bulk oil storage, pipelines and transfer facilities. These all represent potential sources of contamination to the creek environment through seepage and runoff.

### 2.2.1 Historical Data

A 1979 study conducted by CH2MHill (1979) indicated elevated chemical concentrations and degraded benthic community conditions from sediments collected 20 m upstream of 6<sup>th</sup> Street. ERM guideline concentrations for copper, lead, mercury and nickel were exceeded and the RBI was zero, indicating a complete absence of benthic infaunal organisms.

Historical data were cited by the BPTCP as supportive of the 1998 candidate toxic hot spot designation. However, BPTCP 1994 screening and confirmation studies in 1995 and 1997 relied on data from only two locations to form the sole evidentiary basis for Mission Creek candidate toxic hot spot designation, as discussed below.

### 2.2.2 Mission Creek Toxic Hot Spot Designation

The upper end of Mission Creek in the vicinity of 6<sup>th</sup> Street was designated as a candidate toxic hot spot by RWQCB (1998) due to impacts upon aquatic life resulting from contaminated sediment, meeting the criteria prescribed in Condition 2 of the California Water Code candidate toxic hot spot definition (SWRCB 1998).

BPTCP data showed recurrent toxicity in both amphipod and sea urchin development tests at Station 21030 from the study in May 1995 and follow-up in April 1997 (Table 2-2). Amphipod survival was 5% and 19% for the two studies, respectively. Sea urchin development was completely unsuccessful in May 1995 porewater tests (0%) and 11% in the 1997 follow-up sediment-water interface exposure. Test sediments from Station 21030 had high levels of hydrogen sulfide and unionized ammonia that may have contributed to the toxic response since they exceeded toxicity threshold levels for these organisms reported by Knezovich et al. (1996) and EPA (1995). However, the BPTCP conducted a Toxicity Identification Evaluation (TIE) following the initial sampling and found significant toxicity remaining after hydrogen sulfide and ammonia were removed, concluding that residual toxicity had to be due to other chemicals present. The source of the remaining toxicity was not identified (S.R. Hansen & Associates 1996).

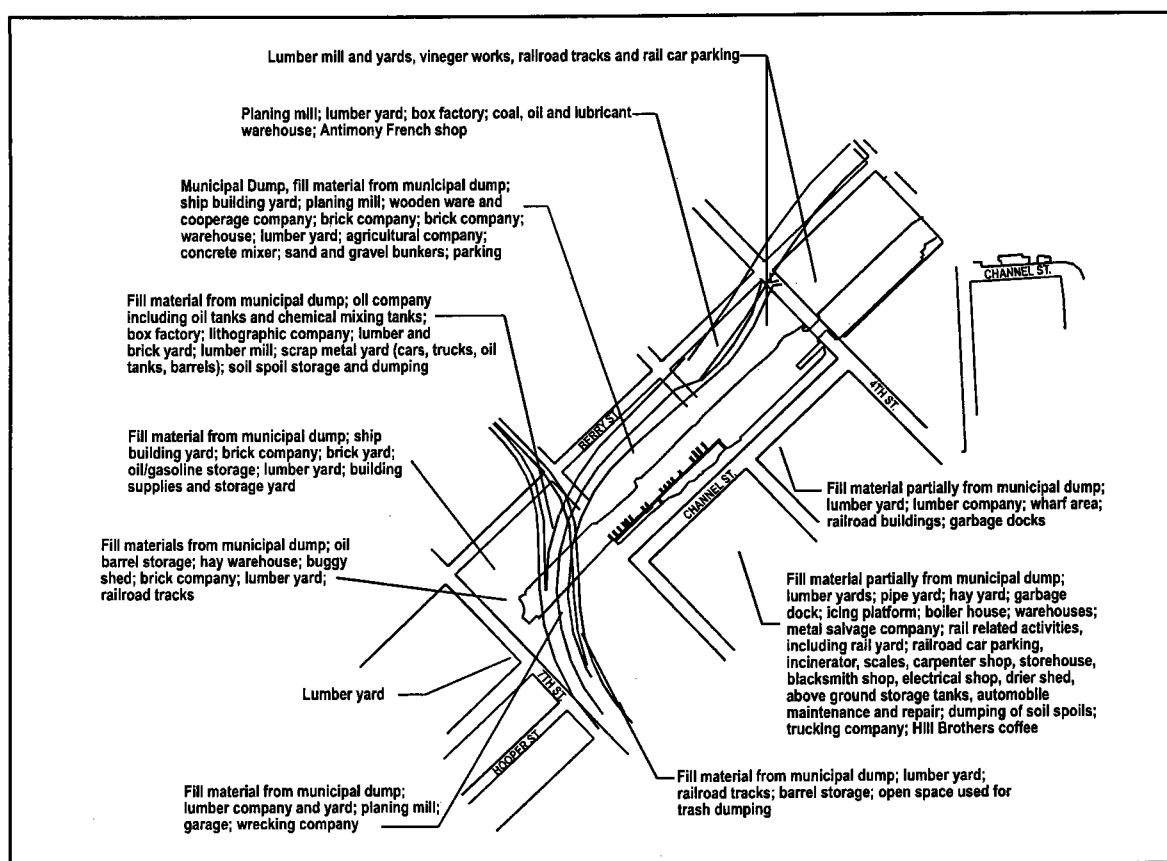


Figure 2-3. Industrial use at Mission Creek.

Table 2-2. Mission Creek BPTCP sediment toxicity and chemistry results.

Station No./ID	Site Description	Date Sampled	Percent Fines <sup>1</sup>	Percent TOC <sup>2</sup>	Amphipod <sup>3</sup> % Survival	Urchin <sup>4</sup> Porewater % devel.	Urchin <sup>5</sup> SWI % devel.	ERM Quotient	Chemicals Exceeding ERM
21030/1507	Mission Creek Site 1 (upper end)	5-2-95	7.22	1.02	5 <sup>6</sup>	0 <sup>6,7</sup>		0.51	Chromium, Lead, Nickel, Chlordane
21030/1732	Mission Creek Site 1 (upper end)	4-1-97	26.44	2.78	19		11 <sup>6</sup>	3.93	Chromium, Copper, Lead <sup>8</sup> , Mercury <sup>9</sup> , Silver, Zinc, Chlordane <sup>9</sup> , dieldrin, PCBs, phenanthrene, lmwPAHs, hmwPAHs
21031/1508	Mission Creek Site 2 (Creek Mouth)	5-1-95	97.72	1.44	83	57 <sup>6,7</sup>		0.22	Nickel
21301/1733	Mission Creek Mid-Gradient	4-1-97	100.00	2.71	58		98	1.00	Chlordane <sup>8</sup> , PCBs, hmwPAHs
21302/1734	Mission Creek End-Gradient (Near Mouth)	4-1-97	100.00	1.52	80		94	0.28	none

<sup>1</sup>fines = % (dry weight) of sediments smaller than 63 microns.

<sup>2</sup>TOC = total organic carbon, % sediment dry weight.

<sup>3</sup>Amphipod test organism = *Eohaustorius estuarius*.

<sup>4</sup>Mean percent normal development of sea urchin larvae (*Strongylocentrotus purpuratus*) in 100% sediment porewater.

<sup>5</sup>Mean percent normal development of sea urchin larvae (*Strongylocentrotus purpuratus*) at the sediment water interface (SWI).

<sup>6</sup>Hydrogen sulfide concentrations exceeded known threshold toxicity levels reported by Knezovich et al.(1996) for amphipod (0.114 mg/L) and urchin development (0.0076 mg/L); lowest observed effects concentrations (LOEC).

<sup>7</sup>Unionized ammonia concentrations exceeded known threshold toxicity level reported by Knezovich et al. (1996) for amphipod (0.8 mg/L), lowest observed effects concentration (LOEC), and urchin development (0.07 mg/L) no observed effects concentration (NOEC).

<sup>8</sup>more than 5 times guideline ERM value.

<sup>9</sup>more than 10 times guideline ERM value.

In support of the candidate toxic hot spot designation, BPTCP sediment chemical analyses indicated chemical concentration levels well in excess of ERM guidelines, as indicated in Table 2-2.

The toxicity from BPTCP Station 21030 (Site 1, near 6<sup>th</sup> Street) was associated with a mean ERM quotient of 0.51 in May 1995 and 3.93 in the follow-up survey. These values exceed the adopted Regional BPTCP threshold standard of 0.5. Organic compounds noted as exceeding ERM thresholds were Chlordane, PCBs, Dieldrin, phenanthrene and both low and high molecular weight PAHs (Table 2-2). Chlordane exceeded the ERM guidelines by more than a factor of ten. Significantly elevated metal concentrations were noted for chromium, lead, mercury, silver and zinc. Mercury exceeded the ERM guideline by more than a factor of 10. The BPTCP benthic community analysis for Station 21030 (upper-end, 6<sup>th</sup> Street) was zero in the 1997 sampling, associated with a mean ERM quotient of 3.93 (Table 2-2). This was cited as supportive evidence for the hot spot designation (RWQCB 1998).

An ERM quotient of 1.00 was evident in the 1997 confirmation phase from the mid-gradient site (BPTCP Station 21301, near 4<sup>th</sup> Street). Chlordane, PCBs and low molecular weight PAHs exceeded ERM guidelines. Chlordane exceeded the guideline by a factor of greater than five (Table 2-2). The

trace metal nickel also exceeded the ERM guideline, but as previously discussed, it was excluded from regulatory concern.

The evidence of contaminated sediments in Mission Creek prompted the RWQCB to define a preliminary assessment of actions required to remedy or restore the toxic hot spot to an unpolluted condition. Corrective actions included: 1) a requirement for a site investigation to delineate the horizontal and vertical extent of contamination along the creek, and relationship to CSOs; 2) preparation of a feasibility study for remedial action based upon site investigation findings; 3) implementation of remediation options from the feasibility study; and 4) follow-up monitoring to establish that the site has been cleaned up and remains clean.

### 3.0 DECISION RULES, STUDY DESIGN AND METHODS

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This section presents the decision rules and study design used to achieve program objectives described in Section 1.1. Decision rules were established for the interpretation of sediment chemistry, toxicity and tissue bioaccumulation data collected at each creek. The decision rules were presented in the Sampling and Analysis Plan (SAP) (ADL 1999) submitted to the RWQCB in November 1999. These rules were established since sediment quality criteria are not promulgated for chemicals of potential concern in San Francisco Bay.

The study design describes environmental analyses and corresponding sampling locations used to determine the spatial extent of environmental impact to sediments that have received and continue to receive combined effluent and stormwater discharges from City-operated CSOs (study objectives 3 & 4, Section 1.1). The analytical chemistry program featured ultra-trace measurements of organic and inorganic compounds consistent with methods and analyte lists used in the San Francisco Bay Regional Monitoring Program (RMP) and the Bay Protection and Toxic Cleanup Program (BPTCP). Modifications in BPTCP toxicity test protocols were made to improve data quality and minimize potentially confounding factors following EPA recommendations. Detailed method descriptions of field activities, laboratory and data analyses, including quality control procedures and criteria, are presented in the Sampling and Analysis Plans contained on the CD-ROM (*Sediment Investigations at Islais Creek and Mission Creek*), accompanying this report.

#### 3.1 DECISION RULES

The question of whether creek sediments are impacted and pose a threat to the ecology of San Francisco Bay relative to reference sediments is answered based on an evaluation of surface sediment chemistry, toxicity and bioaccumulation data. This weight-of-evidence approach extends the toxicity reference envelope, used in the BPTCP, to chemistry and bioaccumulation data. Test results are applied to the decision matrix shown in Table 3-1, which presents specific actions in response to results for each of the three data types, ranging from consideration for remedial or preventative action to no further action at the creeks. Significant impacts that are measured two or more times at any one station are used to confirm findings, following the BPTCP approach for confirmation of toxic hot spots. This approach differs from that used in the BPTCP, in that significant chemical impacts are defined statistically, rather than by exceeding a guideline value (such as an ERM). This approach proved to be more conservative, in that there were no samples that exceeded the ERM value that were not statistically elevated compared to reference concentrations (see Section 6). Bioaccumulation in clam tissue, which was measured in the April 2000 survey only, also is evaluated by comparing chemical tissue concentrations between creek and reference stations. Additionally, biota-sediment accumulation factors (BSAFs) are examined to identify creek sediments with the greatest bioaccumulation potential. This is a reasonable approach, since sediments are assumed to be the primary source of bioaccumulated contaminants.

In brief, chemistry and toxicity results from 1999 and 2000 surveys are compared on a station-by-station basis to an upper 95<sup>th</sup>-predictive limit calculated using corresponding reference data. Since only one reference station was sampled in 1998, data are evaluated following the approach used in the BPTCP (see Section 1.3), where any creek station with an ERM summary quotient greater than 0.5 and amphipod survival

less than 69.5% (of control), is considered potentially impacted. Only those stations that are significantly toxic and contaminated in two or more surveys are considered impacted and in need of further action.

Descriptions of data analysis and statistics used to support decisions are presented in the Sampling and Analysis Plan for the study (ADL 1999). Table 3-1 is an abbreviated decision matrix, which presents the most probable data outcomes. Unlikely outcomes that are not shown, such as significant bioaccumulation in the absence of elevated sediment chemistry, are addressed as they occur in Sections 5 through 7.

**Table 3-1. Decision rules used to assess environmental impact at each creek.**

Chemistry	Toxicity	Bioaccumulation <sup>1</sup>	Action
+	+	-	Consideration for remedial or preventative action; possible studies to determine ecological risk
+	-	-	No remedial action; continued monitoring
+	-	+	Possible studies to determine potential food web effects (ecological risk)
-	+	-	Possible studies to determine cause of toxicity
-	-	-	No further action
+	+	+	Candidate for remedial or preventative action

<sup>1</sup>=performed in April 2000 only; Pluses (+) denote significantly higher values in creek sediments compared to reference sediments for any single test for 2 or more years; Minuses (-) denote no significant differences between creek and reference sediments for 2 or more years.

## 3.2 STUDY DESIGN

Sediment sampling locations were chosen to measure the vertical and horizontal distribution of sediment chemical contaminants throughout each creek, and toxicity in corresponding surface samples. Locations that provided the basis for the toxic hot spot designation of the western segments of Islais and Mission Creeks (RWQCB 1998) were re-sampled in this program. Station transects in each creek extended perpendicular to shore in the vicinities of active and historical CSOs and storm drain locations (see Figures 3-1 and 3-2). Ecological impact was assessed by comparing creek results for each station to a threshold limit calculated using reference station results for the same survey.

Surface sediment chemistry and toxicity were measured in October 1998, October 1999, and April 2000 at each creek and at selected in-bay reference locations (Figures 3-1 through 3-3). Bioaccumulation tests using the clam, *Macoma nasuta* were conducted in April 2000 only. Tissues were analyzed for COPCs (identified in Section 6) known to biomagnify in the marine food web.

Subsurface sediment cores were collected in October 1998 to a nominal depth of 4 feet below the sediment surface and analyzed for bulk chemistry in 1-ft intervals. Sediments in all surveys were analyzed for 20 trace level polychlorinated biphenyl congeners (PCBs), 17 chlorinated pesticides, 41 polynuclear aromatic hydrocarbons (PAHs) - including alkylated homologs and 12 metals. April 2000 tissue samples were analyzed for mercury, PCBs and chlorinated pesticides. Total and resolved saturated hydrocarbons (SHC), linear alkylbenzenes (LABs) and PCB Aroclors also were measured in the October 1998 survey. Aroclors were measured for comparison with historical data. Saturated hydrocarbons were measured to help determine potential petroleum-related sources of co-occurring PAH. Linear alkylbenzenes were measured as



indicators of sewage-related contamination. These chemical tracers were used as ancillary data to identify potential sources of coexisting contaminants, and were not treated as potential COPCs. In addition, sediment grain size and total organic carbon (TOC) were measured to support interpretation of chemistry and toxicity data.

Acute toxicity was measured in surface sediments using the amphipod crustacean *Eohaustorius estuarius* exposed for 10-days in all three sampling events; however, not all stations sampled in 1998 for chemistry were tested for toxicity (Tables 3-3, 3-4, 3-6 and 3-7). In 1998 field samples were split and sent to SFPUC Oceanside Laboratory and Pacific EcoRisk (PER) for testing. This additional quality control measure was taken because the SFPUC laboratory had not previously conducted the 10-day amphipod test. There were no statistical differences between laboratory mean values reported for any of the 19 samples tested (5 replicates per sample,  $p < 0.05$ ). Only SFPUC results are reported for 1998, as the primary purpose of the duplicate analysis was to have backup data (from PER) in the event that SFPUC results did not pass quality control requirements. SFPUC conducted all toxicity tests in the 1999 and 2000 surveys. Conventional sediment parameters were assessed in each test to determine whether observed toxicity was attributable to natural products of organic degradation, such as ammonia and dissolved sulfides. Modifications to the BPTCP toxicity protocol consisted of 1) exchanges of overlying water both before and during (one per day) the test to reduce ammonia, and 2) press sieving of sediments prior to test initiation to remove potential resident predators.

Parameters measured and corresponding laboratories used throughout the program are presented in Table 3-2. Individual analytes and detection limits for each test are presented in Appendices A and B.

Surface sediment sampling inventories for each creek, detailing number of stations sampled and tests performed, are presented in Tables 3-3 and 3-4. Corresponding reference station information is shown in Table 3-7. Reference station location coordinates are shown in Table 3-6. Location coordinates for all samples collected are included in the project database (provided in Excel on the accompanying CD-ROM). Subsurface core descriptions are presented in Table 3-8 for all three creeks. A description of the sampling design used to collect surface sediment in each creek follows.

**Table 3-2. Summary of analytical methods and laboratories used in each survey.**

Parameter	Year Studied <sup>1</sup>	Laboratory	Analytical Method
<b><u>Sediment Chemistry</u></b>			
PAHs	1998-2000	ADL	EPA SW-846 8270 modified using SIM
PCB congeners & Pesticides <sup>2</sup>	1998-2000	ADL	EPA SW-846 8082 modified
Saturated Hydrocarbons (SHC)	1998	ADL	EPA SW-846 8015 modified
LAB	1998	ADL	EPA SW-846 8270 modified using SIM
Metals	1998-2000	SFPUC	EPA SW-846 6010 and 7000 series
Total Organic Carbon (TOC)	1998-2000	SFPUC	EPA SW-846 Method 9060
Grain Size	1998-2000	SFPUC	Plumb et al. 1981
<b><u>Bioassays</u></b>			
10-day solid phase amphipod	1998-2000	SFPUC/PER <sup>3</sup>	ASTM E1367-92 modified using EPA/USACE 1999 (PN 99-3)
28-day clam bioaccumulation	2000	EVS	EPA/USACE 1991
<b><u>Bioaccumulation in Clam Tissue</u></b>			
PCB congeners & Pesticides	2000	ADL	EPA SW-846 8082 modified
Mercury	2000	SFPUC/ Battelle	EPA SW-846 7460

<sup>1</sup>1998-2000 = October 1998, October 1999 & April 2000; <sup>2</sup>Aroclors also measured in 1998; <sup>3</sup>PER performed tests in 1998 only

### 3.2.1 Islais Creek – Surface Sediments

Eighteen stations in Islais Creek were sampled in 1998 and six were re-sampled in 1999 and 2000. Stations were sampled in six (1998) and three (1999 & 2000) cross-creek transects, extending from the west-end to the creek mouth (Figure 3-1). Stations located east of the 3<sup>rd</sup> Street Bridge (i.e., transects 4-6), were found to be unimpacted in 1998 and were not re-sampled in the following surveys. Stations sampled in 1999 and 2000 included all 1998 stations that had less than 68.5% survival in toxicity tests, as well as all stations with ERM quotients greater than 0.5. Sampling parameters are shown for each station for each of the three surveys in Table 3-3.

Table 3-3. Sampling inventory for surface sediments collected at Islais Creek.

Station	Metals	PAH	PCBs & Pesticides <sup>a</sup>	LAB, SHC & Aroclors	Grain Size/TOC	Toxicity
<b>October 1998</b>						
1C	1	1	1	1	1	
1N	1	1	1	1	1	1
1S	1	1	1	1	1	
2C	1	1	1	1	1	
2N	1	1	1	1	1	1
2S	1	1	1	1	1	
3C	1	1	1	1	1	
3N	3	3	3	3	3	
3S	1	1	1	1	1	1
4C	3	3	3	3	3	
4N	1	1	1	1	1	
4S	1	1	1	1	1	1
5C	1	1	1	1	1	1
5N	1	1	1	1	1	
5S	1	1	1	1	1	
6C	1	1	1	1	1	1
6N	1	1	1	1	1	
6S	1	1	1	1	1	
Total samples in 1998	22	22	22	22	22	6
Total stations in 1998	18	18	18	18	18	6
<b>Samples Collected October 1999 and April 2000<sup>1</sup></b>						
1N	1	1	1		1	1
1S	1	1	1		1	1
2N	1	1	1		1	1
2S	1	1	1		1	1
3N	1	1	1		1	1
3S	1	1	1		1	1
Total samples each year	6	6	6	0	6	6
Total stations each year	6	6	6	0	6	6

<sup>1</sup>PCBs, pesticides and mercury were also measured in clam tissue at each station in April 2000

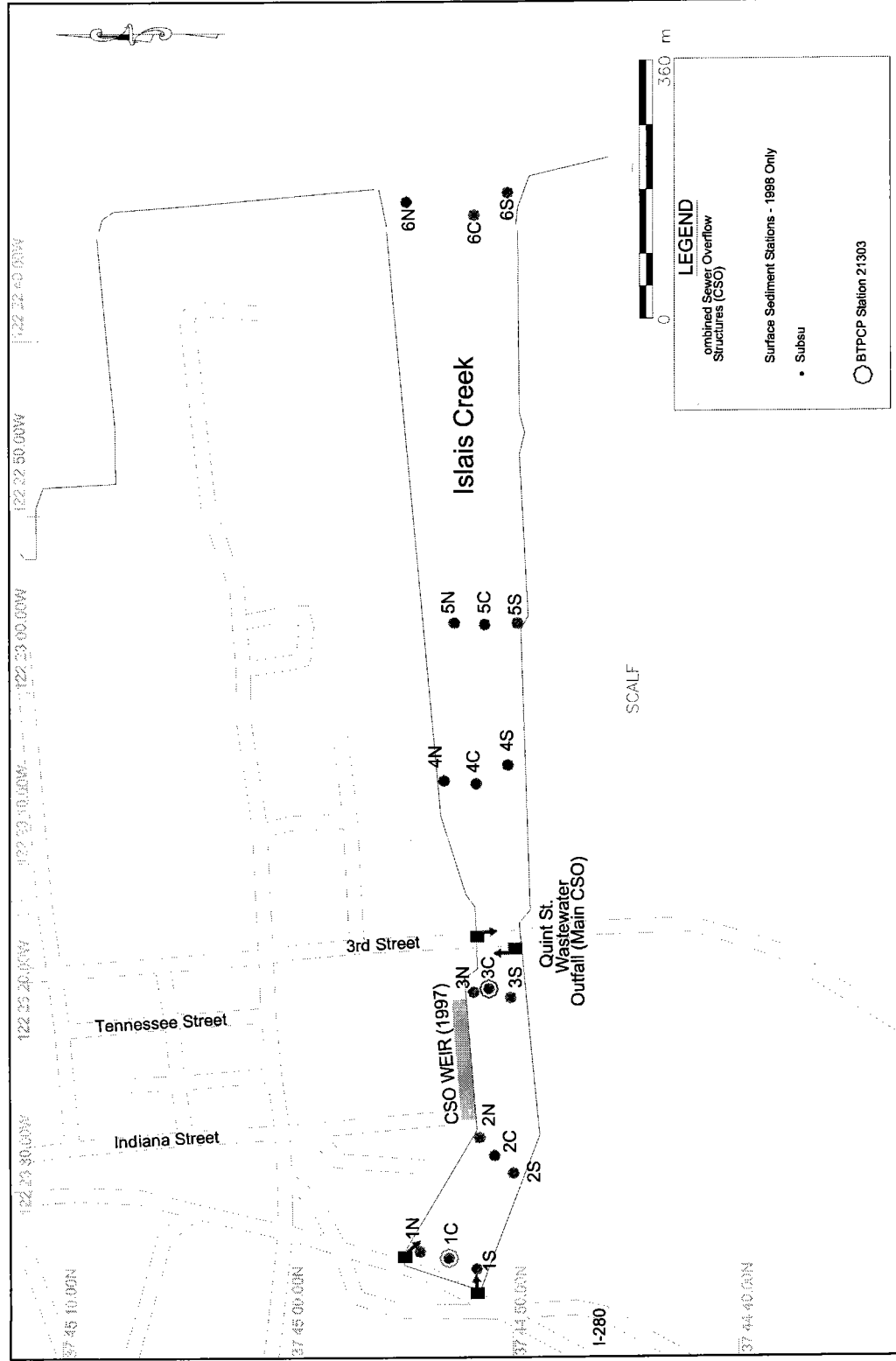


Figure 3-1. Islais Creek Sampling Stations.

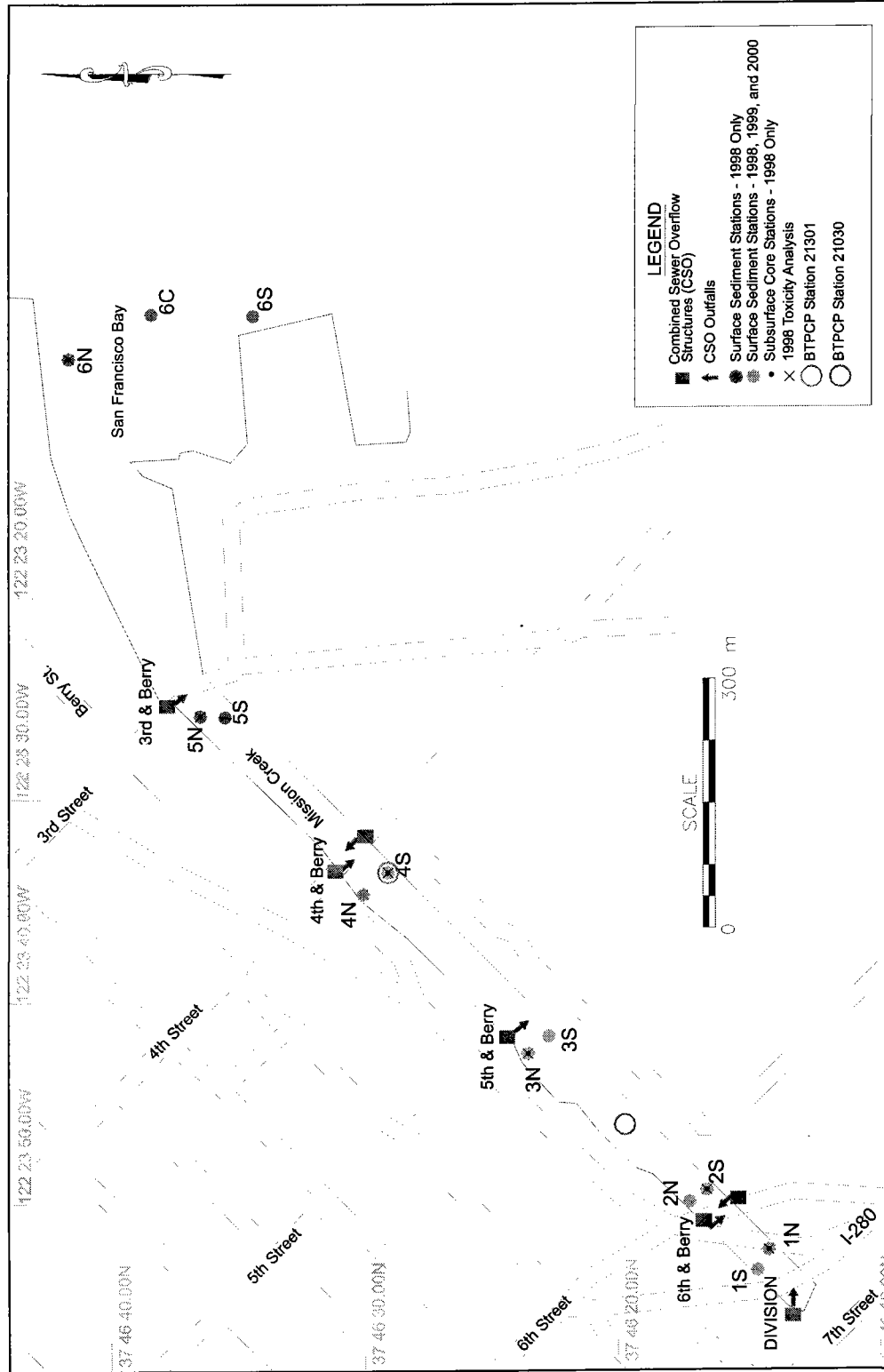
### 3.2.2 Mission Creek

Thirteen stations in Mission Creek were sampled in 1998 and eight were re-sampled in 1999 and 2000 (Figure 3-2). Six stations each from the cross-creek transects, extending from the west end to the creek mouth, were sampled for toxicity. The eight stations re-sampled in 1999 and 2000 extended from the main discharge CSO to the 4th Street intersection. Re-sampled stations included all 1998 stations that had ERM quotients greater than 0.5. There were no 1998 sediments with significant toxicity (i.e., < 69.5% control survival). Sampling parameters are shown for each station for each of the three surveys in Table 3-4.

**Table 3-4. Sampling inventory for surface sediments collected at Mission Creek.**

Station	Metals	PAH	PCBs & Pesticides	LAB, SHC & Aroclors	Grain Size/TOC	Toxicity
<b>October 1998</b>						
1N	1	1	1	1	1	1
1S	1	1	1	1	1	
2N	1	1	1	1	1	
2S	1	1	1	1	1	1
3N	1	1	1	1	1	1
3S	1	1	1	1	1	
4N	3	3	3	3	3	
4S	1	1	1	1	1	1
5N	1	1	1	1	1	1
5S	1	1	1	1	1	
6C	1	1	1	1	1	
6N	1	1	1	1	1	1
6S	1	1	1	1	1	
Total Samples	15	15	15	15	15	
Total Stations	13	13	13	13	13	6
<b>October 1999 and April 2000<sup>1</sup></b>						
1N	1	1	1		1	1
1S	1	1	1		1	1
2N	1	1	1		1	1
2S	1 (5) <sup>2</sup>	1	1 (5) <sup>2</sup>		1	1
3N	1	1	1		1	1
3S	1	1	1		1	1
4N	1	1	1		1	1
4S	1	1	1		1	1
Total Samples	8 (12)	8	8 (12)	0	8	8
Total Stations	8	8	8	0	8	8

<sup>1</sup>=PCBs, pesticides and mercury were also measured in clam tissue at each station in April 2000 only; <sup>2</sup>=Tissue laboratory replicates used in April 2000 only



### 3.2.4 Reference Area

A total of six reference stations were sampled throughout the program, although not all stations were sampled each year (see Tables 3-6 & 3-7). These locations, shown in Figure 3-4, extending from the south to north end of San Francisco Bay, consisted primarily of fine-grained sediments (i.e., >80%) with moderate organic carbon content (ca. 1%). Five of the sites were sampled previously in the Regional Monitoring Program (RMP) and/or BPTCP, and used to define the toxicity reference envelope (Hunt et al. 1998a). In the 1999 SFPUC survey, an additional reference station at Tomales Bay, located approximately 20 kilometers northwest of San Francisco Bay was included. This site was evaluated in the BPTCP but not used in the development of toxicity tolerance limits. It was sampled as a potential "fine-grained" reference site that had consistently produced high amphipod survival and low chemical concentrations in numerous dredge material disposal studies. It was not re-sampled in 2000, as the other in-bay reference stations adequately addressed "fine-grained" conditions.

The 1998 SFPUC survey, in contrast to following years, used only one reference station, Paradise Cove. A single reference location was considered adequate to address the initial study objective, which was to "confirm or refute BPTCP findings." Use of BPTCP toxic hot spot criteria eliminated the need for statistical comparisons between creek and in-bay reference sediments. Reference sites were expanded in 1999 and 2000 to provide background data sufficient to calculate corresponding reference envelopes relevant to each survey. This was considered necessary after reduced survival was observed in toxicity tests performed at Paradise Cove in 1998, in the absence of elevated chemical contaminants.

Although unimpacted, the in-bay reference stations are not well matched with the environmental conditions of the creeks under investigation, due to differences in grain size/mineralogy, total organic carbon, hydrodynamics and other conditions (e.g., temperature, depth, salinity). Any of these factors can affect the parameters of interest, potentially confounding interpretation of results. These stations were used because of their established history within the BPTCP and RMP, and the lack of other suitable reference locations that may have better represented creek conditions. Since creek and reference sediments are not well-matched, chemistry results are normalized to minimize effects that may be due to physical characteristics. This is a common approach that is used to correct disparities between test and reference areas that are independent of contaminant inputs. Chemical results were normalized using total organic carbon, since it is known to have a significant influence on sediment contaminant concentrations and associated toxicity (Di Toro 1991; Schwartz et al. 1984).

Table 3-5. Reference Area surface sediment sampling locations.

Location	BPTCP Station ID	Latitude (N) <sup>1</sup>	Longitude (W) <sup>1</sup>	Location Description
Paradise Cove	20005	37° 53' 57.00"	122° 27' 51.60"	Central San Francisco Bay
Tubbs Island	20006	38° 06' 52.20"	122° 25' 09.60"	San Pablo Bay
Island #1	20007	37° 06' 43.20"	122° 19' 42.60"	San Pablo Bay
North Site	20013	37° 34' 13.80"	122° 08' 58.50"	South San Francisco Bay
South Site	20014	37° 32' 10.80"	122° 07' 09.60"	South San Francisco Bay
Marconi Cove	20009	38° 08' 21.60"	122° 52' 27.60"	Tomales Bay

<sup>1</sup> Station coordinates shown in NAD 83 datum

Table 3-6. Sample inventory for surface sediments collected at Reference Area.

Station	Metals	PAH	PCBs & Pesticides	LAB, SHC & Aroclors	Grain Size/TOC	Toxicity
<b>October 1998</b>						
Paradise	1	1	1	1	1	1
Total Stations	1	1	1	1	1	1
<b>October 1999</b>						
Island#1	1	1	1		1	1
Marconi Cove	1	1	1		1	1
North Site	1	1	1		1	1
Paradise	1	1	1		1	1
South Site	1	1	1		1	1
Tubbs Island	1	1	1		1	1
Total Stations	6	6	6	0	6	6
<b>April 2000<sup>1</sup></b>						
Island#1	1	1	1		1	1
North Site	1	1	1		1	1
Paradise	1	1	1		1	1
South Site	1	1	1		1	1
Tubbs Island	1	1	1		1	1
Total Stations	5	5	5	0	5	5

<sup>1</sup> PCBs, pesticides and mercury were measured in clam tissue in April 2000



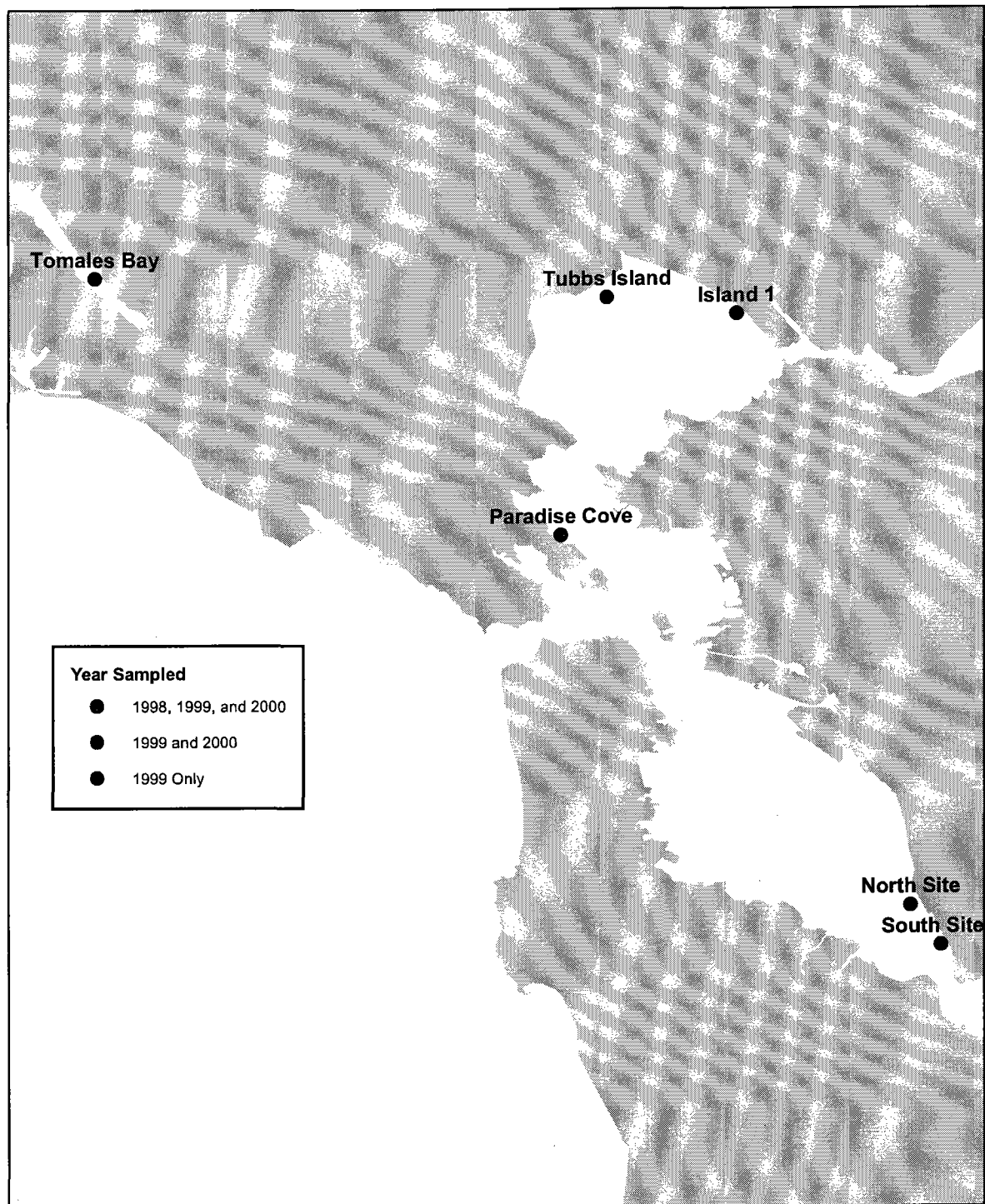


Figure 3-3. San Francisco Bay reference sites.

### 3.2.5 Subsurface Sediments

Five to six subsurface cores were collected in October 1998 from each creek that penetrated a maximum depth of 4 feet. The top two 1-ft core intervals (i.e., 0-1 and 1-2 ft) were analyzed for bulk chemistry, grain size and total organic carbon. The remaining core intervals (i.e., 2-3 ft and 3-4 ft) were stored frozen until they were analyzed in 1999 for bulk chemistry only. Core locations, which corresponded with surface sediment stations, are shown for each creek in Figures 3-1 through 3-3. Cores were not collected at any of the in-bay reference stations. Subsurface data were collected to determine whether significant vertical contaminant gradients exist in each creek. Gradients are tested using linear regression models for each chemical of concern (see Section 6). Statistically significant results ( $p < 0.01$ ) are used to verify that buried sediments are "in-place", and contaminants are not being resuspended or re-released into the bay.

### 3.2.5 Summary of statistical comparisons between creek and reference stations

Individual comparisons are made for each station within each creek and year sampled using a group tolerance limit, to produce a "reference envelop" for each parameter evaluated. Since this method requires group replicates, only reference data collected in 1999 and 2000 were used. For the 1998 survey, toxicity and chemistry data were compared to the established BPTCP reference envelop toxicity criterion and corresponding ERM, respectively.

The probit method ( $\phi = 0.85$ ;  $\alpha = 0.05$  for one-sided test) used in the BPTCP (see Hunt et al. 1998a) was used to calculate the toxicity tolerance limit. A 95<sup>th</sup> percent one-sided predictive limit ( $\alpha = 0.05$  for one-sided test) was calculated for chemical parameters measured in reference surface sediments. The predictive interval is a modification of the confidence interval and is used when comparing individual results to grouped data (see Steel and Torrie 1960). Nonparametric tolerance interval bounds were used (Hahn and Meeker 1991) for chemical data that failed test assumptions for the predictive limit (e.g., non-normally distributed data). A lower tolerance limit was calculated for reference survival (to identify stations more toxic than reference); and an upper predictive limit was calculated for reference chemistry (to identify stations more contaminated than reference).

Table 3-7. Sample inventory for subsurface sediments collected in October 1998.

Creek/Station	Core Intervals <sup>1</sup>	Analyses <sup>2</sup>	Comments
<b>Islais Creek</b>			
1C	1, 2, 3, 4	Grain size, PCB, pesticides, PAH, metals, TOC	
2N	1, 2, 3, 4	Grain size, PCB, pesticides, PAH, metals, TOC	
3S	1, 2, 3, 4	Grain size, PCB, pesticides, PAH, metals, TOC	
4S	1, 2	Grain size, PCB, pesticides, PAH, metals, TOC	2-3 & 3-4 ft cores collected but not analyzed
5C	1, 2	Grain size, PCB, pesticides, PAH, metals, TOC	"
6C	1, 2	Grain size, PCB, pesticides, PAH, metals, TOC	"
<b>Mission Creek</b>			
1N	1, 2, 3	Grain size, PCB, pesticides, PAH, metals, TOC	3-4 ft core not collected due to refusal
2S	1, 2, 3, 4	Grain size, PCB, pesticides, PAH, metals, TOC	
3N	1, 2, 3, 4	Grain size, PCB, pesticides, PAH, metals, TOC	
4S	1, 2, 3, 4	Grain size, PCB, pesticides, PAH, metals, TOC	
5N	1, 2	Grain size, PCB, pesticides, PAH, metals, TOC	2-3 & 3-4 ft cores collected but not analyzed
6N	1, 2	Grain size, PCB, pesticides, PAH, metals, TOC	"

<sup>1</sup>=core interval 1=1-2 ft, 2=2-3 ft, 3=3-4 ft, 4=4-5 ft; <sup>2</sup>=Grain size & TOC - measured in 1-2 & 2-3 ft intervals only

### 3.3 METHODS SUMMARY

Abbreviated field and analytical methods follow. Detailed method descriptions for sample collection, handling, laboratory, data analysis and quality control are presented in the Sampling and Analysis/Quality Assurance Plans for each survey are contained in the CD-ROM accompanying this report.

#### 3.3.1 Field Methods

Surface sediments were collected with a 0.05-m<sup>2</sup> Ponar grab sampler, constructed of stainless steel and coated with Halar to reduce cross-contamination. A sufficient number of grabs (4-5) were collected at each station to ensure adequate sediment for testing. Surface sediment was subsampled from the top 5 cm of each grab and homogenized in a Halar-coated bucket. Subsurface sediments were sampled using a gravity corer with a butyrate liner. The liners were capped and sediments were sub-sectioned into 1-ft intervals and homogenized in SFPUC's Oceanside Laboratory prior to subsampling. Organic chemistry samples were placed in borosilicate glass jars; metal samples were placed in polycarbonate jars and TOC and grain size samples were stored in plastic bags. All samples were stored on ice and transferred within 48 hours from the vessel to SFPUC's Oceanside Laboratory for subsequent shipment or analysis.

#### 3.3.2 Laboratory Methods

All samples were analyzed using standard analytical methods referenced in individual laboratory standard operating procedures (SOPs). Quality control samples for laboratory and field samples were analyzed. Laboratory quality control samples consisted of calibration standards, matrix spikes, duplicate

samples, standard reference materials, surrogates, and laboratory blanks where appropriate. Table 3-8 lists chemistry, toxicity and physical tests, and analytical laboratories for the program.

**Table 3-8. Summary of sediment analytical methods and laboratories.**

Parameter	Laboratory	Analytical Method
<b><u>Chemistry</u></b>		
Polynuclear Aromatic Hydrocarbons (PAH)	ADL	EPA SW-846 8270 modified using SIM
Polychlorinated Biphenyl Congeners (PCBs) & Chlorinated Pesticides	ADL	EPA SW-846 8082 modified for congener analysis
Metals	SFPUC	EPA SW-846 6010 and 7000 series
Total Organic Carbon (TOC)	SFPUC	EPA SW-846 Method 9060
Grain Size	SFPUC	Plumb et al. 1981
<b><u>Toxicity</u></b>		
10-day solid phase amphipod	SFPUC <sup>1</sup>	ASTM E1367-92 modified using EPA/USACE 1999 (PN 99-3)

<sup>1</sup>=Pacific EcoRisk Laboratory also analyzed samples in 1998 survey

### 3.3.2.1 Physical Laboratory Methods

Sediment grain size was analyzed using a sieve and pipette method by SFPUC, which produced results for four grain size classes (gravel, sand, silt and clay). Results reported for 1998 samples only, included mean diameter, percent sediment contribution for each of 16 size classes, Phi sorting coefficient, skewness and kurtosis. Percent gravel, sand, silt and clay only were reported for 1999 and 2000 data. Total organic carbon (TOC) was analyzed by SFPUC using EPA Method SW-846 9060, combustion followed by infrared detection of carbon dioxide, and reported as a percentage of total sediment dry weight.

### 3.3.2.2 Chemical Laboratory Methods

Sediment hydrocarbon analyses consisting of polycyclic aromatic hydrocarbons (PAH), polychlorinated biphenyl (PCB) congeners, and chlorinated pesticides were analyzed by ADL's Environmental Laboratory. Additionally, saturated hydrocarbons (SHC) and linear alkylbenzenes were analyzed for source identification purposes in the 1998 survey only. A total of 41 PAH compounds were measured using gas chromatography with mass spectrometer - selected ion monitoring (SIM). PCBs were measured as 22 congeners in all surveys and additionally as Aroclors in 1998 only. Dry weight detection limits for organic analytes were all in the sub-part-per-billion range, ranging from 0.01 ng<sub>g</sub><sup>-1</sup> for pesticides and PCBs to 0.1 ng<sub>g</sub><sup>-1</sup> for PAHs.

Table 3-9 shows detection limits and corresponding methods for the 12 heavy or trace metals measured throughout the investigation. Sediment metals were analyzed by SFPUC using nitric acid and hydrochloric acid digestion followed by inductively coupled plasma spectroscopy (ICP), or atomic

absorption with either a flame or graphite furnace detector, except mercury, which was analyzed using atomic absorption following cold vapor extraction.

**Table 3-9. Methods and detection limits for metals ( $\mu\text{g g}^{-1}$  dry weight).**

Metal	Minimum Detection Limit	Analytical Method*
Aluminum (Al)	0.2/0.01	ICP/AAGF
Arsenic (As)	0.5	ICP
Cadmium (Cd)	1.0/0.025	ICP/AAH
Chromium (Cr)	0.1/0.01	ICP/AAGF
Copper (Cu)	0.2	ICP
Iron (Fe)	0.2	ICP
Mercury (Hg)	0.3	ICP
Nickel (Ni)	0.0005	CVAA
Lead (Pb)	0.2	ICP
Selenium (Se)	1.0/0.07	ICP/AAGF
Silver (Ag)	0.025	AAH
Zinc (Zn)	0.1	ICP

AAH = Atomic absorption hydride; ICP= Inductively coupled plasma emission spectroscopy; AAGF= Atomic absorption with graphite furnace; CVAA = Cold vapor atomic absorption

### 3.3.2.3 Toxicity Laboratory Methods

The acute 10-day amphipod test was performed by SFPUC following ASTM E1367-92 modified following EPA/USACE guidelines in Public Notice 99-3 to remove potential confounding toxicity from elevated levels of ammonia and/or hydrogen sulfide. All test sediments were press-sieved (through 0.5 mm mesh stainless steel screens) and picked to remove possible amphipod predators and native amphipods prior to test initiation. Eighty percent (80%) of overlying water was exchanged and allowed to equilibrate for 24-hours for all sediment samples with ammonia porewater values greater than  $20 \text{ mg L}^{-1}$  prior to test initiation. Dissolved oxygen, pH, salinity and temperature were measured and recorded daily. After 10-days of exposure, amphipods were carefully removed by wet-sieving, counted, placed on clean sediment and permitted to rebury. Percent survival and percent reburial were reported for each of the five laboratory replicates run for each sample. The test was considered valid if after ten days of exposure the average control survival was  $\geq 90\%$  and each control replicate had at least 80% survival.

### 3.3.2.4 Bioaccumulation Laboratory Methods

A 28-day clam bioaccumulation test was undertaken to evaluate the potential for chemical uptake and subsequent food chain transfer. Bioaccumulation of certain organic chemicals and metals is known to occur across trophic levels. The test animal, *Macoma nasuta* is widely distributed and native to San Francisco Bay, commonly used in dredged sediment studies, known to actively ingest surface sediments, and provides enough tissue for trace level tissue analysis. Laboratory bioaccumulation was performed following the EPA/USACE (1991) "Greenbook" protocol, modified to use one laboratory "replicate" instead of five as recommended. One composite sample of 25 clams was analyzed for chemistry at all stations except Station 2N in Mission Creek, to make use of the highly replicated sampling design (i.e., 6 stations in Islais & 8 stations in Mission). Station 2N was tested using the standard five laboratory

replicates as a quality control measure. The reduced number of laboratory replicates for the remaining samples (i.e.,  $n=1$  instead of  $n=5$  in EPA/USACE [1991]) was validated, as chemistry results for the five laboratory replicates were extremely consistent (i.e.,  $CV < 20\%$  for all analytes). Laboratory control samples (e.g., zero time) also were analyzed for quality control purposes. Results were reported based on dry weight, wet weight and lipid weight. Only dry weight results are presented in Appendices A3 and B3.

## **4.0 PHYSICAL CHARACTERISTICS OF SEDIMENT**

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Grain size and total organic carbon (TOC) results for surface and subsurface sediments are presented in this section. These physical parameters are known to influence contaminant distribution and amphipod toxicity in sediment, and are therefore important in the interpretation of data. Grain size categories are summarized in Table 4-1. Interpretation of results focuses primarily on surface and near-surface sediments due to their influence on resident biota and contaminant bioavailability. For this reason, physical parameters were measured only at the surface and in the top two core intervals (i.e., 0-1 and 1-2 ft). Surface sediment results for each creek and reference area are summarized in Tables 4-2 through 4-3. Surface distributions of percent fines (silt + clay) and TOC are shown in Figures 4-1 through 4-4 for each creek. Surface sediment results are presented in Appendices A1 and B1 for each station. Results for core intervals are presented in Appendices A2 and B2.

Sediment grain size characteristics are emphasized for their controlling influence upon benthic community dynamics, and because they correlate with biologically meaningful variables such as sediment porosity, compaction, oxygen tension, water content and retention of organic matter. Grain size characteristics are equally important in controlling sediment chemical concentrations due to an increase in adsorptive capacity with finer-grained particles. Total organic carbon concentrations provide an indication of the amount of organic matter present in sediment. High organic content is typical of fine-grained sediments from low-energy depositional areas and areas impacted by anthropogenic activities, such as discharges from sewage outfalls. High levels of organic carbon also occur naturally in sediments from detrital inputs from terrestrial and aquatic plants.

Most studies of marine and brackish sediments show a high positive correlation between fine-grained particles and organic carbon. Since contaminants are strongly bound to organic particles that are complexed with fine mineral particles, there is a high potential for contaminant accumulation in habitats where settlement of finer-grained, organically enriched sediment occurs. Deposition, resuspension and sorting processes influenced by the nearshore wave and current regime normally create a gradient of diminishing grain size proceeding offshore. As they are introduced into the coastal system, the smallest particles remain in suspension for longer periods and, following deposition, are more readily re-suspended from the seabed by waves, currents and turbidity flows. Ultimately, fine-grained sediments progress offshore into the deeper stable basins that are the ultimate repositories for contaminants.

### **4.1 OVERVIEW**

Most creek sediments were collected from less than 10 m water depth and consisted of greater than 90% fine-grained material. Predominantly fine-grained sediments were evident in each of the two creeks (see Figures 4-1 & 4-3). However, gradients of increasingly fine-grained material with distance from Islais and Mission Creek ends were observed. These trends more than likely occurred because the necessary energy to move fine-grained particles toward the bay was provided from storm-related flow.

A full suite of grain size parameters including mean grain size for 12 classes, Phi sorting coefficient and percent gravel, sand, and fines (silt + clay) were reported for 1998 surface sediment data only. Percent fines and sand were the only grain size metrics reported in 1999 and 2000.

Sediment sorting (expressed as the "mean Phi sorting coefficient" in standard deviation units) influences pore space and water retention in sediment. Poorly sorted sediment (i.e., Phi sorting coefficients >2 standard deviation units) typically has reduced pore space and water retention compared to well-sorted sediment, and is generally more efficient in binding contaminants. Sediments collected in 1998 east of the mid-section of each creek were moderately well sorted (Phi sorting coefficient <1.5), where sediments collected near active CSOs were typically poorly sorted.

**Table 4-1. Sediment grain size classes (adapted from Folk 1968).**

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

— denotes criterion (i.e., < 0.0625 mm) for fines (silt + clay)

Percent fines and TOC were not strongly correlated in any of the creek sediments (i.e.,  $r^2 < 0.2$ ). The strong correlation between TOC and percent fines typically observed in marine sediments was undermined primarily because of the high fraction of coarse-grained material (sands) associated with relatively high concentrations of TOC at the end of each creek. Likely sources of coarse material at both creeks include active CSOs, construction fill from concrete structures at the creek end, and debris from the Interstate 280 overpass. Concrete debris and large rocks were observed in runoff from Interstate 280 into Mission Creek during the October 1998 field sampling.

## 4.2 REFERENCE AREA

Most reference area samples had greater than 90% fines (mean=79.2%) that remained consistent across surveys, except for North Site sediments (Figure 3-3, Section 3), which were sandy in 2000 (26.3% fines) yet fine-grained in 1999. South Site sediments were consistently coarser with less than 60% fines in two consecutive samplings. In general, reference site sediments had similar grain size distributions



compared to most creek sediments. Only a few sediment samples collected at the end of Islais and Mission Creeks were consistently coarser-grained than the reference area. Concentrations of total organic carbon in reference sediments ranged from 0.4 to 1.8% (mean=0.9%) for all three sampling events. These concentrations were significantly lower than TOC concentrations measured in most creek sediments, especially those located near active CSOs (which approached 4%). Reference surface sediment results are summarized in Table 4-2. There were no subsurface cores collected at any of the reference sites.

It is important that reference site sediment attributes, such as grain size and organic carbon content, are similar to those found at the creeks. These attributes can affect test results, including toxicity and chemistry, as discussed in Section 1. In particular, grain size and organic carbon affect adsorption and retention of sediment contaminants and their subsequent bioavailability. Many creek sediments had significantly higher concentrations of TOC compared to reference area sediments (see Tables 4-2 through 4-3). In order to “normalize” potential differences in bioavailability, contaminant concentrations are based on grams of organic carbon (OC) instead of grams dry sediment for all comparisons between creek and reference sediments (see Section 6). This approach is consistent with methods used to address varying TOC concentrations in other sediment investigations (Schwartz et al. 1994; MacDonald 2000).

### 4.3 ISLAIS CREEK

With the exception of sandy sediments that characterized the creek end (Figure 4-1), surface and subsurface sediments were characterized by fine (silt + clay) fractions exceeding 90% of sediment dry weight. Percent fines were variable in creek surface sediments, ranging from 5.4 to 99.48% (mean=85.8%); however, distribution patterns were consistent between study years (Figure 4-1). Transect 1 stations, located at the creek end near the historical main CSO, were much more variable and had significantly coarser sediments compared to sediments east of the CSO Weir. Percent fines at Station 1S (located at the creek end) ranged from 5.4 to 96.3% fines across sampling events. Subsurface samples collected at Station 1C were notably lower in percent fines, ranging from 38.1% at the 1-2 ft interval to 62.2% in the upper segment (0-1 ft). The mean and range of percent fines in creek and reference area surface sediments are shown in Table 4-2 for each year sampled.

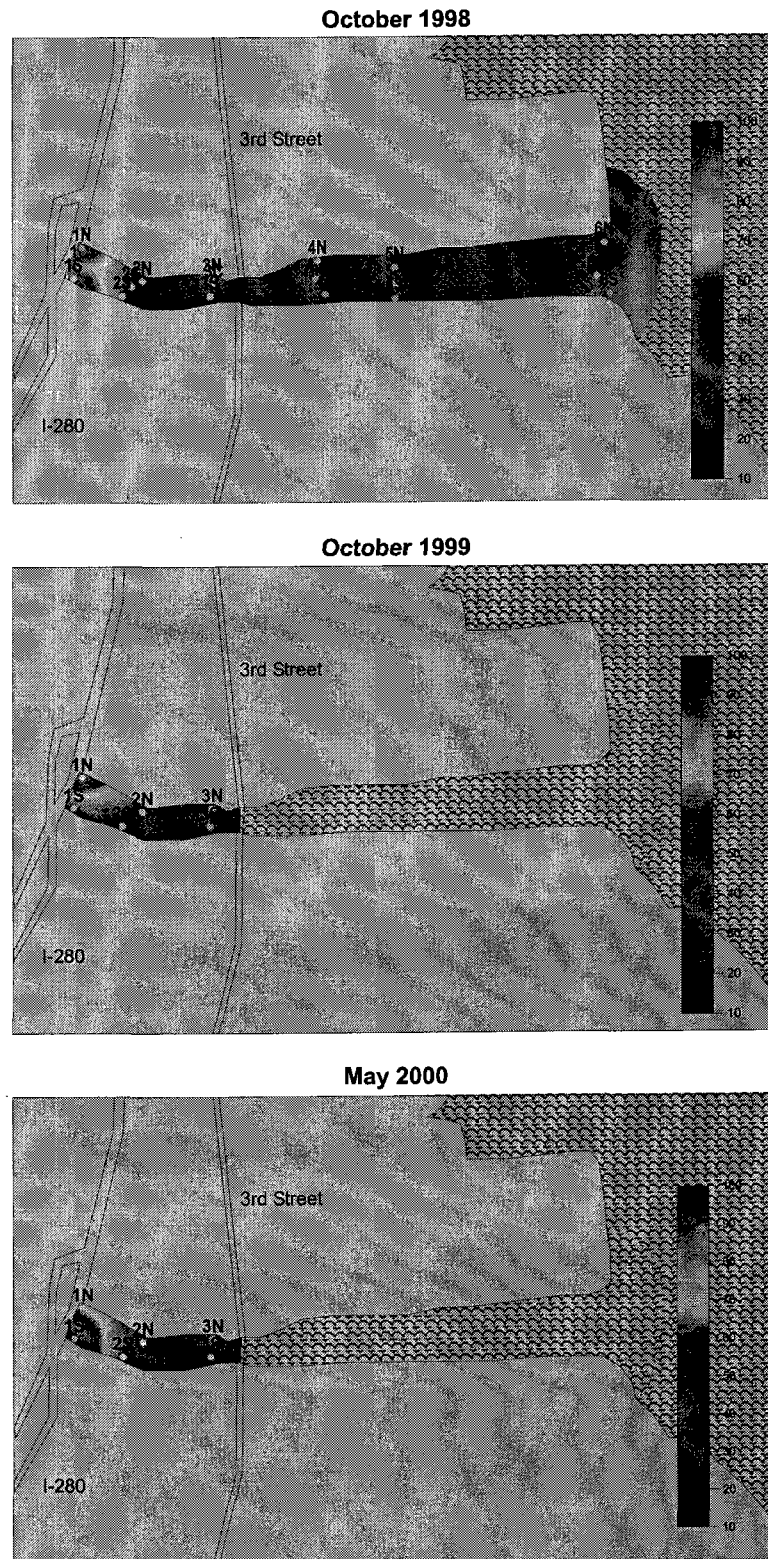
**Table 4-2. Islais Creek - percent fines and TOC in surface sediments.**

Year Sampled	Area Sampled	No. of Stations	Mean % Fines	Range % Fines	Mean % TOC	Range % TOC
1998	Islais Creek	18	87.2	5.4 - 98.9	2.0	1.2 - 4.8
	Reference Sites	1	90.3	90.3 - 90.3	1.2	1.2 - 1.2
1999	Islais Creek	6	87.7	38.1 - 99.4	1.9	1.2 - 2.5
	Reference Sites	6	80.9	30.7 - 99.7	0.9	0.4 - 1.8
2000	Islais Creek	6	79.4	13.2 - 99.1	2.1	1.3 - 4.4
	Reference Sites	5	74.8	26.3 - 97.9	0.9	0.5 - 1.2
All	Islais Creek	30	85.8	5.4 - 99.4	2.0	1.2 - 4.8
	Reference Sites	12	79.2	26.3 - 99.7	0.9	0.4 - 1.8

na=not applicable; \*Fines = silt+clay, < 0.063 mm diameter;

Total organic carbon results are summarized in Table 4-2 for creek and reference area surface sediments. Although chemical exchange processes in Islais Creek sediments have not been investigated, high organic content combined with low overlying oxygen levels from limited water circulation may present a reducing environment in creek sediments. As an indication of minimal oxygen tension, sediments collected at the creek end were darkened, suggesting a reducing environment that may be largely responsible for the reduced biological diversity of benthic infauna observed in previous studies (Hunt et al. 1998a).

Concentrations of total organic carbon ranged from 1.2 to 4.8% (mean=2.0%) in creek surface sediments, and from 0.4 to 1.8% (mean=0.9%) in reference sediments for all three sampling events. Total organic carbon concentrations were moderate to high (>1 to ~5%) for sediments collected west of the 3<sup>rd</sup> Street Bridge, and primarily low (<1%) at stations east of the bridge. However, higher organic carbon concentrations were not always associated with finer-grained sediments.



**Figure 4-1. Percent fines (silt + clay) in Islais Creek.**

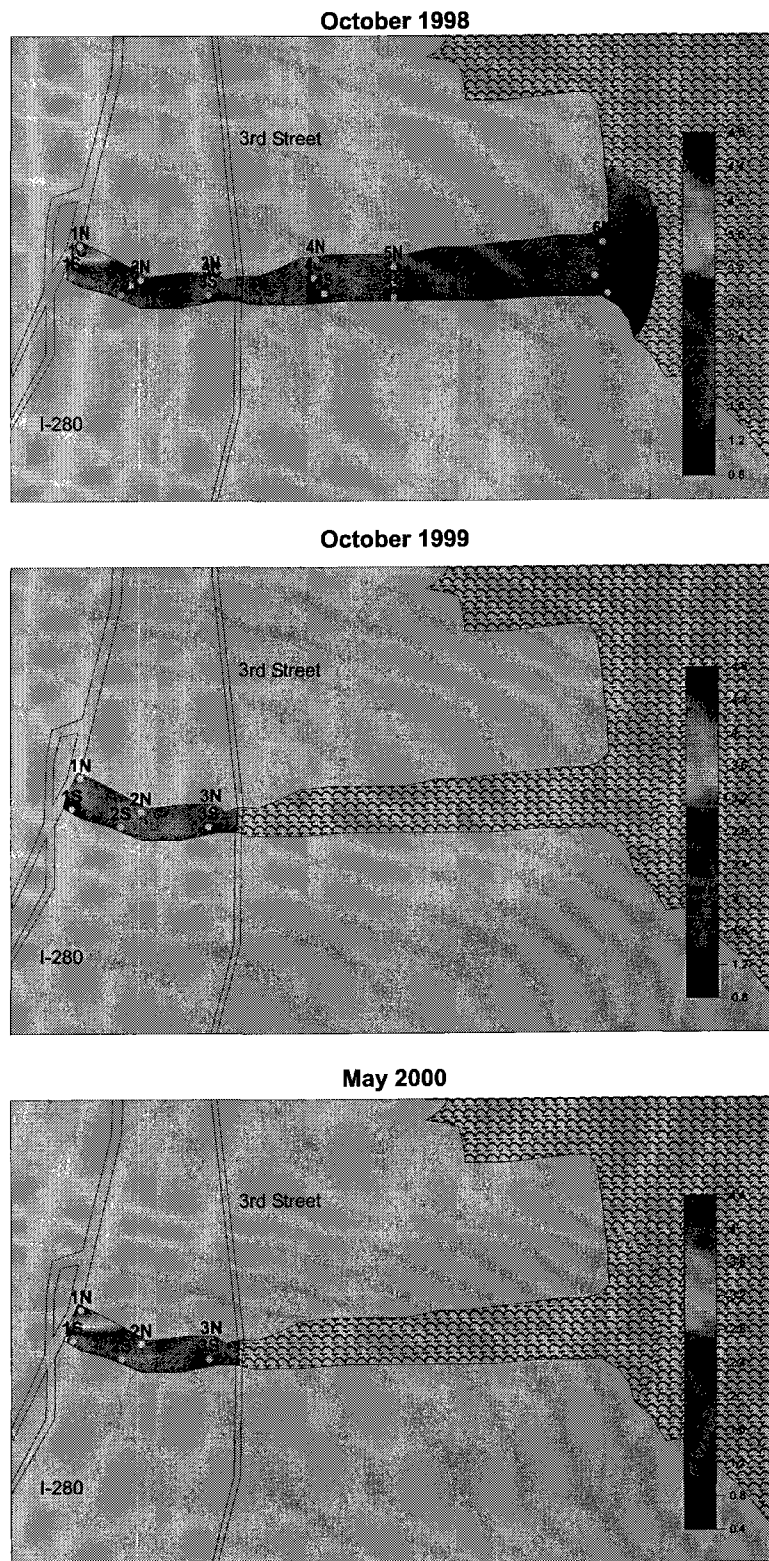


Figure 4-2. Total Organic Carbon (%) in Islais Creek.

#### 4.4 MISSION CREEK

The distribution of fine-grained surface sediments was fairly consistent between sampling events in Mission Creek (Figure 4-3). In the two wet-weather sampling events (1998 & 2000), sands (particles >63 $\mu$ m diameter) dominated the western creek end, grading into fines (particles <63 $\mu$ m) in between 4<sup>th</sup> and 6<sup>th</sup> Streets. A similar trend was observed in the 1999 dry season, except that the gradient was comprised of finer-grained particles. Fine fractions at the creek end (Transect 1) ranged from 24-63.6% of sediment dry weight in surface sediments. Extending toward the creek mouth (Transects 2-6), fine fractions exceeded 78% in all surface samples, with most exceeding 90%.

The presence of coarser sediments from the creek-end gradient may be due to the episodic erosion of creek bed sediments near the Division Street overflow at the west end. This CSO accounts for approximately 95% of total overflow volume into the creek (Hunt et al. 1998a). Erosion may also be increased by the narrowing of the creek channel and turbulence created by the disruption of flows in the vicinity of the west end.

Total organic carbon content varied more than grain size between sampling events, with concentrations greater than 4% measured between 6<sup>th</sup> and 5<sup>th</sup> Streets (Transects 2-3) in 1998 (Figure 4-4). Sediments collected in 1998 were targeted for the dry season, but were collected during a storm event that produced a significant combined sewer overflow. This area remained elevated in the two following years sampled, with average concentrations of 2.1%, compared to remaining creek areas with about 1% TOC.

Subsurface sediments from the 0-1 ft composite approached 4% at the west end, significantly exceeding levels recorded elsewhere along the creek (Appendix B2). This may represent a past accumulation from the major CSO prior to improved design and reduction in overflows. However, the deeper (1-2 ft) sediments at Station 1N were the lowest amongst the creek stations. Interpretation of subsurface data in relation to surface data is confounded by the fact that samples were taken approximately six weeks apart. Sediments extending out to Transect 5 showed reduced TOC in subsurface sediments; however, concentrations exceeded corresponding surface TOC concentrations.

**Table 4-3. Mission Creek - percent fines and TOC in surface sediments.**

Year Sampled	Area Sampled	No. of Stations	Mean % Fines	Range % Fines	Mean % TOC	Range % TOC
1998	Mission Creek	13	72.7	24.0 - 98.2	2.7	1.3 - 4.5
1998	Reference	1	90.3	90.3 - 90.3	1.2	1.2 - 1.2
1999	Mission Creek	8	79.8	46.4 - 99.0	1.8	0.8 - 3.2
1999	Reference	6	80.9	30.7 - 99.7	0.9	0.4 - 1.8
2000	Mission Creek	8	63.6	28.8 - 96.8	1.8	1.4 - 2.6
2000	Reference	5	74.8	26.3 - 97.9	0.9	0.5 - 1.2
All	Mission Creek	29	72.1	24.0 - 99.0	2.2	0.8 - 4.5
All	Reference	12	79.2	26.3 - 99.7	0.9	0.4 - 1.8

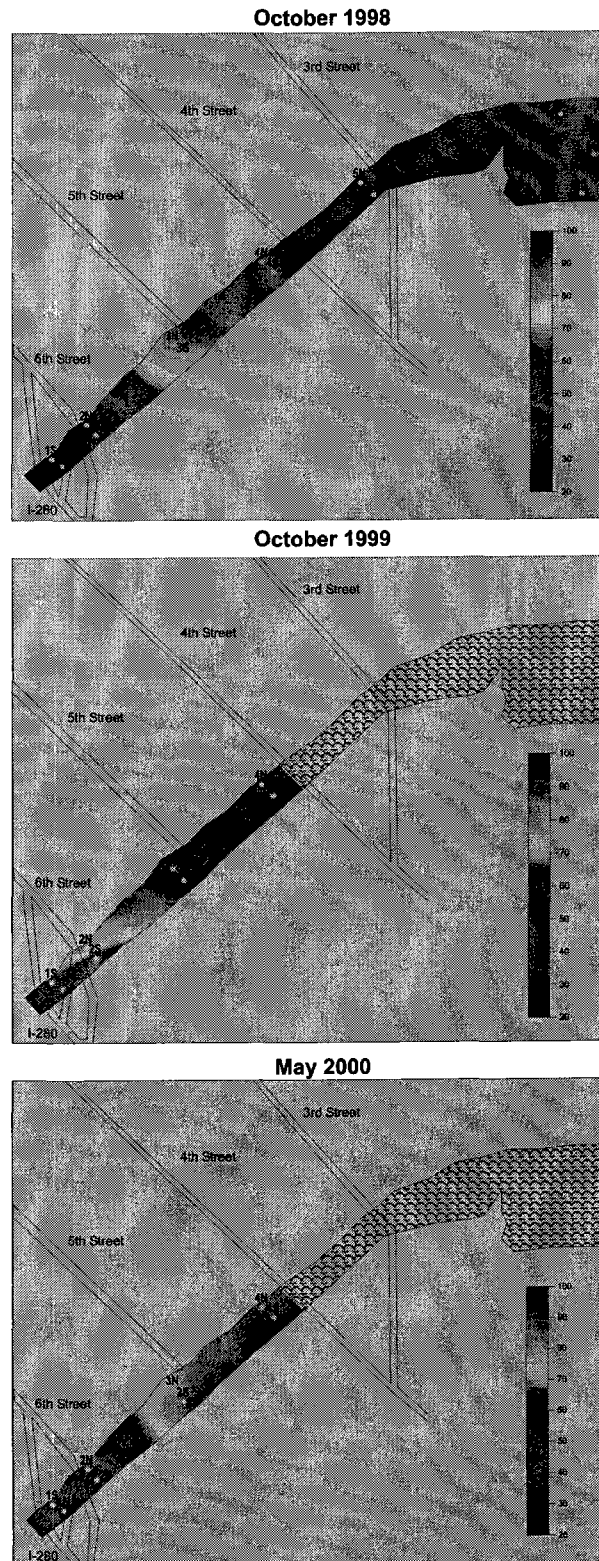
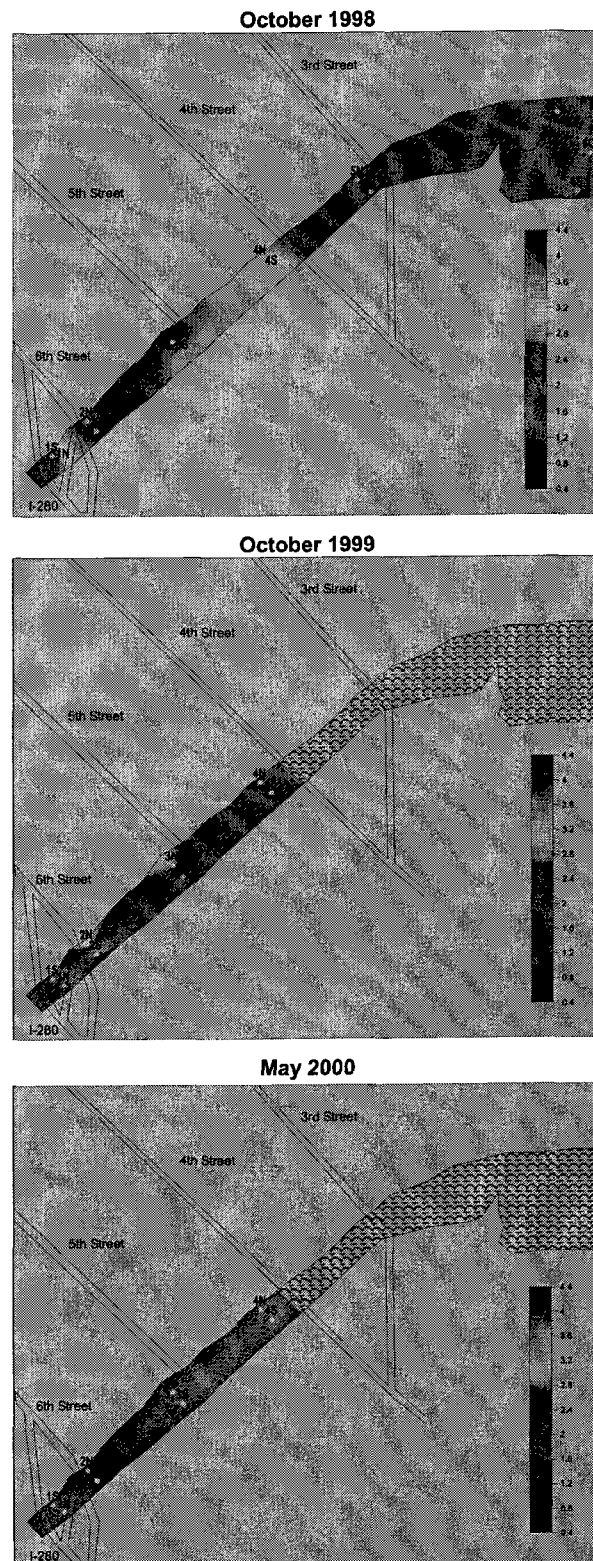


Figure 4-3. Percent fines (silt + clay) in Mission Creek.





**Figure 4-4. Total Organic Carbon (%) in Mission Creek.**

## 5.0 SEDIMENT TOXICITY

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Toxicity results for surface sediments are presented in this section. *Eohaustorius estuarius*, an estuarine amphipod of the family Haustoriidae, common in the evaluation of marine sediments, was used in a 10-day acute test. Percent survival, based on the average of five laboratory replicates, is summarized for reference and creek stations in Tables 5-1 through 5-4. Test results are compared to a reference envelope tolerance limit based on BPTCP and survey-specific data. Complete results are presented in Appendices A1 and B1 for Islais and Mission Creeks, respectively.

A total of 61 sediment stations were sampled and tested for acute toxicity to examine conditions over time and determine potential differences between wet (October 1998, April 2000) and dry (October 1999) seasons. These studies were responsive to RWQCB's requirement for site investigations, and were designed to address and/or reduce confounding factors apparent in previous BPTCP testing programs. Toxicity tests consistent with ASTM and EPA/USACE protocols were used to ameliorate four potential confounding factors: 1) high ambient levels of ammonia; 2) high ambient levels of hydrogen sulfide; 3) low levels of dissolved oxygen; and 4) experimentally induced organism sensitivity. Elevated levels of ammonia and hydrogen sulfide were reduced through careful replacement of overlying water, following a 24-hour equilibration period for each test chamber (see Section 3.3.2.3). This procedure has the potential for removing, not only unwanted confounding factors, but also soluble chemical contaminants. Considering the fact that contaminant chemicals sequestered in tested sediments have been subjected to continuous natural water exchange, it is believed that the benefits derived from reducing confounding factors far outweigh the potential minimal reduction of these chemicals. The renewal process, combined with the increased water aeration effectively eliminated low dissolved oxygen levels experienced in previous BPTCP testing of creek sediments. Experimentally induced test organism sensitivity was addressed through close interaction with the *Eohaustorius* supplier (Northwest Aquatic Sciences [NWAS], also used in the BPTCP). The salinity acclimation process for *Eohaustorius* was begun by NWAS prior to shipping the amphipods, and continued at SFPUC's Oceanside Biology Laboratory.

October 1998 samples were tested at two laboratories (SFPUC and Pacific EcoRisk) using identical protocols as a performance measure. Inter-laboratory comparisons support the findings of Jirik et al. (2000) that "Testing by multiple laboratories does not appear to reduce the precision of the results." Consequently, October 1999 and April 2000 samples were analyzed by SFPUC only. October 1998 toxicity test results evaluated herein are the laboratory-averaged results for each station, as there were no statistical differences between mean sample values for laboratory replicates ( $p > 0.05$  for all comparisons). October 1998 sampling was initially intended to establish "dry period" sediment conditions. Heavy rainfall, which produced combined sewer overflows in each of the two creeks just prior to and during sampling produced wet weather conditions. October 1999 samples were collected after a prolonged dry season; April 2000 samples were collected in the wet season, with many samples collected during rainfall.



## 5.1 OVERVIEW

With the exception of the October 1998 survey, in which creek toxicities are compared only to the BPTCP reference tolerance limit (i.e., 69.5% of control survival), sediment toxicity is evaluated using a survey-specific reference tolerance envelope. Tolerance limits were calculated for the October 1999 and April 2000 surveys using toxicity values measured at six and five San Francisco Bay reference stations, respectively. All reference sites, except for Tomales Bay (measured in 1999 only), are established Regional Monitoring Program reference sites. The resulting survey-specific (SFPUC) tolerance limits were calculated using the same method used to calculate the BPTCP tolerance limit of 69.5% (Hunt et al. 1998). The resulting SFPUC tolerance limits are 65.3 and 56.6% for October 1999 and April 2000 data, respectively. When results are compared to the historic BPTCP reference envelope, the tolerance limit (69.5%) is adjusted to account for control survival (i.e., 69.5% x fractional control survival) as recommended by Hunt et al. (1998a).

## 5.2 REFERENCE AREA

Toxicity results for reference area stations are shown in Table 5-1. Only one station (Paradise Cove) was sampled in October 1998 with a result of 65.0% survival, which was just below the BPTCP tolerance limit of 68.5% (i.e., 69.5% of 1998 control survival). Paradise Cove, an established RMP reference site, has shown intermittent toxicity in other studies (K.Taberski, RWQCB personal communication 5/99).

Survival results for the six stations sampled in October 1999 ranged from 59.0 to 99.0%, averaging 81.3%. One reference station, Island #1, fell below the BPTCP tolerance limit, indicating toxicity by this standard. All other reference site survivals exceeded the BPTCP criterion of 68.8% (i.e., 69% of 1999 control survival). All 1999 stations, except Tomales Bay, were resampled in April 2000. In general, lower survivals were observed in 2000 under wet weather conditions, compared with October 1999, which was dry. Only Island #1 had a substantially higher survival value in 2000. North Site survival values of 83% in 1999 and 89% in 2000 were commensurate.

**Table 5-1. Reference area toxicity results for the 10-day amphipod test with *Eohaustorius estuaries* and BPTCP (1998) and SFPUC (1999 & 2000) reference tolerance limits.**

Station	Percent Survival		
	October 1998 <sup>1</sup>	October 1999	April 2000
Island #1		59.0	68.0
Marconi Cove (Tomales Bay)		83.0	
North Site		83.0	89.0
Paradise Cove	65.0	94.0	65.0
South Site		99.0	80.0
Tubbs Island		70.0	59.0
Home Sediment Control	98.5	99.0	95.5
<b>Tolerance Limit</b>	<b>68.5<sup>†</sup></b>	<b>65.3<sup>‡</sup></b>	<b>56.6<sup>‡</sup></b>

<sup>1</sup>=1998 results are for average data from SFPUC and Pacific EcoRisk laboratories; <sup>†</sup>=BPTCP tolerance limit; <sup>‡</sup>= SFPUC tolerance limit

### 5.3 ISLAIS CREEK

Mean survival for all creek stations sampled in October 1998 was 70.9%. These results were a significant improvement over 1994 and 1997 BPTCP test results, which showed significant toxicity at the upper end of Islais Creek. Results ranged from 58.5 to 83.0%, with the highest survival measured at the end of the creek (Station 1N). Four of the six stations (Figure 5-1, Table 5-2) had survivals above the BPTCP tolerance limit of 68.5% (69.5% of control), indicating no significant toxicity by this standard. Stations 2N and 3S, located near the CSO Weir and the Quint Street outfall, exhibited results marginally below the standard, with average survivals of 58.5 and 61.5%, respectively. Four Islais Creek stations had greater survival values than the Paradise Cove reference area (i.e., > 65%).

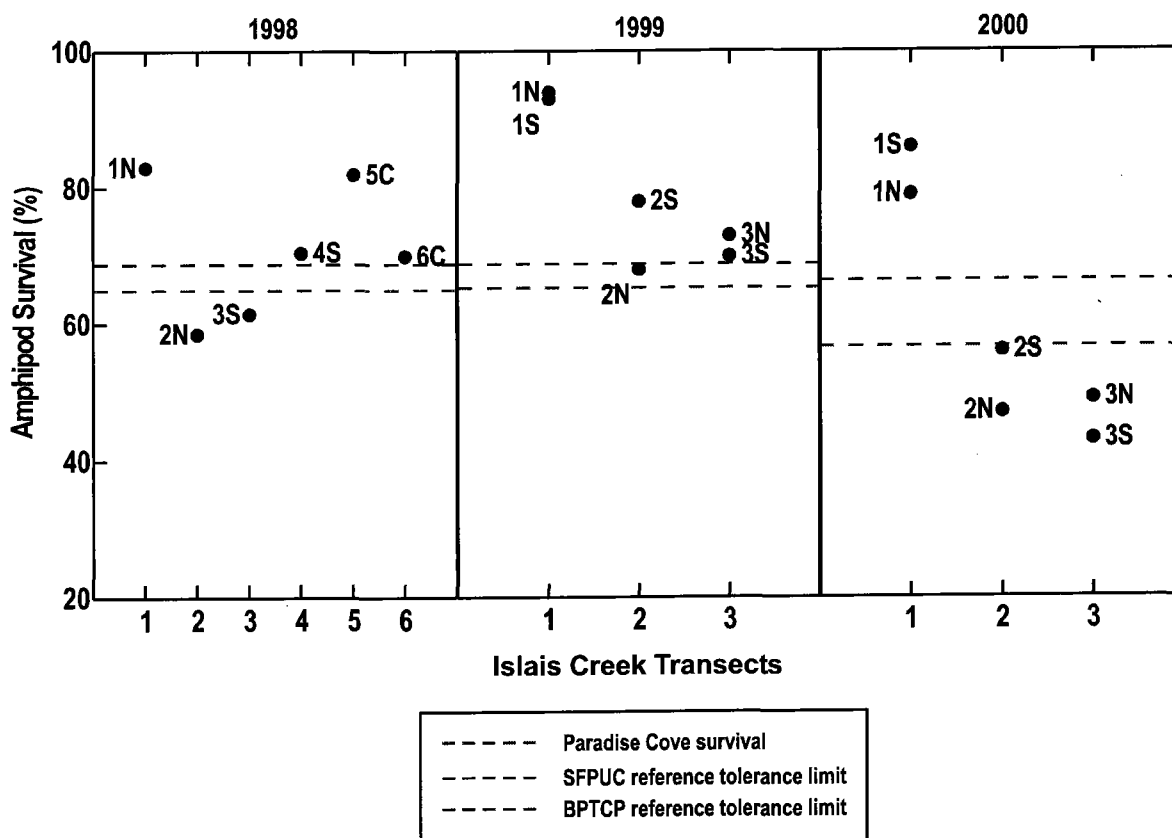


Figure 5-1. Islais Creek toxicity results and corresponding BPTCP and SFPUC survey-specific tolerance limits. Stations with recurrent toxicity ( $\geq 2$  toxic events) are shown in red.

October 1999 survival was measured at the north and south stations of transects 1 through 3 (Figure 3-1, Section 3), focusing on the upper portion of the creek surrounding the CSO Weir and the Quint Street outfall. Stations included all October 1998 stations that had less than 68.5% survival in toxicity tests and ERM quotients greater than 0.5 (see Section 6). Average amphipod survival was 79.3%, ranging from

68.0 to 94.0%. Survival at Station 2N (68.0%) was slightly below the BPTCP tolerance limit of 68.8% (69.5% of control). However, no stations were below the study-specific tolerance limit of 65.3%. Results were comparable to the six concurrently sampled reference stations.

April 2000 sampling, conducted during wet weather, re-examined all October 1999 stations. Average survival was 60.0%, ranging from 43.0 to 86.0%. Four of the six stations sampled (2N, 2S, 3N, 3S) fell below the BPTCP reference threshold of 66.4% (69.5% of 2000 control survival) and the study-specific tolerance limit of 56.6%. Toxicity at these four stations, located near the CSO Weir and Quint Street outfall, suggest that toxicity was confined to a localized area at the time of the survey.

Table 5-2 compares amphipod survival at each of the 18 stations sampled during the three programmatic surveys to tolerance limits for the BPTCP (69.5% of control survival) and the 1999 and 2000 SFPUC surveys. Amphipod survival, observed at Stations 2S and 3N, indicated recurrent sediment toxicity located mid-creek at the west and east end of the CSO Weir for both tolerance limits. The extent of toxicity appears to be confined to this immediate area. General toxicity also appears to be influenced by seasonal rainfall. Bay-wide conditions appear stressed during the April 2000 sampling period (wet weather) as reflected in lower overall survival rates measured at both creek and reference stations.

**Table 5-2. Islais Creek amphipod survival for each station and year. Stations with recurrent toxicity are shown in bold.**

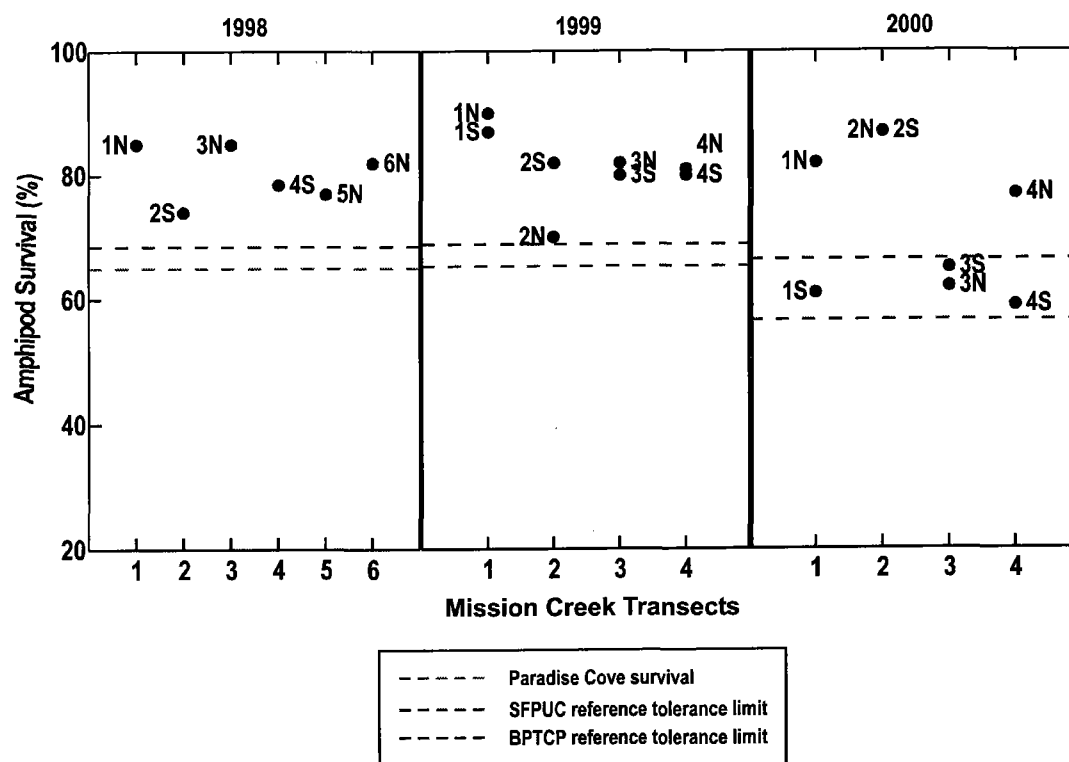
Station	Survey Year*	Percent Survival	SFPUC Tolerance Limit	BPTCP Tolerance Limit (69.5% of control)
1N	1998	83.0	NA	68.5
1N	1999	94.0	65.3	68.8
1N	2000	79.0	56.6	66.4
1S	1999	93.0	65.3	68.8
1S	2000	86.0	56.6	66.4
2N	1998	58.5	NA	68.5
2N	1999	68.0	65.3	68.8
2N	2000	47.0	56.6	66.4
2S	1999	78.0	65.3	68.8
2S	2000	56.0	56.6	66.4
3N	1999	73.0	65.3	68.8
3N	2000	49.0	56.6	66.4
3S	1998	61.5	NA	68.5
3S	1999	70.0	65.3	68.8
3S	2000	43.0	56.6	66.4
4S	1998	70.5	NA	68.5
5C	1998	82.0	NA	68.5
6C	1998	70.0	NA	68.5

Red=corresponding creek station below tolerance limit; Bold=recurrent toxicity measured in 2 years based on exceedance of SFPUC and BPTCP tolerance limit

## 5.4 MISSION CREEK

October 1998 sampling produced an average amphipod survival of 80.3% from all creek stations, ranging from 74 to 85%. This represents a significant improvement over BPTCP amphipod survival results in 1995 and 1997, which indicated significant toxicity (5% and 19% survival) at the end of Mission Creek. Figure 5-2 and Table 5-3 compare amphipod survival at each of the 22 stations sampled during the three programmatic surveys to the BPTCP tolerance limit (69.5% of control survival) and the SFPUC survey-specific tolerance limit. All six stations examined produced survivals above the BPTCP reference tolerance limit of 68.5% (69.5% of control), indicating no significant toxicity by this standard. All six sites had survivals greater than the 65% measured at the Paradise Cove reference area.

October 1999 amphipod survival was measured at eight stations in the west end of the creek between the main CSO discharge to the 4<sup>th</sup> Street intersection (Figure 3-2, Section 3) during a dry sampling period. Sampling and testing focused on stations that had ERM quotients greater than 0.5, since no significant toxicity was reported in October 1998. Average amphipod survival was 81.5%, ranging from 70.0 to 90.0%. All eight stations (Figure 5-2) examined had survivals above BPTCP and SFPUC survey-specific tolerance limits (68.8 & 65.3%, respectively), indicating no significant toxicity by these standards. The average amphipod survival for all eight stations was slightly higher than the average of the six reference sites sampled at the same time: 81.5 and 81.3%, respectively.



**Figure 5-2. Mission Creek toxicity results and corresponding BPTCP and SFPUC survey-specific tolerance limits.**

Table 5-3. Mission Creek amphipod survival for each station and year.

Station	Survey Year*	Percent Survival	SFPUC Tolerance Limit	BPTCP Tolerance Limit (69.5% of control)
1N	1998	85.0	NA	68.5
1N	1999	90.0	65.3	68.8
1N	2000	82.0	56.6	66.4
1S	1999	87.0	65.3	68.8
1S	2000	61.0	56.6	66.4
2N	1999	70.0	65.3	68.8
2N	2000	87.0	56.6	66.4
2S	1998	74.0	NA	68.5
2S	1999	82.0	65.3	68.8
2S	2000	87.0	56.6	66.4
3N	1998	85.0	NA	68.5
3N	1999	82.0	65.3	68.8
3N	2000	62.0	56.6	66.4
3S	1999	80.0	65.3	68.8
3S	2000	65.0	56.6	66.4
4N	1999	81.0	65.3	68.8
4N	2000	77.0	56.6	66.4
4S	1998	78.5	NA	68.5
4S	1999	80.0	65.3	68.8
4S	2000	59.0	56.6	66.4
5N	1998	77.0	NA	68.5
6N	1998	78.0	NA	68.5

Red=corresponding creek station below tolerance limit; Note: no recurrent toxicity measured in 2 years based on exceedance of SFPUC and BPTCP tolerance limit

April 2000 amphipod survival was generally lower than that measured in 1998 and 1999. Average survival was 72%, ranging from 59.0 to 87.0%. Four of the eight stations sampled (1S, 3N, 3S, 4S) fell below the BPTCP tolerance limit of 66.4% (69.5% of control). However, all stations were above the SFPUC survey-specific tolerance limit (56.6%), indicating no significant toxicity compared with reference sites sampled during the same survey. April 2000 sampling took place during wet weather, which may have contributed to bay-wide stressed conditions reflected as an overall depression in amphipod survival at both creek and reference stations.

## 5.5 CONCLUSIONS

Recurrent toxicity was indicated at two stations in Islais Creek (2N, 3S) located west of the 3<sup>rd</sup> Street Bridge. Mission Creek, in stark contrast to BPTCP investigations, produced no recurrent toxicity. In general, reduced amphipod survival observed in the April 2000 survey for both creeks and the five corresponding reference sites, may have been due to bay-wide stressed conditions, as control survival at 95.5% was lower in 2000 than the previous two surveys (i.e., 98.5 and 99.0%). These results indicate that test animals may have been slightly stressed independent of sediment conditions in the creeks.

## 6.0 SEDIMENT CHEMISTRY

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Sediment chemical results, consisting of organic compounds and metals for creek and reference areas, are presented in this section. Results describe distribution patterns within each creek and identify chemicals of potential concern (COPCs) through comparison with reference sediments and corresponding effects-range median (ERM) guideline values. Surface sediment distributions are examined in relation to distance from active and historic combined sewer overflows (CSOs) and across surveys, which included both wet and dry weather events. The distribution of subsurface sediment chemicals is discussed qualitatively to estimate the vertical extent of contamination and the resuspension potential of buried sediment. Results are organized by creek into chemical suites, consisting of metals, polycyclic aromatic hydrocarbons (PAH), organochlorine pesticides, and polychlorinated biphenyl (PCB) congeners. Surface sediment results for each station are presented in Appendices A1 and B1 for Islais and Mission Creeks, respectively. Corresponding subsurface results are presented in Appendices A2 and B2. Graphical displays of surface and subsurface results are shown in sub-appendices 4 and 5, respectively (e.g., A4 & A5 for Islais Creek).

Following the primary objective of the October 1998 survey, sediment chemical concentrations were evaluated using ERM guidelines (see Section 1.3.1.2) to compare results with previous Bay Protection and Toxic Cleanup Program (BPTCP) findings (study objective 1, Section 1). Sediment chemicals measured in October 1999 and April 2000 were compared to upper 95<sup>th</sup> percentile predictive limits (UPL), calculated using synoptically collected reference site data (see Section 3.1). This approach is similar to the reference envelope tolerance limit used to evaluate amphipod toxicity results (see Section 5). Comparisons of creek and reference area sediments were made using total organic carbon (TOC) normalized data (e.g., ng or  $\mu$ g [chemical] per gram TOC in dry weight). All other presentations and discussions of chemical data are based on sediment dry-weight.

### 6.1 OVERVIEW

Chemical concentrations in the two creeks varied considerably as a function of location, sediment type and TOC concentration. Most chemical concentrations were positively correlated with sediment TOC; however, with few exceptions similar relationships were not observed with grain size. In addition, many metals were positively correlated with either aluminum or iron, which are major constituents of sediment minerals. Sediment concentrations of chlorinated pesticides, polychlorinated biphenyls (PCB) and polycyclic aromatic hydrocarbons (PAH), which are nonionic organic compounds, increased with increasing TOC, as expected, due to their relative insolubility in water and high affinity for particulate matter. In addition, most metals, especially those significantly elevated above background concentrations, were positively correlated with TOC. TOC-normalized chemical concentrations exceeding either the corresponding TOC-normalized reference area upper-95<sup>th</sup> predictive limit or one-half of the ERM value in two or more surveys were identified as COPCs and evaluated further in Sections 7 and 9.

Overall PAH contamination was evaluated using separate sums of seven low-molecular-weight (LMW) PAH compounds and six high-molecular-weight (HMW) PAH compounds based on Long et al. (1995). An additional 28 PAH compounds were analyzed to provide information on hydrocarbon type and

source. Potential contaminant sources of PAH and other COPCs to creek sediments are discussed in Section 8.

Organochlorine pesticides measured in sediments included five categories of compounds: 1) six isomers of DDT (including DDD and DDE degradation products), 2) four Chlordane isomers and congeners: alpha-Chlordane, gamma-Chlordane, cis-Nonachlor and trans-Nonachlor, 3) Aldrin and its metabolites: Dieldrin and Endrin, 4) Lindane, and 5) Mirex. Each of these compounds has potentially toxic effects upon marine organisms if sufficiently concentrated and bioavailable. Some compounds, such as DDT, are notable for their biomagnification in fatty tissues at successively higher levels of the food chain. These pesticides also are nonionic organic compounds that have a high affinity for organic carbon (EPA 1993), preferentially concentrating in TOC-enriched sediments.

Polychlorinated biphenyls are also nonionic organic compounds, which generally increase with increasing TOC due to their relative insolubility in water and high affinity for organic matter (Section 2.3.3). The sum of 18 NOAA Status and Trends PCB congeners were used to assess the extent of contamination in creek sediments. PCB data used by Long et al. (1995) to derive a total PCB ERM value of 180 parts-per-billion ( $\text{ng g}^{-1}$ ) include Aroclor data as well as other forms of congeners. ERM values for total PCB and chlorinated pesticides are based on sediments with an average of 1.2% TOC, and are therefore not directly comparable to corresponding concentrations found in TOC-enriched creek sediments. Additionally, these nonionic organic compounds tend to bioaccumulate in the food chain, but are not necessarily toxic in sediment bioassays, even at elevated concentrations (i.e.,  $>1 \text{ } \mu\text{g g}^{-1}$ ). Because of this, potential adverse effects from PCBs and pesticides were evaluated for bioaccumulation potential, using a standard 28-day test with clams (see Section 7).

## 6.2 REFERENCE AREA

San Francisco Bay reference area sediments were sampled at one location in October 1998, six locations in October 1999 and five locations in April 2000. Locations and sample inventory for each station are shown in Section 3, Figure 3-4 and Table 3-7, respectively. These primarily fine-grained, low-TOC sediments were relatively free of contamination across all three surveys. In particular, metal concentrations were commensurate with pristine sediments located along the California coast and elsewhere, with the exception of copper, mercury, nickel and silver, which were elevated 2-8 times at in-bay reference sites (Table 6-1). Mean reference area concentrations were in excellent agreement with Regional Monitoring Program (RMP) data collected from 1993 to 1997 from a total of 32 San Francisco Bay offshore and delta stations, including five locations sampled in this study.

Reference area concentrations of total PCB, total DDT and total Chlordane were similar to background concentrations measured in nearshore sediments of relatively unimpacted areas (Table 6-2). Trace concentrations of these ubiquitous anthropogenic contaminants to otherwise pristine sediments are due largely to atmospheric fallout and hydrodynamic transport of discharged waste. These and other chlorinated hydrocarbons are found at trace concentrations in areas far-removed from human populations, including the Antarctic (Kennicut et al. 1992).

Reference sediment concentrations of total PAH were moderately elevated compared to other pristine areas (Table 6-2); however, at less than 1 part-per-million (i.e.,  $<1000 \text{ ng g}^{-1}$ ), concentrations were well

below recognized threshold levels (e.g., ERL, ERM). Background concentrations measured at in-bay reference stations are most likely from atmospheric fallout of fossil fuel combustion related products (see Section 8).

Upper 95<sup>th</sup> predictive limits for TOC-normalized and sediment dry weight (non-normalized) data are shown in Table 6-3 for the 1999 and 2000 surveys. Upper predictive limits were not calculated for 1998 data, as only one reference station was sampled. Analyte results for individual creek stations are compared to corresponding UPLs shown in Table 6-3; results are presented in Sections 6-3 and 6-4 for Islais Creek and Mission Creek, respectively. TOC-normalized analyte concentrations that exceeded corresponding UPLs or non-normalized analyte concentrations that exceeded corresponding ERMs in two or more surveys were retained as COPCs for each creek.

**Table 6-1. Mean metal concentrations for reference locations (all surveys combined) and other areas.**

Metal (?g/g <sup>1</sup> )	Reference Area	San Francisco Bay <sup>1</sup>	Clean California Coast <sup>2</sup>	Southwest English Estuary <sup>2</sup>	Continental Crust <sup>3</sup>
Arsenic	6.98	5.65 - 10.6	12	6.4	2
Cadmium	0.32	0.23 - 0.89	0.33	0.23	0.2
Chromium	96.4	64 - 123.4	22	30	126
Copper	38.1	18.9 - 53.7	18.3	7	45
Lead	18.0	7.7 - 42	10	25	15
Mercury	0.23	0.09 - 0.53	0.04	0.03	0.06
Nickel	86.7	59.9 - 109.2	-	17	56
Selenium	0.25	0.06 - 0.65	-	0.11	0.12
Silver	0.52	0.01 - 0.56	-	0.07	0.07
Zinc	103.2	61.7 - 181.3	43	59	40

<sup>1</sup>ranges in mean concentrations from SFEI RMP Report 1999 ([http://www.sfei.org/rmp/1999/RMP99\\_Results.pdf](http://www.sfei.org/rmp/1999/RMP99_Results.pdf)); <sup>2</sup>from Kennish (1997); <sup>3</sup>from Wedepohl (1995)

**Table 6-2. Mean organic chemical concentrations for reference locations (all surveys combined) and other "clean" areas.**

Organic Chemical(s) (ng/g <sup>1</sup> )	Reference Area	San Francisco Bay <sup>1</sup>	Clean California Coast <sup>2</sup>	North Atlantic <sup>2</sup>	Baltic Sea <sup>2</sup>
Total PAH	732	83.8 - 2695.5	160	120	258
Total DDT	5.67	0.39 - 17.84	5-30	0.4	2
Total Chlordane	0.9	0.18 - 7.77	-	-	-
Dieldrin	0.7	ND - 1.07	-	-	-
Total PCB	10.16	0.45 - 41.77	1-13	15	8.4-10.8

<sup>1</sup>ranges in mean concentrations from SFEI RMP Report 1999 ([http://www.sfei.org/rmp/1999/RMP99\\_Results.pdf](http://www.sfei.org/rmp/1999/RMP99_Results.pdf)); <sup>2</sup>adopted from Table 2.1, Appendix 2 in Kennish (1997)



Table 6-3. 95<sup>th</sup> UPLs for reference stations sampled in 1999 and 2000 (reported for TOC-normalized and non-normalized data).

Analyte	October 1999 Survey		April 2000 Survey	
	TOC-normalized UPL	Non-normalized UPL	TOC-normalized UPL	Non-normalized UPL
<b>Metals (<math>\mu\text{g g}^{-1}</math>)</b>				
Arsenic	1179	13.7	1192	14.6
Cadmium	41.1	0.41	61.3	0.6
Chromium	11,116	156	10,275	119
Copper	6928	71	6329	73
Lead	2912	27	2213	26
Mercury	32.4	0.32	40.5	0.44
Nickel	10,088	166	8641	98
Selenium	29.3	0.39	46.8	0.49
Silver	87.3	0.85	28.5	0.25
Zinc	14,891	144	12,903	152
<b>Organics (<math>\text{ng g}^{-1}</math>)</b>				
Total LMW PAH	18,216	179	22,595	272
Total HMW PAH	112,581	1117	114,760	1056
Dieldrin	250.6	2.53	20.6	0.25
Endrin	34.7	0.44	20.6	0.25
Total Chlordane	70.5	1.4	20.6	1.8
Total DDT	944	9.4	1283	13.1
Total PCB	1952	20.2	1336	18.6

TOC-normalized=mass of chemical  $\text{g}^{-1}$  TOC; samples with < 1% TOC were normalized using 1% TOC

### 6.3 ISLAIS CREEK

Sediments were sampled at 18 locations in 1998, spanning from the creek end (Transect 1) to the mouth (Transect 6) (see Figure 3-1, Section 3). In general, surface sediment chemical concentrations measured east of the 3<sup>rd</sup> Street Bridge (Transects 4-6) were commensurate with sediment concentrations measured at Paradise Cove and at the relatively unimpacted in-bay and Delta sites measured in the Regional Monitoring Program (SFEI 1997). These stations, along with the center stations from Transects 1-3, were omitted from subsequent sampling in 1999 and 2000.

Chemical distribution patterns in surface sediments were fairly consistent over time, with the highest concentrations measured directly below the Interstate 280 overpass at the creek end near the main CSO Weir. In addition, many chemical concentrations decreased with distance from the creek end, indicating the CSO Weir or runoff from Interstate 280 as likely sources without additional information. Surface sediment distributions of key chemicals are shown for each survey in Appendix A4. In general, chemical concentrations increased significantly with depth, indicating that buried sediments are most likely in-place and that inputs have diminished over time. Subsurface distribution plots for key chemicals sampled in October 1998 are shown in Appendix A5 for 1-ft core intervals sampled from 0-4 ft depths in October 1998. All core intervals were analyzed from Transects 1-3; only 0-2 ft depths were analyzed from Transects 4-6, since corresponding surface sediments from these stations were relatively clean.

Many organic as well as inorganic contaminants were significantly correlated with TOC; however, similar relationships were not observed for grain size characteristics, including percent fines, shown in Table 6-4.

**Table 6-4. Islais Creek - correlation results for selected chemical concentrations with TOC and grain size.**

	Copper	Mercury	Lead	Zinc	Total DDT	Dieldrin	HMW PAH	LMW PAH	Total PAH	Total PCB	Total Chlordane
<b>Percent Fines</b>											
r	-0.43	-0.50	-0.87	-0.60	-0.41	-0.37	0.22	0.23	0.22	-0.58	-0.61
p	0.02	0.005	<0.001	<0.001	0.02	0.04	0.25	0.22	0.24	<0.001	<0.001
<b>TOC</b>											
r	<b>0.81</b>	<b>0.83</b>	0.35	<b>0.75</b>	<b>0.85</b>	<b>0.75</b>	<b>0.52</b>	<b>0.49</b>	<b>0.52</b>	<b>0.66</b>	<b>0.80</b>
p	<0.001	<0.001	0.055	<0.001	<0.001	<0.001	0.003	0.006	0.003	<0.001	<0.001

r=correlation coefficient; p=probability; **bold**=significant positive correlation at  $p < 0.05$ .

### 6.3.1 Islais Creek Metals

**Surface results.** Surface sediment concentrations of mercury, lead, and zinc consistently exceeded corresponding ERM values at several stations in all three surveys. The ERM for nickel at  $51.6 \text{ } \mu\text{g/g}$  (micrograms per gram; parts-per-million [ppm]), was exceeded at nearly all stations; however, nickel is generally not considered a COPC, as naturally elevated concentrations are found in non-toxic sediments throughout San Francisco Bay (Hunt et al. 1998a). Concentrations of nickel, as well as arsenic, cadmium, chromium, copper, selenium and silver were commensurate with concentrations measured at in-bay reference stations, showing little evidence of anthropogenic enhancement. Selenium concentrations ranged from  $0.13$  to  $0.75 \text{ } \mu\text{g/g}$  with an average concentration of  $0.40 \text{ } \mu\text{g/g}$ . These results are comparable to RMP results reported from 1993 through 1997 with a mean concentration of  $0.37 \text{ } \mu\text{g/g}$  (and a standard deviation of  $\pm 0.31 \text{ } \mu\text{g/g}$ ). There is no corresponding ERM value for selenium; however, concentrations below  $0.33 \text{ } \mu\text{g/g}$  have been reported as uncontaminated background for San Francisco Bay sediments (Walters and Gartner 1985).

Concentrations of lead, mercury and zinc measured in 1998 were elevated at the creek end (Transect 1, Figure 6-1), somewhat less at Transects 2 and 3 (near the main CSO Weir and Quint Street Outfall) and not identifiable at Transects 4, 5 and 6 (east of 3<sup>rd</sup> Street bridge). These metals remained elevated in 1999 and 2000, again with the highest concentrations measured at the creek end. Station 1N sediments posted consistently high concentrations of lead, exceeding the ERM of  $218 \text{ } \mu\text{g/g}$  in all three sampling events. However, concentrations were an order of magnitude lower in Transect 2 sediments (Figure 6-1). Zinc exceeded the ERM of  $419 \text{ } \mu\text{g/g}$  only once, at Station 1N in 1998. Mercury was elevated at Station 1N in all three sampling events, with the highest concentration ( $2.49 \text{ } \mu\text{g/g}$ ) measured in 1998. Station 3N, adjacent to the east end of the CSO Weir, also showed elevated concentrations of mercury, lead and zinc, often approaching ERM values. Table 6-5 shows that maximum metal concentrations from all three surveys were measured either at Station 1N, 3N, or 3S (arsenic only).

**Subsurface results.** Concentrations increased with depth and decreased with distance from the creek end for most metals, approaching background concentrations east of the 3<sup>rd</sup> Street Bridge. Asymptotic trends were observed for copper, mercury, lead, silver and zinc (Figure 6-2 & Appendix A5), confirming that loading of most metals to Islais Creek has decreased over time. These patterns also indicate that creek contaminants are not vertically well-mixed and likely are not being resuspended. Sediment age-dating studies should be performed to confirm this premise. Only concentrations of arsenic, nickel, and selenium remained constant across depth and distance, or actually increased with distance from the creek end. Summary results for lead, mercury and zinc are presented in Table 6-6.

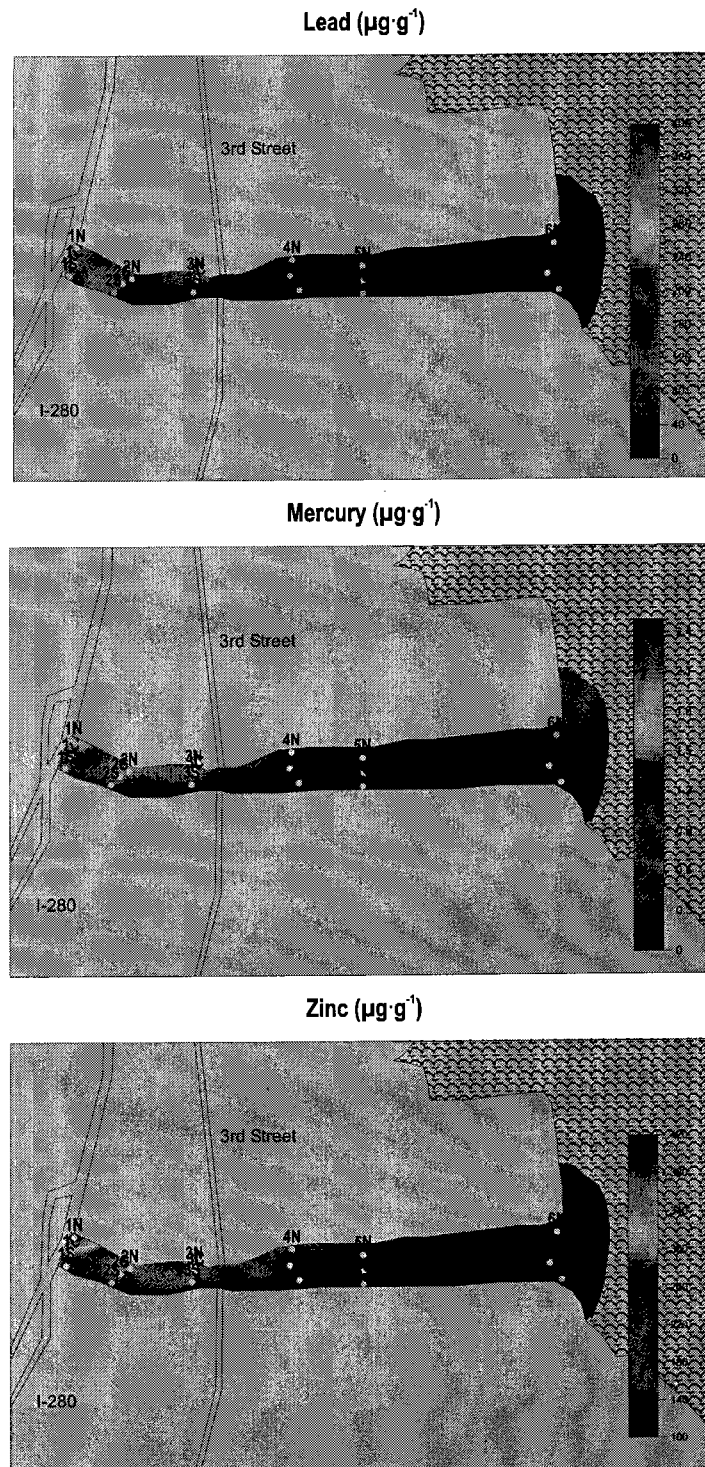
**Table 6-5. Islais Creek – Surface metal concentrations, Transects 1-3 (21 stations), all surveys ( $\mu\text{g g}^{-1}$ , ppm dry weight).**

Metal	Minimum	Maximum	Maximum Station	Mean	Standard Deviation	ERM <sup>1</sup>	Reference Mean <sup>2</sup>
Arsenic	1.94	13.4	3S	8.36	3.08	70	6.98
Cadmium	0.48	2.42	1N	1.00	0.54	9.6	0.32
Chromium	69.9	143	3N	107.5	21.2	370	96.4
Copper	55.3	139	3N	83.1	24.3	270	38.1
Lead	30.0	<b>402.8</b>	1N	124.7	125.8	218	18.0
Mercury	0.22	<b>2.49</b>	1N	0.54	0.51	0.71	0.23
Nickel	49.0	<b>144.33</b>	3N	<b>98.11</b>	23.5	51.6	<b>86.7</b>
Selenium	0.05	0.77	3N	0.43	0.19	NA	0.25
Silver	0.50	2.40	3N	0.92	0.52	3.7	0.52
Zinc	140.0	<b>419</b>	1N	227.5	88.4	410	103.2

<sup>1</sup>=source Long et al. (1995); <sup>2</sup>=mean reference site concentration for all surveys; **bold** indicates > ERM

**Table 6-6. Islais Creek – Subsurface lead, mercury and zinc concentrations ( $\mu\text{g g}^{-1}$ , ppm dry weight).**

Depth	Lead		Mercury		Zinc	
	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration
0-1 ft	65.3	1C; 232.0	0.34	1C; 0.78	192.2	1C; 402.0
1-2 ft	101.2	1C; 356.0	0.45	1C; 0.82	241.2	1C; 546.0
2-3 ft	182.1	1C; 355.7	0.79	1C; 1.30	448.1	2N; 586.7
3-4 ft	211.6	1C; 383.1	0.82	1C; 1.03	368.4	1C; 574.9



**Figure 6-1. Distribution of lead, mercury and zinc at Islais Creek – October 1998**

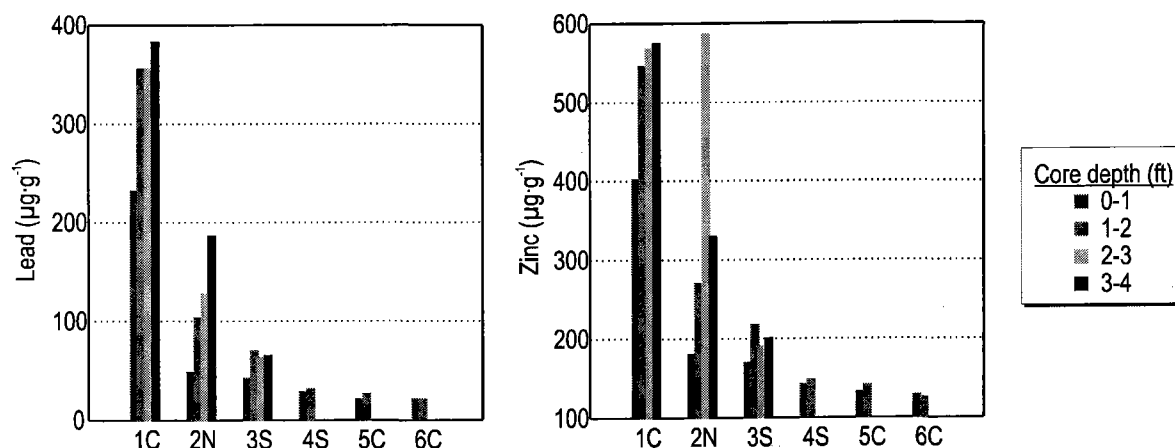


Figure 6-2. Subsurface distributions of lead and zinc at Islais Creek – October 1998.

### 6.3.2 Islais Creek Polycyclic Aromatic Hydrocarbons (PAH)

**Surface results.** Consistently higher concentrations of high molecular weight (HMW) PAH compared to low molecular weight (LMW) PAH were measured in surface sediments in all three surveys. Distribution patterns were similar to those observed for metals, with the highest concentrations measured near the creek end. Mean total PAH concentrations were relatively low, ranging from 1338 to 5324 nanograms per gram ( $\text{ng}\cdot\text{g}^{-1}$ , ppb) for Transects 4-6 (east of 3<sup>rd</sup> Street), which were sampled in 1998 only. Summary results for sediments collected west and east of the 3<sup>rd</sup> Street Bridge are presented in Table 6-7. The following discussion focuses on sediments collected west of the 3<sup>rd</sup> Street Bridge (Transects 1-3).

Table 6-7. Islais Creek - PAH in surface sediments, all surveys combined ( $\text{ng}\cdot\text{g}^{-1}$  dry weight).

Parameter	Minimum	Maximum	Maximum Station	Mean	ERM <sup>1</sup>	Reference Mean <sup>2</sup>
<b>Transects 1-3</b>						
LMW PAH	269	4371	2N	1565	3160	269
HMW PAH	1330	22,330	2N	7237	9600	956
<b>Transects 4-6 (1998 only)</b>						
LMW PAH	326	1217	5S	552	3160	269
HMW PAH	1012	4107	5S	2043	9600	956

<sup>1</sup>=source Long et al. (1995); <sup>2</sup>=all surveys combined

The ERM value for HMW PAH was exceeded twice at two stations: 3N in 1998 and 1999 and 2N in 1999 and 2000. However, these stations also had high concentrations of TOC (mean = 2.9 & 2.3%, respectively), which tends to concentrate non-ionic organic compounds. Only Station 3N exceeded the ERM value for total LMW PAH, with a concentration of 4371  $\text{ng}\cdot\text{g}^{-1}$  in April 2000. Figure 6-3 shows the distribution of HMW PAH measured throughout the creek in October 1998. Normalization of HMW PAH to TOC produces a more even distribution of the relative concentrations of HMW PAH in surface sediments. As previously discussed in Section 1.2.4.2, EPA has draft criteria for PAH compounds based

on TOC content; however, ERM criteria are based on sediments with an average TOC concentration of 1.2%. The premise for TOC normalization for nonionic organic compounds, such as PAH, is based on substantial evidence that these contaminants adsorb to organic carbon, potentially reducing their toxicity (WDOE 1993; Swartz et al. 1990 & 1994).

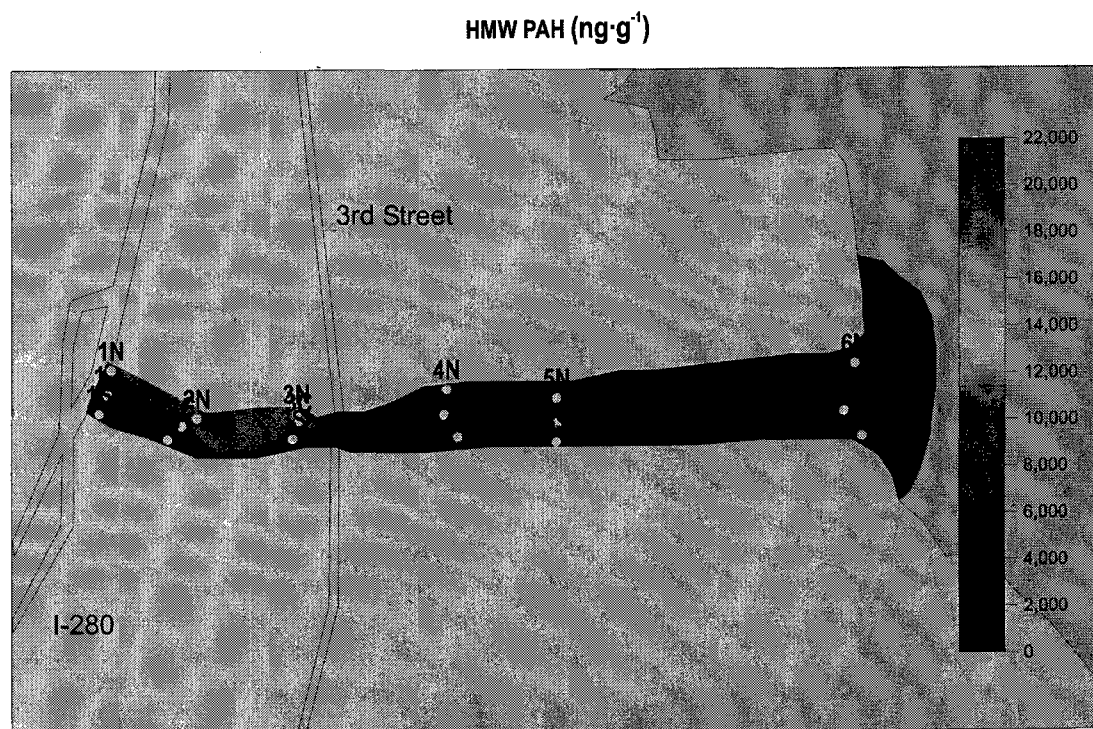


Figure 6-3. Distribution of HMW PAH at Islais Creek – October 1998.

**Subsurface results.** In general, concentrations of most individual PAH compounds, as well as total LMW and HMW PAH, were fairly consistent across depth, with the exception of extremely high concentrations (i.e., >44,000 ng·g<sup>-1</sup> total PAH) observed in the 2-3 and 3-4 ft intervals at Station 2N (Figure 6-4). In general, PAH concentrations diminished with distance from Station 2N, both up and down the creek. Most subsurface LMW PAH concentrations were below the ERM guideline (i.e., <3160 ng·g<sup>-1</sup>), except for several core intervals collected below 1 ft at Stations 1C and 2N. Mean concentrations of LMW PAH ranged from 1240 to 3778 ng·g<sup>-1</sup> across all depths. As with surface sediments, concentrations of subsurface HMW PAH were much higher, averaging 5302 to 20,638 ng·g<sup>-1</sup> across all depths (Table 6-8).

Table 6-8. Islais Creek - mean concentrations of LMW PAH, HMW PAH and total PCB in subsurface sediment ( $\text{ng g}^{-1}$ , ppb dry weight).

Depth	LMW PAH		HMW PAH		Total PCB	
	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration
0-1 ft	1240	2N; 2393	5302	2N; 14,210	79.1	1C; 342.2
1-2 ft	2022	1C; 5881	6556	2N; 12,530	94.3	1C; 345.9
2-3 ft	3778	2N; 7040	18,952	2N; 44,760	312.2	1C; 599.8
3-4 ft	2753	2N; 3784	20,638	2N; 47,120	363.5	1C; 577.9

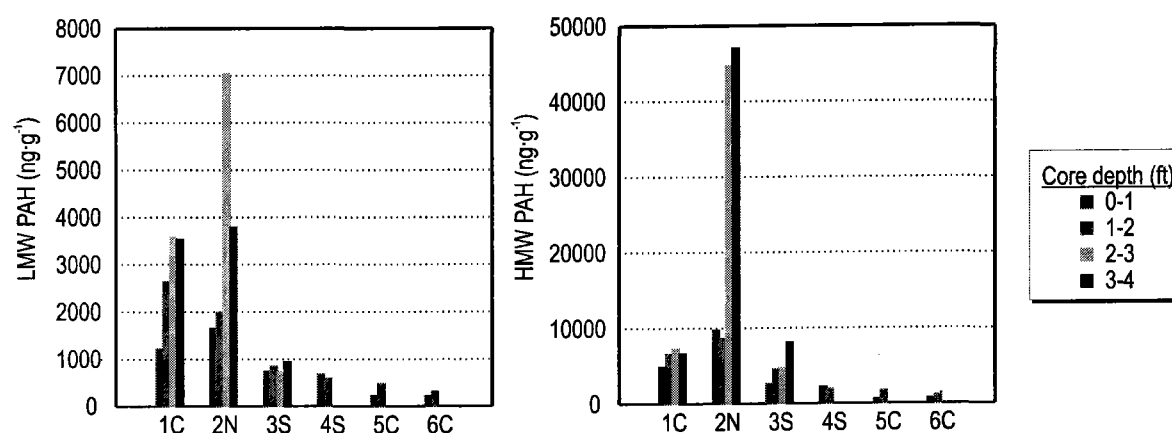


Figure 6-4. Subsurface distributions of LMW and HMW PAH at Islais Creek – October 1998.

### 6.3.3 Islais Creek Polychlorinated Biphenyls (PCBs)

**Surface sediments.** The horizontal distribution of total PCB in Islais Creek was similar to that for PAH, varying considerably as a function of location, sediment type and TOC concentration. Total PCB concentrations ranged from 13.97 to 414.1  $\text{ng g}^{-1}$  in surface sediments, with the highest concentrations measured at stations 1N and 3N (412.6 and 414.1  $\text{ng g}^{-1}$ , respectively). The concentration of 414.1  $\text{ng g}^{-1}$  reported for Station 3N in 1998, was the average of three field replicates measured at 220.85, 399.22 and 622.95  $\text{ng g}^{-1}$ . This high station variability along with the fact that low concentration samples were collected nearby at Stations 2C, 2N and 3C (all <40  $\text{ng g}^{-1}$ ) indicate an extremely heterogeneous distribution of PCB at the west end of the creek. Surface distribution of total PCB measured in 1998 is shown throughout the creek in Figure 6-5. Concentrations also varied between surveys, with a distinct downward trend for the most contaminated stations. Stations 1N and 3N had concentrations of 166 and 172  $\text{ng g}^{-1}$ , respectively in October 1999, falling to 126 and 68  $\text{ng g}^{-1}$  in April 2000. Additional sampling is required to verify this downward trend. In the 1998 survey, total PCB concentrations east of 3<sup>rd</sup> Street approximated concentrations measured at reference stations throughout the study (reference mean = 10.2  $\text{ng g}^{-1}$ ), indicating that PCBs at the creek end are not significantly transported to the creek mouth and bay. The distribution of total PCB, like total PAH, was significantly correlated with TOC (Table 6-4). Summary statistics for total PCB in surface sediments including concentration mean and range are shown

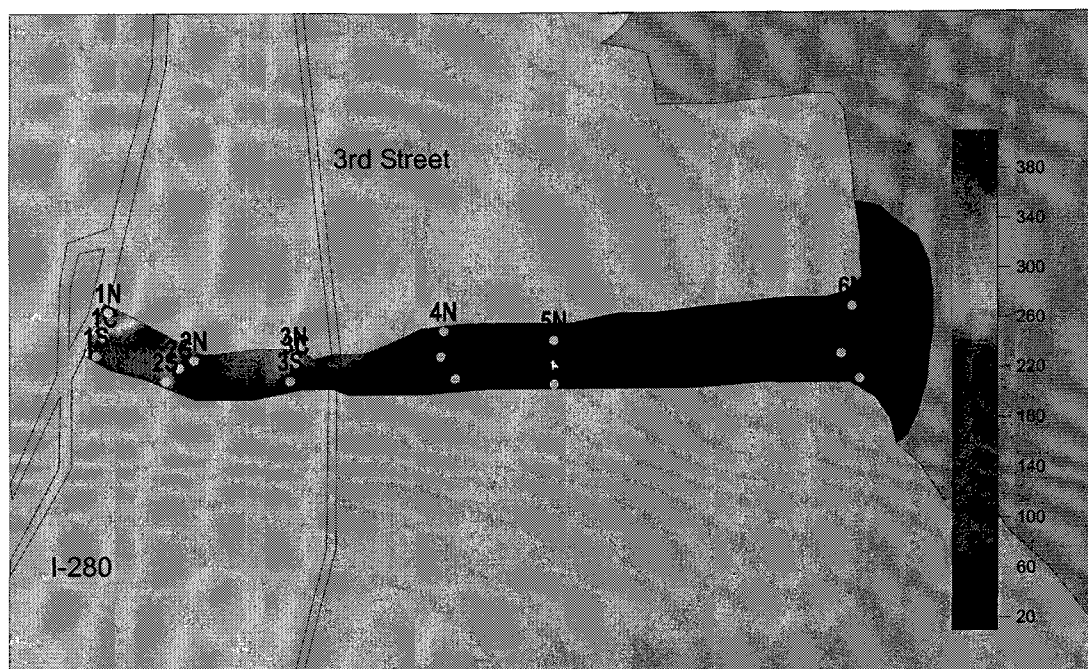
in Table 6-9. Results from the 1998 survey indicated that all PCBs were present in approximately equal concentrations of Aroclor 1254 and 1260. Total Aroclor concentrations were approximately double those measured for total PCB, which was based on 18 out of 209 possible PCB congeners. Aroclors were not quantified in the following two surveys.

**Table 6-9. Islais Creek – pesticides and total PCB in surface sediments, all studies combined ( $\text{ng}\cdot\text{g}^{-1}$ , ppb dry weight).**

Parameter	Minimum	Maximum	Maximum Station	Mean	ERM <sup>1</sup>	Reference Mean
<b><u>Transects 1-3 (21 stations)</u></b>						
Total DDT	10.8	86.6	1N	32.9	NA	5.67
Total Chlordane	2.9	79.0	1N	21.4	6 <sup>#</sup>	0.90
Dieldrin	0.5	34.3	3N	7.3	8 <sup>#</sup>	0.70
Total PCB	31.4	*414.1	3N	148.0	180 <sup>&amp;</sup>	10.16
<b><u>Transects 4-6 (1998 only - 9 stations)</u></b>						
Total DDT	7.1	12.1	4C	9.6	NA	5.67
Total Chlordane	0.3	2.1	4N	1.4	6 <sup>#</sup>	0.90
Dieldrin	1.2	1.8	5S	1.5	8 <sup>#</sup>	0.70
Total PCB	14.0	25.7	4S	18.4	180 <sup>&amp;</sup>	10.16

<sup>&</sup>=source Long et al. (1995); <sup>#</sup>=source Long & Morgan (1991); \* =average value for 3 field replicates: 220.8, 399.2, 622.9

**Total PCBs ( $\text{ng}\cdot\text{g}^{-1}$ )**



**Figure 6-5. Distribution of Total PCB at Islais Creek – October 1998.**



**Subsurface sediments** In general, total PCB concentrations increased with depth and decreased with distance from the creek end, returning to background surface concentrations in the 0-1 and 1-2 ft core segments collected east of the 3<sup>rd</sup> Street Bridge (Table 6-8, Figure 6-6). Core segments deeper than 2 ft were not analyzed east of the bridge.

The highest concentrations were measured at Station 1C, located at the center of the creek end near the historical CSO (Table 6-8), ranging from 342.2 to 599.8 ng $\text{g}^{-1}$  in the 0-1 and 2-3 ft core segments, respectively. The only other core segments with concentrations greater than 100 ng $\text{g}^{-1}$  were collected below 2 ft at Stations 2N and 3S.

### 6.3.4 Islais Creek Organochlorine Pesticides

Many individual pesticide compounds were below laboratory detection limits (usually <0.5 ng $\text{g}^{-1}$ ), including Aldrin, Endrin, Lindane, Mirex, and certain individual Chlordane and DDT isomers. Total DDT (including DDE and DDD isomers), total Chlordane and Dieldrin were routinely detected at concentrations exceeding 10 ng $\text{g}^{-1}$ , usually at stations with correspondingly high TOC concentrations (e.g., >2%). Like metals and total PCB, the most elevated pesticide concentrations were measured at Station 3N. Summary results for DDT, Chlordane and Dieldrin in surface sediments are presented in Table 6-9. Distributions of these compounds in surface sediments are shown in Appendix A4.

Subsurface concentrations generally increased with depth and decreased with distance from the creek end. Subsurface distributions for total Chlordane are shown in Figure 6-6; other pesticide distributions are shown in Appendix A5. Station 1C had the highest concentrations of Chlordane, DDT and Dieldrin of all core segments measured (Table 6-10).

**Table 6-10. Islais Creek - mean concentrations of total Chlordane, total DDT and Dieldrin in subsurface sediments (ng $\text{g}^{-1}$ , ppb dry weight).**

Depth	Total Chlordane		Total DDT		Dieldrin	
	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration
0-1 ft	13.3	1C; 60.8	41.9	1C; 181.5	7.4	1C; 30
1-2 ft	16.9	1C; 75.8	34.6	1C; 129.7	9.7	1C; 35
2-3 ft	42.8	1C; 91.0	71.5	1C; 134.9	7.7	1C; 14
3-4 ft	53.2	1C; 98.0	73.5	1C; 124.4	8.8	1C; 15

**Total DDT.** Total DDT averaged 32.9 ng $\text{g}^{-1}$  in surface sediments, with a maximum of 86.5 ng $\text{g}^{-1}$  at the creek end, near the Selby Street CSO (Station 1N). Total DDT also was elevated near the CSO Weir (Station 3N) in the October 1999 dry survey only. The major compounds contributing to total DDT were breakdown products 4,4-DDD and 4,4-DDE (Appendix A1), indicating weathered sources of this relic pesticide. There was little variation in between surveys, except that elevated concentrations observed at Station 1N in 1998 and 2000, were considerably reduced in October 1999 (a dry event). Subsurface sediments had higher total DDT concentrations than surface sediments, with a maximum concentration of 181.5 ng $\text{g}^{-1}$  measured in the 0-1 ft core at Station 1C. Unlike most other contaminants, increased concentrations with depth were not observed for DDT, indicating fairly consistent inputs over time, even though this pesticide was banned nearly 30 years ago.

All sediments had total DDT concentrations far below the RWQCB approved criterion of  $100 \mu\text{g g}^{-1}$  organic carbon (adopted from Schwartz et al. [1994]). DDT concentrations were highly correlated with sediment TOC levels ( $r=0.85$ ,  $p<0.001$ ), indicating that the most elevated concentrations of DDT may not be readily available to benthic organisms (Schwartz et al. 1994).

**Total Chlordane.** Total Chlordane concentration averaged  $21.4 \text{ ng g}^{-1}$  in surface sediments collected west of the 3<sup>rd</sup> Street Bridge and  $13.5 \text{ ng g}^{-1}$  throughout the entire creek. The highest surface concentration was measured in 1998 at Station 1N at  $79 \text{ ng g}^{-1}$ . Concentrations in all creek-end surface sediments (Transects 1-3) were elevated compared with east-end creek (Transects 4-6) and reference area sediments. Much higher subsurface concentrations were measured in the deeper core segments collected west of the 3<sup>rd</sup> Street Bridge (Figure 6-6). However, concentrations returned to reference area levels in the shallower depths (0-2 ft) throughout the creek, except at Station 1C.

The most prevalent isomers of Chlordane were alpha- and gamma-Chlordane, and trans-Nonachlor. Heptachlor and Heptachlor epoxide were not found above detection limits (ca.  $0.5 \text{ ng g}^{-1}$ ). Chlordane, like other non-polar organic compounds, has high affinity for organic matter, as indicated in Table 6-4. Therefore, higher concentrations are expected in sediments with high TOC, providing a contaminant source exists. Use of a TOC-normalized criterion for total Chlordane might help explain why Station 1N (highest in total Chlordane) had the highest amphipod survival (83%) of all creek sediments in 1998.

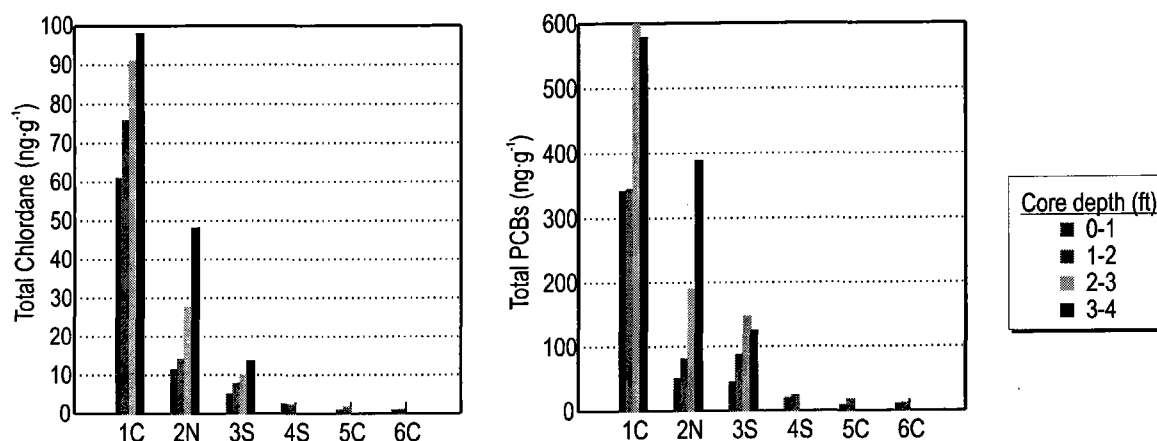


Figure 6-6. Subsurface concentrations of total Chlordane and total PCB at Islais Creek – October 1998.

**Dieldrin.** The average concentration of Dieldrin in surface samples was  $7.3 \text{ ng g}^{-1}$ , with a maximum of  $34.3 \text{ ng g}^{-1}$  measured at Station 3N. Dieldrin had an ERM value of  $8 \text{ ng g}^{-1}$  (Long and Morgan 1991), which was dropped from the 1995 listing due to lack of confidence in the previously published ERM value. Concentrations were substantially reduced in sediments east of the 3rd Street Bridge (see Appendices A1 and A4).

Subsurface core concentrations were significantly elevated compared with surface concentrations at Station 1C only. All other core segments were below  $10 \text{ ng g}^{-1}$ . A maximum of  $35 \text{ ng g}^{-1}$  was recorded from the 0-2 ft segment from the creek center, near the Shelby Street overflow at the creek end (Station 1C) in 1998. Reduced surface concentrations indicate reduced inputs of Dieldrin in recent sediment deposits. Dieldrin, like DDT and Chlordane, was significantly correlated with sediment TOC (Table 6-4).

### 6.3.5 Islais Creek Chemicals of Potential Concern (COPCs)

Chemicals of potential concern were identified from the 16 individual or summary compounds used in the BPTCP. For 1998 data, 13 “compounds” from this list were used to calculate an ERM quotient for each station following BPTCP guidelines, modified as described in Section 1.2.4, where any compound or compound group (e.g., PCBs) with an ERM quotient greater than 0.5 (i.e., concentration > one-half of the ERM) was retained as a preliminary COPC. Chemicals without an ERM were retained as preliminary COPCs if creek concentrations were greater than the corresponding concentration measured at Paradise Cove.

Sediment COPCs were evaluated using TOC-normalized data (i.e., ng or  $\mu\text{g}$  chemical per fractional percent TOC). For 1998 comparisons, a 1% (i.e., 0.01) TOC concentration was assumed for ERM values (see Long et al. 1995), and any “chemical” greater than the TOC-normalized ERM was retained as a preliminary COPC. For 1999 and 2000 data, any chemical greater than the 95<sup>th</sup> UPL (see Table 6-3) calculated using corresponding reference station data was retained as a preliminary COPC. Chemicals were retained as final COPCs if they exceeded either of these criteria at any station in at least two of the three surveys. Additional studies, such as ecological risk assessments, are necessary to determine whether elevated contaminant concentrations are biologically meaningful. This is especially important in the case of metals and PAH, where nearly all creek concentrations were statistically elevated compared to reference conditions; however, very few analytes exceeded one-half the ERM criterion when TOC-normalized.

The remaining chemicals, consisting entirely of chlorinated compounds (e.g., pesticides and PCBs), were elevated in relation to reference sediments and one-half the ERM. DDT was the only chlorinated contaminant that was statistically elevated compared to reference sediments but well below the 1998 numeric criterion – a normalized value published by Schwartz et al. (1994) of  $100 \mu\text{g g}^{-1}$  organic carbon. This inconsistency is due to the fact that TOC-normalized criterion for DDT is 2-4 orders of magnitude greater than ERM values for other chlorinated compounds. TOC-based criteria for sediment contaminants exist and are in use in other regulatory programs (e.g., Washington Department of Ecology); however, DDT is the only compound routinely evaluated in this manner in San Francisco Bay regulatory programs.

Chemicals that were elevated in sediments and known to bioaccumulate in the food web were further evaluated using 28-day clam bioaccumulation tests. Bioaccumulation results are discussed in Section 7. Chemicals of potential concern for Islais Creek are shown in Table 6-11, which displays exceedance factors for each station based on either one-half the ERM (1998 data) or the reference area UPL (1999 and 2000 data). For example, the value displayed for lead at Station 1N in 1999 indicates that the lead concentration was 7.83 times greater than the 95<sup>th</sup> UPL calculated using 1999 combined reference station data for TOC-normalized lead; and the 1998 lead concentration was 0.66 times one-half the ERM value of  $109 \mu\text{g}\cdot\text{g}^{-1}$  sediment or  $10,900 \mu\text{g}\cdot\text{g}^{-1}$  TOC (i.e., 1%).

All stations west of the 3<sup>rd</sup> Street Bridge had four or more COPCs, with Chlordane, DDT, PCBs and PAHs the most ubiquitous. Three metals - cadmium, lead and zinc, qualified as COPCs; however, only zinc was elevated east of Transect 1. Chlordane, DDT, PCBs and mercury were further evaluated for bioaccumulation potential in 28-day bioaccumulation tests (Section 7).

Although preliminary COPCs are statistically elevated at creek stations compared to the reference area or one-half of the ERM value, additional studies, such as ecological risk assessments, are necessary to determine whether concentrations are sufficient to negatively impact the local ecology.

**Table 6-11. Islais Creek - Surface sediment COPCs and corresponding ratios for station concentrations and corresponding guideline values.**

COPC	Survey Year	1N*	1S*	2N*	2S*	3N*	3S*
Cadmium	1998	0.08	0.17	0.06	0.05	0.11	0.07
	1999	<b>3.33</b>	<b>1.98</b>	0.68	0.85	<b>1.45</b>	0.86
	2000	0.57	<b>1.23</b>	<b>1.14</b>	0.65	0.51	0.72
Lead	1998	0.66	<b>1.97</b>	0.16	0.17	0.37	0.17
	1999	<b>7.83</b>	<b>6.78</b>	0.53	0.69	<b>1.33</b>	0.62
	2000	<b>2.29</b>	<b>12.43</b>	<b>1.32</b>	<b>1.19</b>	0.86	<b>1.22</b>
Zinc	1998	<b>1.47</b>	1.00	0.39	0.36	0.49	0.46
	1999	<b>1.54</b>	<b>1.4</b>	0.43	0.51	0.79	0.54
	2000	0.66	<b>1.55</b>	0.87	0.83	0.71	0.98
LMW PAH	1998	0.42	0.16	0.49	0.29	0.45	0.39
	1999	<b>1.32</b>	<b>1.28</b>	<b>4.9</b>	<b>3.8</b>	<b>3.79</b>	<b>3.83</b>
	2000	0.7	<b>1.33</b>	<b>9.4</b>	<b>3.15</b>	<b>4.96</b>	<b>9.83</b>
HMW PAH	1998	0.18	0.24	0.93	0.46	0.68	0.52
	1999	<b>1.25</b>	<b>1.21</b>	<b>5.37</b>	<b>3.03</b>	<b>4.28</b>	<b>3.17</b>
	2000	<b>4.09</b>	<b>1.17</b>	<b>9.45</b>	<b>2.58</b>	<b>4.82</b>	<b>5.84</b>
Total Chlordane	1998	0.27	<b>10.2</b>	<b>1.1</b>	<b>1.9</b>	<b>4.4</b>	0.85
	1999	<b>1.16</b>	<b>26.6</b>	<b>4.67</b>	<b>9.19</b>	<b>14.35</b>	<b>4.71</b>
	2000	<b>1.82</b>	<b>78.0</b>	<b>48.7</b>	<b>29.2</b>	<b>21.1</b>	<b>38.1</b>
Total DDT	1998	<b>5.46</b>	0.02	0.01	0.01	0.02	0.01
	1999	<b>22.5</b>	<b>3.74</b>	0.97	<b>1.53</b>	<b>2.06</b>	<b>1.01</b>
	2000	<b>51.4</b>	<b>1.82</b>	<b>1.06</b>	0.77	0.7	0.99
Dieldrin	1998	0.02	<b>2.34</b>	0.55	0.8	<b>2.39</b>	0.48
	1999	<b>2.39</b>	0.09	0.08	0.58	0.82	0.48
	2000	<b>1.2</b>	<b>9.39</b>	<b>8.5</b>	<b>6.01</b>	<b>5.49</b>	<b>12.83</b>
Total PCB	1998	<b>1.76</b>	<b>2.02</b>	0.25	0.58	<b>1.28</b>	0.21
	1999	0.93	<b>13.6</b>	<b>1.47</b>	<b>1.96</b>	<b>3.97</b>	<b>1.56</b>
	2000	<b>7.24</b>	<b>13.7</b>	<b>2.79</b>	<b>2.40</b>	<b>1.91</b>	<b>3.04</b>

Red > 0.5 x ERM (1998 only); **Bold** > reference upper 95% predictive limit (1999 & 2000); \*=recurrent contamination measured in ?2 surveys

## 6.4 MISSION CREEK

Sediments were sampled at 13 locations in 1998, spanning from the creek end (Transect 1) to the mouth (Transect 6). Transects 1-4, consisting of eight stations located from the creek end to the west side of the 4<sup>th</sup> Street Bridge, were resampled in 1999 and 2000. Sampling locations are shown in Figure 3-2, Section 3 for all surveys.

In general, surface sediment chemical concentrations measured east of 5<sup>th</sup> Street (Transects 4-6) and more often east of 4<sup>th</sup> Street (Transects 5-6) were commensurate with sediment concentrations measured at in-bay reference stations. Chemical distribution patterns in surface sediments were fairly consistent over time, with the highest concentrations measured near the CSO structures at 6<sup>th</sup> Street, approximately 150 m from the main CSO. Most chemical concentrations decreased with distance from the CSOs located at 6<sup>th</sup> and Berry Streets, and the Interstate 280 overpass, except for HMW PAH, which displayed elevated concentrations in sediments adjacent to creosote-soaked pier pilings located between 5<sup>th</sup> and 4<sup>th</sup> Streets (Transects 3 and 4). Surface sediment distributions of key chemicals are shown for each survey in Appendix B4. In general, chemical concentrations increased significantly with depth, indicating that buried contaminants are most likely in-place and that inputs have diminished over time. Subsurface distribution plots for key chemicals are shown in Appendix B5 for 1-ft core intervals sampled from 0-4 ft depth in October 1998.

Many organic as well as inorganic contaminant concentrations were significantly and positively correlated with TOC as shown in Table 6-12. Inverse relationships were observed for grain size (percent fines), as all significant correlations indicated a decrease in contaminant concentration with increasing percent fines (i.e.,  $r < 0$ ).

**Table 6-12. Mission Creek - correlation results for selected chemicals with TOC and grain size.**

	Copper	Mercury	Lead	Zinc	Total DDT	Dieldrin	HMW PAH	LMW PAH	Total PAH	Total PCB	Total Chlordane
<b>Percent Fines</b>											
r	-0.18	-0.38	<b>-0.56</b>	<b>-0.52</b>	-0.27	<b>-0.49</b>	-0.19	<b>-0.55</b>	-0.32	<b>-0.46</b>	-0.26
p	0.35	0.05	<0.001	<0.001	0.16	0.01	0.33	<0.001	0.10	0.01	0.17
<b>TOC</b>											
r	<b>0.59</b>	<b>0.43</b>	<b>0.42</b>	<b>0.60</b>	<b>0.40</b>	<b>0.72</b>	<b>0.39</b>	0.14	0.33	<b>0.44</b>	0.36
p	<0.001	0.02	0.02	<0.001	0.03	<0.001	0.04	0.47	0.08	0.02	0.06

r=correlation coefficient; p=probability level; significant correlations ( $p < 0.05$ ) are shown in **bold**

### 6.4.1 Mission Creek Metals

**Surface results.** Concentrations of mercury, lead, nickel, silver and zinc exceeded their corresponding ERM value at several stations in all three surveys. Overall, the highest concentrations were found at Transects 1 and 2, located at the west end of the creek. Although highest at the creek end, concentrations of lead and mercury were extremely variable between stations and surveys. For example, at Station 2S, mercury was  $3.76 \text{ } \mu\text{g g}^{-1}$  in 1999, and 0.72 and  $0.84 \text{ } \mu\text{g g}^{-1}$  in 1998 and 2000, respectively. Mercury, lead, zinc and silver were highest in October 1999, a dry event, at Station 2N (Table 6-13, Figure 6-7).

Selenium concentrations ranged from 0.10 to 0.88  $\mu\text{g g}^{-1}$ , with an average concentration of 0.38  $\mu\text{g g}^{-1}$ . Selenium concentrations below 0.33  $\mu\text{g g}^{-1}$  are reported as uncontaminated background for San Francisco Bay sediments (Walters and Gartner 1985). The ERM for nickel at 51.6  $\mu\text{g g}^{-1}$ , was exceeded at nearly all stations; however, nickel is generally not considered a COPC, as naturally elevated concentrations are found in non-toxic sediments throughout San Francisco Bay (Hunt et al. 1998a).

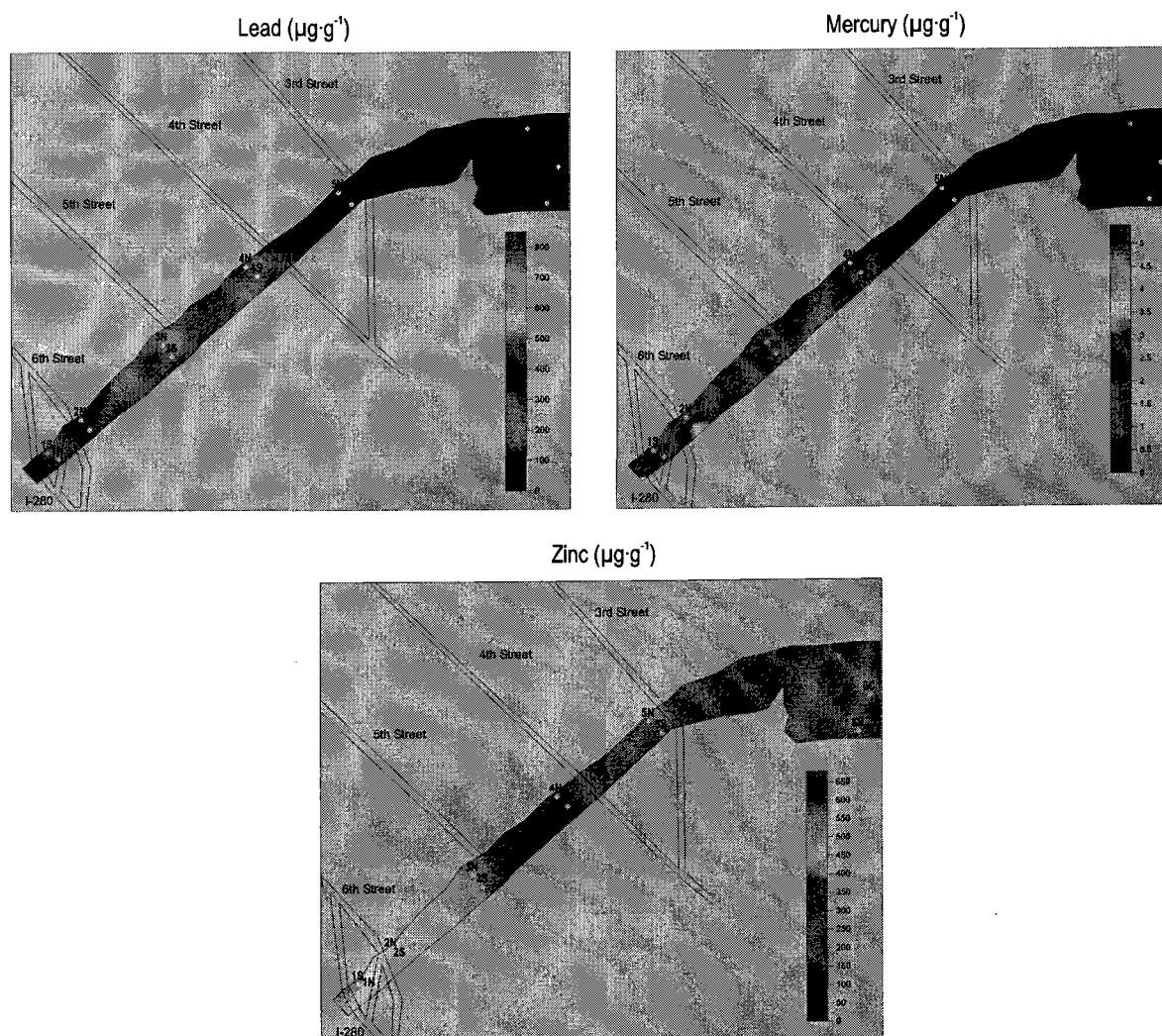
Most metals, including cadmium, copper, lead, mercury, silver and zinc were significantly correlated with sediment TOC. Correlation coefficients for selected metals are shown in Table 6-12. Like Islais Creek, sediments in Mission Creek with the highest concentrations of TOC also displayed the highest metal concentrations. TOC concentrations in Transects 2 and 3 were consistently greater than 2%, ranging up to 4.5% at Station 2N.

October 1998 surface distributions for lead, mercury and zinc throughout the creek are shown in Figure 6-7. Surface distributions for these and other metals for all surveys are shown in Appendix B4.

**Table 6-13. Mission Creek Transects 1-4 (24 stations) surface sediment metal concentrations ( $\mu\text{g g}^{-1}$ , ppm dry weight).**

Metal	Minimum	Maximum	Maximum Station	Mean	Standard Deviation	ERM <sup>1</sup>	Reference Mean <sup>2</sup>
Arsenic	5.5	17.72	4N	10.54	2.83	70	6.98
Cadmium	0.63	2.94	2N	1.57	0.5	9.6	0.32
Chromium	82	124	4S	103.28	12.19	370	96.4
Copper	89.6	161	2S	127.99	19.72	270	38.1
Lead	90.1	858.3	2N	296.27	170.11	218	18.0
Mercury	0.55	5.37	2N	1.47	1.18	0.71	0.23
Nickel	50.6	104	3N	81.15	15.95	51.6	86.7
Selenium	0.1	0.88	3N	0.38	0.2	NA	0.25
Silver	0.87	6.29	2N	3.18	1.53	3.7	0.52
Zinc	210	678.6	2N	377.41	105.40	410	103.2

<sup>1</sup>=source Long et al. (1995); <sup>2</sup>=mean reference site concentration for all surveys



**Figure 6-7. Distribution of lead, mercury and zinc at Mission Creek – October 1998.**

**Subsurface results.** Mission Creek subsurface sediments displayed similar metal distributions to those measured in Islais Creek, with concentrations generally increasing with depth and decreasing with distance from the CSOs at 6<sup>th</sup> and Berry Streets, and the Interstate 280 overpass. Representative vertical profiles for lead and zinc are shown in Figure 6-8. Plots for other key metals are shown in Appendix B5.

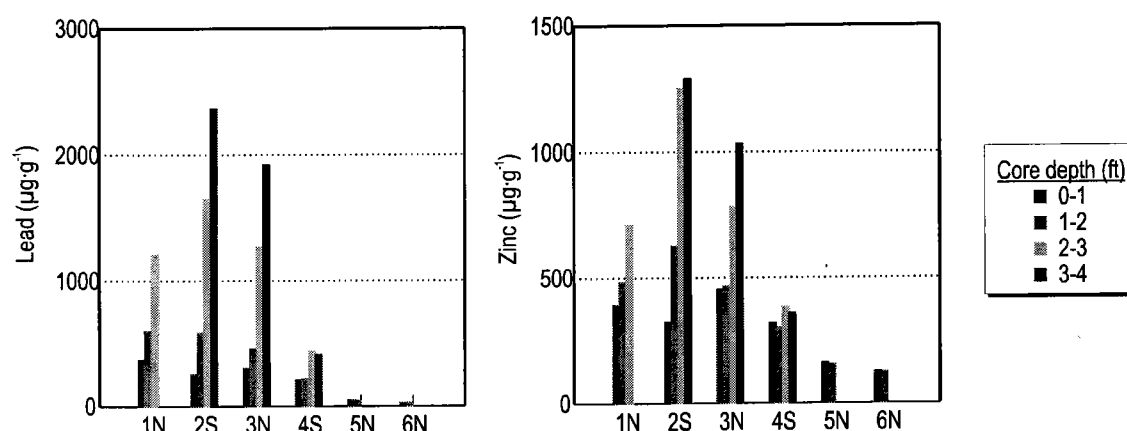
The highest concentrations of lead, mercury and zinc were measured in the 3–4 ft interval at Station 2S (Table 6-14). Concentrations of these and other metals were commensurate with in-bay surface sediment reference concentrations in the 0–1 and 1–2 ft intervals, east of 4<sup>th</sup> Street, where deeper cores were not analyzed. Concentrations in the shallower cores were significantly lower, with the most dramatic decreases observed above 2 ft, indicating substantial decreases in metal inputs over time west of 4<sup>th</sup> Street. Concentrations of arsenic, chromium and nickel were fairly consistent across stations and depths, indicating no obvious anthropogenic inputs of these metals to creek sediments. Other metals, including



silver and selenium also were fairly stable, with the exception of localized spikes at the 2-4 ft depth at Transects 2 and 3 (see Appendix B5). Concentrations of aluminum and iron decreased significantly west of 6<sup>th</sup> Street, indicating an abrupt change in sediment mineralogy at the creek end. These metals, generally not considered contaminants, are used to identify various geological origins of sediment metals. Aluminum/metal relationships are evaluated to identify potential sources of elevated metals in Section 8.

**Table 6-14. Mission Creek - mean concentrations of lead, mercury and zinc in subsurface sediment ( $\mu\text{g g}^{-1}$ , ppm dry weight).**

Depth	Lead		Mercury		Zinc	
	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration
0-1 ft	202.0	1N; 372.0	0.76	3N; 1.28	295.8	3N; 455.0
1-2 ft	321.3	1N; 597.0	1.06	1N; 1.60	358.5	2S; 626.0
2-3 ft	1140.8	2S; 1646.5	2.40	2S; 3.77	781.5	2S; 1250.9
3-4 ft	1564.9	2S; 2362.7	3.74	2S; 6.27	893.1	2S; 1288.6



**Figure 6-8. Subsurface concentrations of lead and zinc at Mission Creek.**

#### 6.4.2 Mission Creek Polycyclic Aromatic Hydrocarbons (PAH)

**Surface results.** Concentrations of PAH, like metals, in Mission Creek surface sediments varied considerably as a function of location, sediment type and TOC concentration. Higher relative concentrations of HMW PAH compared to LMW PAH were measured in all surface sediments. Mean concentrations were 2826 and 9741  $\text{ng g}^{-1}$ , respectively, for LMW and HMW PAH measured in Transects 1-4 in all surveys (Table 6-15). PAH concentrations were consistently elevated at Stations 1N and 4S in all three surveys; however, distributions of individual PAH compounds indicated different contaminant sources for these two areas (see Section 8 for discussion on PAH source). Although

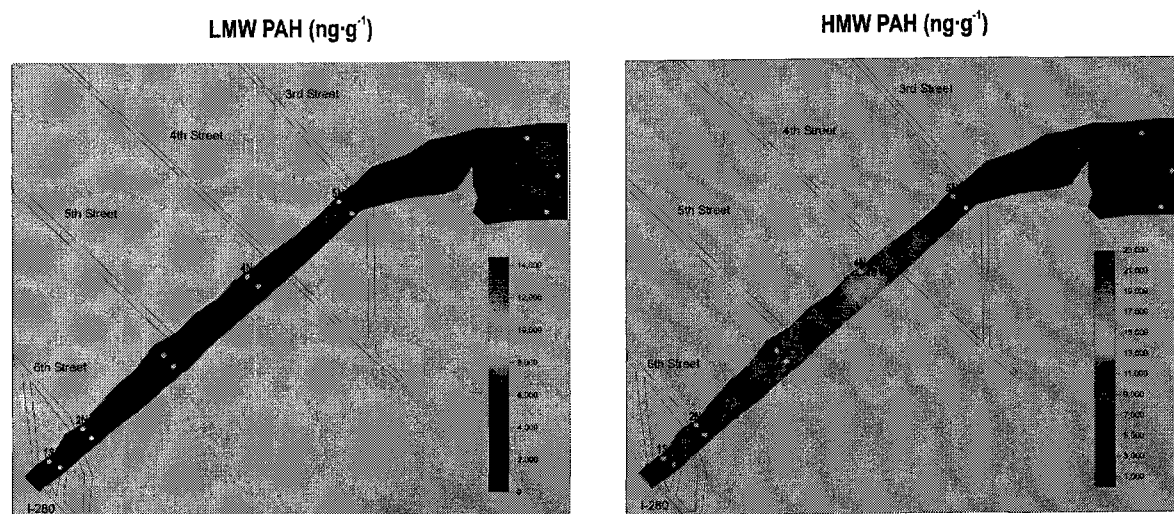
Stations 1N and 4S had consistently high PAH concentrations, high concentrations of TOC (i.e., > 2%) were measured only in 1998. A weak, but significant correlation was observed between TOC and HMW PAH only, using all station data (Table 6-12).

Figure 6-8 shows the horizontal distribution of LMW- and HMW-PAH in 1998, where both groups were elevated at the creek origin near the main CSO (Transect 1) and at 4<sup>th</sup> Street, primarily on the south side. Surface sediment concentrations east of the 4<sup>th</sup> Street Bridge (Transects 5 & 6) remained elevated in relation to reference sediments; however, these sediments were much lower than corresponding ERM values for LMW- and HMW-PAH (Table 6-15).

**Table 6-15. Mission Creek - PAH in surface sediments, all studies combined (ng·g<sup>-1</sup>, ppb dry weight).**

Parameter	Minimum	Maximum	Maximum Station	Mean	ERM <sup>1</sup>	Reference Mean
<b>Transects 1-4 (24 stations)</b>						
LMW PAH	655	11,492	1N	2826	3160	118
HMW PAH	3140	23,390	1N	9741	9600	562
<b>Transects 5-6 (5 stations)</b>						
LMW PAH	441	690	5N	587	3160	118
HMW PAH	1451	2919	5S	2187	9600	562

<sup>1</sup>=source Long et al. (1995)



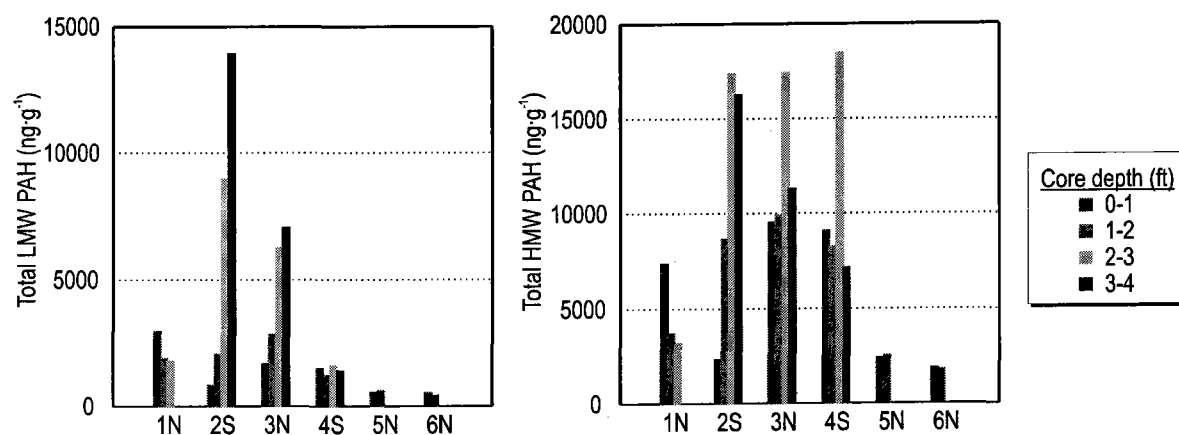
**Figure 6-9. Distribution of LMW and HMW PAH at Mission Creek – October 1998.**

**Subsurface results.** In general, concentrations of most PAH compounds, including summary compounds LMW- and HMW-PAH increased with depth and decreased with distance from Transect 2, located near the creek end; however, these trends were much more pronounced for LMW compounds (Figure 6-10). Concentrations of LMW PAH were especially elevated at Stations 2S and 3N, reaching a maximum concentration of 13,930 ng·g<sup>-1</sup> in the 3-4 ft core segment at Station 2S. All of the subsurface cores

collected from 0-2 ft depth had LMW PAH concentrations below the ERM guideline (i.e.,  $<3160 \text{ ng g}^{-1}$ ); however, concentrations in the deeper cores were substantially higher, particularly near the creek end. Concentrations of HMW PAH were elevated across all depths in cores collected west of the 4<sup>th</sup> Street Bridge, with the highest concentrations measured in the deeper cores at Stations 2N, 3N and 4S. Subsurface sediment distributions indicate that diminishing but chronic inputs of varying PAH sources exist near the 6<sup>th</sup> and Berry Street CSO (Station 2S) and between Transects 3 and 4 (most likely from creosote-soaked pier pilings), as corresponding surface sediments had elevated concentrations as well. Potential PAH sources to these sediments are evaluated using chemical fingerprinting methods in Section 8.

**Table 6-16. Mission Creek - mean concentrations of LMW PAH, HMW PAH and total PCB in subsurface sediments ( $\text{ng g}^{-1}$ , ppb dry weight).**

Depth	LMW PAH		HMW PAH		Total PCB	
	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration
0-1 ft	1344	1N; 2974	5428	3N; 9520	202.9	1N; 390.2
1-2 ft	1507	3N; 2850	5803	3N; 9890	292.3	3N; 570.8
2-3 ft	4648	2S; 8945	14,122	4S; 18,480	923.4	2S; 1471.9
3-4 ft	7462	2S; 13,930	11,570	2S; 16,280	1598.0	3N; 2760.4



**Figure 6-10. Subsurface concentrations of LMW- and HMW-PAH at Mission Creek – October 1998.**

### 6.4.3 Mission Creek Polychlorinated Biphenyls (PCB)

**Surface results.** Horizontal distributions of total PCB in Mission Creek were similar to those observed for PAH, varying considerably as a function of location, sediment type and TOC concentration. Total PCB sediment concentrations ranged from 13.3 to  $869.9 \text{ ng g}^{-1}$  throughout the creek, with the highest concentrations measured at the creek end (Transects 1 & 2). Figure 6-11 (left) shows the surface

distribution of total PCB measured in October 1998 throughout the creek. Surface distributions for sediments sampled in 1999 and 2000 are shown in Appendix B4. The mean total PCB concentration for sediments collected east of 4<sup>th</sup> Street was approximately double ( $25.2 \text{ ng g}^{-1}$ ) the mean concentration measured at in-bay reference stations for all surveys ( $10.16 \text{ ng g}^{-1}$ ), although concentrations were more than an order of magnitude (ca. 10x) lower than those measured west of 4<sup>th</sup> Street. Transect 6 concentrations were commensurate with in-bay reference areas, indicating that little, if any creek end PCBs are transported to the creek mouth and bay. The distribution of total PCB, like PAH, was significantly correlated with TOC (Table 6-12). Range and mean concentrations of total PCB measured in creek and reference area surface sediments in all three surveys are shown in Table 6-17.

Results from the 1998 survey indicated that all PCBs were present in approximately equal concentrations of weathered Aroclor 1254 and 1260. Total Aroclor concentrations were approximately double those measured for total PCB, which was based on 18 out of 209 possible PCB congeners. PCB congeners, but not Aroclors, were quantified in the following two surveys.

**Table 6-17. Mission Creek – total PCB and selected pesticides in surface sediments, all surveys combined ( $\text{ng g}^{-1}$ , ppb dry weight).**

Parameter	Minimum	Maximum	Maximum Station	Mean	Guideline Value	Reference Mean (all surveys)
<b>Transects 1-4</b>						
Total DDT	23.4	229	2N	71.4	NA	5.67
Total Chlordane	15.5	306	2N	71.6	6 <sup>1</sup>	0.90
Dieldrin	0.5	60	1S	16.6	8 <sup>1</sup>	0.7
Total PCB	107.3	870	2N	337.0	180 <sup>2</sup>	10.16
<b>Transects 5-6 (1998 only)</b>						
Total DDT	7.8	18.4	5S	11.7	NA	5.67
Total Chlordane	0.6	6.6	5S	2.7	6 <sup>1</sup>	0.90
Dieldrin	1.2	4.7	5S	2.4	8 <sup>1</sup>	0.70
Total PCB	13.3	46.3	5S	25.2	180 <sup>2</sup>	10.16

<sup>1</sup>=source Long & Morgan (1991); <sup>2</sup>=source Long et al. (1995)

**Subsurface results** In general, the vertical distribution of total PCB concentrations in sediments increased with depth and distance from Transects 1 through 3, where a maximum concentration of  $2760 \text{ ng g}^{-1}$  was measured in the 3-4 ft core segment at Station 3N (Figure 6-11, right). Overall concentrations decreased rapidly east of the 4<sup>th</sup> Street Bridge (Station 4S) (mean= $38.5 \text{ ng g}^{-1}$ ), following trends observed for other subsurface contaminants.

Subsurface PCB concentrations were highest in the deeper cores at all stations, except 1N. Concentrations greater than  $1000 \text{ ng g}^{-1}$  were measured in the 2-3 and 3-4 ft core segments at Stations 2S and 3N (Figure 6-11). However, concentrations measured in the 0-1 and 1-2 ft core segments were fairly consistent west of 4<sup>th</sup> Street, averaging  $202.9$  and  $293.3 \text{ ng g}^{-1}$ , respectively. The maximum concentration measured in the 0-1 ft core segment was  $390 \text{ (ng g}^{-1})$  at Station 1N, where a higher surface maxima was also recorded in 1998.

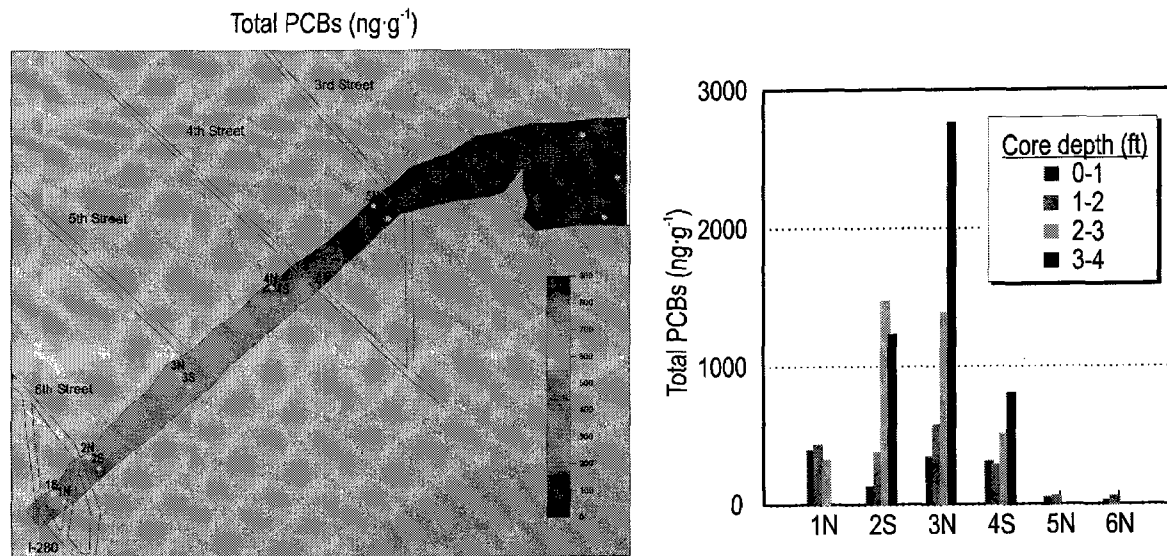


Figure 6-11. Surface and subsurface distributions of total PCB at Mission Creek – October 1998.

#### 6.4.4 Mission Creek Organochlorine Pesticides

Many of the individual pesticide compounds were present at trace concentrations near the detection limit (usually  $< 0.5 \text{ ng} \cdot \text{g}^{-1}$ ), including Aldrin, Endrin, Lindane, Mirex, and many of the individual Chlordane and DDT isomers. Total DDT, total Chlordane and Dieldrin were routinely measured at concentrations exceeding  $1 \text{ ng} \cdot \text{g}^{-1}$ ; however, all but total Chlordane were significantly correlated with TOC (Table 6-12). Surface sediment concentrations varied widely across surveys, with the greatest concentrations of DDT and Chlordane occurring in the 1998 and 2000 wet weather surveys. The highest concentrations of Dieldrin, the only other significantly elevated organochlorine pesticide, also were detected in October 1998 during wet weather. However, significantly decreased concentrations were measured in the following two surveys.

Subsurface maxima for all compounds were measured in the deeper cores at either 6<sup>th</sup> or 5<sup>th</sup> Street (Transects 2 or 3) near the creek end. Summary results for surface and subsurface sediments for selected pesticides are presented in Tables 6-17 and 6-18, respectively.

Table 6-18. Mission Creek - mean concentrations of total Chlordane, total DDT and Dieldrin in subsurface sediments ( $\text{ng} \cdot \text{g}^{-1}$ , ppb, dry weight).

Depth	Total Chlordane		Total DDT		Dieldrin	
	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration	Mean	Max. Station; Concentration
0-1 ft	41.1	1N; 94.2	63.0	4S; 107.5	20.8	3N; 40
1-2 ft	87.2	1N; 201	67.1	2S; 171.6	26.7	3N; 62
2-3 ft	362.1	3N; 626	274.6	2S; 510.3	44.3	2S; 93
3-4 ft	679.7	3N; 922	423.9	2S; 645.2	71.0	3N; 120

**DDT.** Total DDT averaged  $44.6 \text{ ng g}^{-1}$  in surface sediments, with a maximum of  $228 \text{ ng g}^{-1}$  at Station 2N, near the 6<sup>th</sup> and Berry Street CSO. Following patterns observed in Islais Creek, the major DDT metabolites contributing to total DDT were weathered 4,4-DDD and 4,4-DDE isomers (Appendix B1). Total DDT concentrations of  $60 \text{ ng g}^{-1}$  and higher in surface sediments also were recorded at Stations 1N, 2N, 3N and 3S in the creek end west of 5<sup>th</sup> Street. Subsurface sediments had higher DDT concentrations than surface sediments, averaging  $63.0$  to  $423.9 \text{ ng g}^{-1}$  from core segments of 0-1 to 3-4 ft, respectively. Maximum concentrations of  $645.2$  and  $510.3 \text{ ng g}^{-1}$  were collected from the deeper cores of 2-3 and 3-4 ft at Station 2S. Comparatively lower total DDT concentrations in surface sediments indicate a trend of declining DDT in more recent deposits.

All surface sediments had total DDT concentrations below the BPTCP criterion of  $100 \text{ ng g}^{-1}$  organic carbon, even in samples containing 1% or less TOC. A maximum total DDT concentration of  $229 \text{ ng g}^{-1}$  was measured at Station 2S in April 1999, which had a corresponding TOC concentration of 3.2%. The corresponding sediment dry weight criterion (adopted from Schwartz et al. [1994]) would be  $3200 \text{ ng g}^{-1}$  for this sample. DDT was significantly correlated with sediment TOC as indicated in Table 6-12. Further examination showed that stations with greater than  $50 \text{ ng g}^{-1}$  total DDT had correspondingly high concentrations of TOC (i.e., >2.5%). However, this relationship was not verified for subsurface sediments collected below 2 ft, as TOC was not measured in these samples.

**Chlordane.** Total Chlordane concentrations averaged  $71.6$  and  $2.7 \text{ ng g}^{-1}$  in all surface sediments collected from Transects 1-4 and 5-6, respectively. The average concentration for all in-bay reference sediments was extremely low at  $0.25 \text{ ng g}^{-1}$ . Concentrations measured in the deeper core segments reached  $922 \text{ ng g}^{-1}$ , averaging 10-100 times most creek surface concentrations (Table 6-18). The highest surface concentration was measured at Station 2N during dry weather in 1999 at  $382 \text{ ng g}^{-1}$ . The highest concentrations of subsurface Chlordane concentrations were measured west of 5<sup>th</sup> Street in the 2-3 and 3-4 ft core segments at Station 3N, located adjacent to the CSO Weir. The most prevalent isomers of Chlordane were alpha- and gamma-Chlordane, and trans-Nonachlor. Heptachlor and Heptachlor epoxide were not measured above detection limits ( $<0.25$ - $0.5 \text{ ng g}^{-1}$ ) in any creek sediments.

Chlordane, like other nonionic organic compounds, usually has high affinity for organic matter; no significant correlation was observed between total Chlordane and TOC for all Mission Creek surface sediments (Table 6-12); however, the relationship was significant (i.e.,  $r=0.82$ ,  $p<0.001$ ) when total Chlordane concentrations exceeded  $30 \text{ ng g}^{-1}$ .

**Dieldrin** The average concentration of Dieldrin in surface sediments collected from Transects 1-4 was  $16.5 \text{ ng g}^{-1}$ , with a maximum of  $60 \text{ ng g}^{-1}$  at Station 1S near the creek origin. Similar to other contaminants, concentrations of Dieldrin dropped significantly east of 5<sup>th</sup> Street, returning to background levels of less than  $3 \text{ ng g}^{-1}$  at the creek mouth (Transect 6).

Subsurface sediment concentrations at shallower depths were similar to surface concentrations, averaging  $20.8$  and  $26.7 \text{ ng g}^{-1}$  from the 0-1 ft and the 1-2 ft segments, respectively. Subsurface maxima of  $93$  and  $120 \text{ ng g}^{-1}$  were recorded from the 3-4 ft core at Stations 2S and 3N, respectively. All surface and subsurface concentrations near the creek mouth (Transects 5 and 6) were at or below  $5 \text{ ng g}^{-1}$ . Dieldrin showed the same pattern as total DDT and total Chlordane, with concentrations generally increasing with depth and decreasing with distance from Transect 2 (Appendix B5).

**Other Pesticides.** Nearly all other organochlorine pesticides, including Aldrin, Endrin and Mirex had surface sediment concentrations below detection limits (i.e.,  $<0.5 \text{ ng}\cdot\text{g}^{-1}$ ). Only Lindane was detected in surface sediment slightly above the detection limit west of 5<sup>th</sup> Street in October 1999, with all concentrations below  $0.5 \text{ ng}\cdot\text{g}^{-1}$ . Lindane and Mirex were detected in the deeper cores at the creek end; however, all concentrations were below  $5 \text{ ng}\cdot\text{g}^{-1}$ .

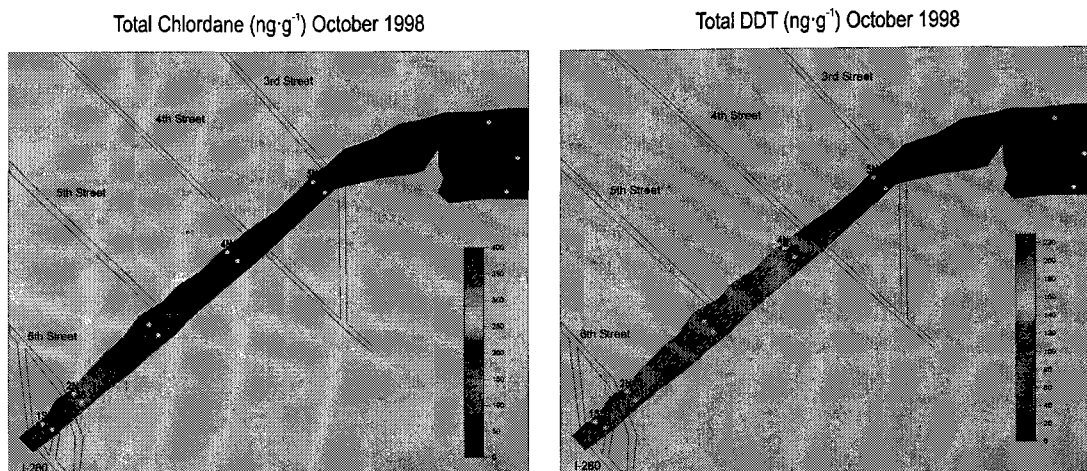


Figure 6-12. Distribution of total Chlordane and total DDT in Mission Creek – October 1998

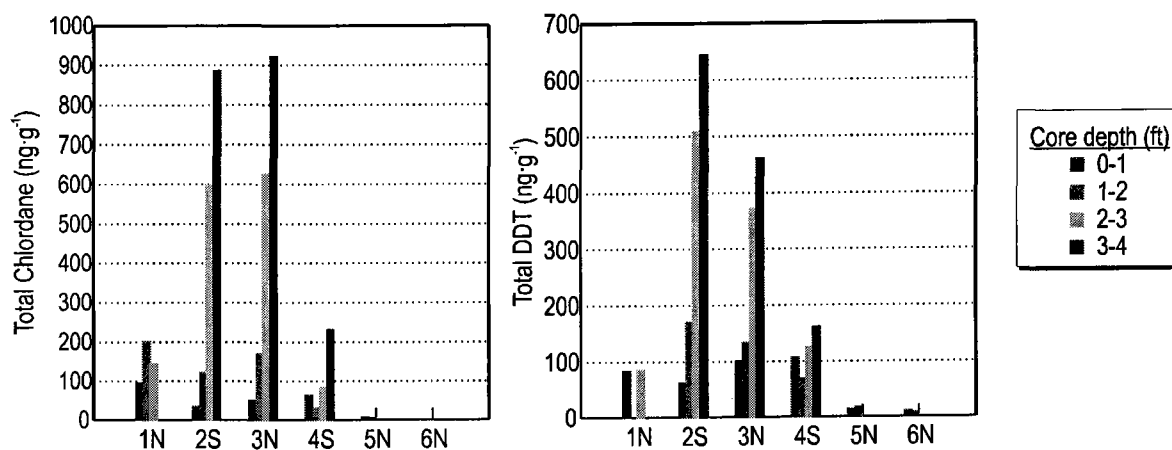


Figure 6-13. Subsurface distributions of total Chlordane and total DDT in Mission Creek – October 1998.

#### 6.4.5 MISSION CREEK CHEMICALS OF POTENTIAL CONCERN (COPCs)

Chemicals of potential concern were selected following the same procedure used for Islais Creek (see Section 6.3.5). Table 6-19 displays Mission Creek COPCs, identified for each station by exceedance factors using TOC-normalized data based on either one-half the ERM (1998 data) or the upper 95<sup>th</sup> predictive limit for corresponding reference stations (1999 and 2000 data). For example, the “exceedance factor” displayed for cadmium at Station 1N in 1999 indicates that the TOC-normalized concentration was 2.97 times greater than the upper 95<sup>th</sup> predictive limit calculated using 1999 combined reference station data.

All stations west of the 4<sup>th</sup> Street Bridge had seven or more COPCs, with lead, zinc, Chlordane, DDT, PCBs and PAHs the most ubiquitous. Six metals qualified as COPCs primarily because they were statistically elevated compared to reference concentrations. Only lead, mercury and marginally silver (1.03x at one station) exceeded half the corresponding ERM value, and this occurred only at the creek end. Many of the chlorinated pesticides, including Chlordane, DDT, and Dieldrin posted exceedance factors much greater than one (1) in Table 6-19, primarily because either all or most of the reference area concentrations were at or below sub-part-per-billion detection limits. DDT was the only chlorinated contaminant that was statistically elevated compared to reference sediments but well below the 1998 numeric criterion – a normalized value published by Schwartz et al. (1994) of 100  $\mu\text{g g}^{-1}$  organic carbon.



**Table 6-19. Mission Creek - Surface sediment COPCs and corresponding ratios for station concentrations and corresponding guideline values.**

COPC	Survey Year	1N*	1S*	2N*	2S*	3N*	3S*	4N*	4S*
Cadmium	1998	0.13	0.13	0.07	0.09	0.09	0.1	0.07	0.08
	1999	<b>2.97</b>	<b>3.39</b>	<b>2.23</b>	<b>3.52</b>	<b>1.65</b>	<b>2.67</b>	<b>2.01</b>	<b>2.4</b>
	2000	<b>1.73</b>	<b>1.37</b>	<b>1.09</b>	<b>1.01</b>	<b>1.09</b>	<b>1.11</b>	0.75	0.84
Copper	1998	0.31	0.31	0.21	0.28	0.26	0.33	0.27	0.27
	1999	<b>1.26</b>	0.93	0.71	<b>1.29</b>	0.75	<b>1.23</b>	<b>1.41</b>	<b>1.22</b>
	2000	<b>1.11</b>	0.97	0.87	0.96	0.99	<b>1.12</b>	<b>1.31</b>	<b>1.43</b>
Lead	1998	<b>1.31</b>	<b>1.12</b>	0.77	0.93	0.52	0.62	0.44	0.52
	1999	<b>10.0</b>	<b>7.24</b>	<b>9.19</b>	<b>9.4</b>	<b>2.66</b>	<b>4.42</b>	<b>3.37</b>	<b>3.75</b>
	2000	<b>13.7</b>	<b>8.45</b>	<b>8.25</b>	<b>8.08</b>	<b>3.49</b>	<b>3.55</b>	<b>2.96</b>	<b>3.1</b>
Mercury	1998	<b>1.46</b>	<b>1.53</b>	0.67	<b>2.54</b>	0.68	0.88	0.7	0.74
	1999	<b>1.79</b>	<b>1.69</b>	<b>5.17</b>	<b>2.21</b>	<b>1.01</b>	<b>1.59</b>	<b>1.88</b>	<b>1.59</b>
	2000	<b>3.01</b>	<b>2.07</b>	<b>2.79</b>	<b>3.74</b>	<b>1.8</b>	<b>1.13</b>	0.99	<b>1.08</b>
Silver	1998	<b>1.03</b>	0.45	0.37	0.75	0.43	0.4	0.3	0.5
	1999	<b>2.2</b>	<b>2.84</b>	<b>2.25</b>	<b>2.98</b>	<b>1.27</b>	<b>2.04</b>	<b>1.35</b>	<b>1.56</b>
	2000	<b>9.65</b>	<b>5.53</b>	<b>7.75</b>	<b>7.2</b>	<b>4.73</b>	<b>3.71</b>	<b>2.22</b>	<b>2.43</b>
Zinc	1998	0.79	0.79	0.49	0.57	0.47	0.57	0.45	0.45
	1999	<b>1.71</b>	<b>1.67</b>	<b>1.42</b>	<b>2.14</b>	<b>0.89</b>	<b>1.56</b>	<b>1.28</b>	<b>1.28</b>
	2000	<b>2.19</b>	<b>1.93</b>	<b>1.43</b>	<b>1.36</b>	<b>1.15</b>	<b>1.24</b>	<b>1.18</b>	<b>1.18</b>
LMW PAH	1998	0.72	0.76	0.35	0.31	0.3	0.27	0.53	0.48
	1999	<b>42.1</b>	<b>5.65</b>	<b>4.13</b>	<b>3.6</b>	<b>2.75</b>	<b>3.67</b>	<b>10.3</b>	<b>6.12</b>
	2000	<b>34.0</b>	<b>8.71</b>	<b>5.15</b>	<b>4.37</b>	<b>3.04</b>	<b>2.56</b>	<b>3.52</b>	<b>7.98</b>
HMW PAH	1998	0.62	0.64	0.36	0.38	0.38	0.4	0.85	0.88
	1999	<b>13.6</b>	<b>3.73</b>	<b>3</b>	<b>2.79</b>	<b>2.55</b>	<b>3.26</b>	<b>8.02</b>	<b>7.35</b>
	2000	<b>15.0</b>	<b>5.22</b>	<b>2.46</b>	<b>2.63</b>	<b>2.23</b>	<b>2.63</b>	<b>3.78</b>	<b>8.3</b>
Total Chlordane	1998	<b>9.2</b>	<b>8.9</b>	<b>6.3</b>	<b>7.9</b>	<b>4</b>	<b>4.3</b>	<b>2.8</b>	<b>3.9</b>
	1999	<b>95</b>	<b>78</b>	<b>169</b>	<b>56</b>	<b>25</b>	<b>34</b>	<b>22</b>	<b>36</b>
	2000	<b>376</b>	<b>148</b>	<b>258</b>	<b>124</b>	<b>92</b>	<b>57</b>	<b>54.8</b>	<b>94</b>
Total DDT	1998	0.02	0.02	0.01	0.03	0.01	0.02	0.01	0.02
	1999	<b>7.7</b>	<b>7</b>	<b>7.5</b>	<b>4.5</b>	<b>2.5</b>	<b>4.6</b>	<b>3.1</b>	<b>4.2</b>
	2000	<b>4.2</b>	<b>3.9</b>	<b>2.7</b>	<b>1.7</b>	<b>1.6</b>	<b>1.5</b>	<b>1.2</b>	<b>1.8</b>
Dieldrin	1998	<b>3.1</b>	<b>5.9</b>	<b>1.9</b>	<b>3.4</b>	<b>1.6</b>	<b>1.9</b>	<b>1.2</b>	<b>1.4</b>
	1999	<b>4.5</b>	0.08	<b>4.7</b>	0.09	0.05	<b>1.4</b>	<b>1.3</b>	<b>1.5</b>
	2000	<b>39.5</b>	<b>18.4</b>	<b>18.1</b>	<b>10.6</b>	<b>24.1</b>	<b>7.42</b>	<b>8.1</b>	<b>9.1</b>
Total PCB	1998	<b>1.59</b>	<b>2.18</b>	0.83	<b>1.22</b>	0.74	0.76	0.57	0.7
	1999	<b>13.7</b>	<b>16.3</b>	<b>13.9</b>	<b>10.1</b>	<b>8.4</b>	<b>6.6</b>	<b>6.5</b>	<b>10.6</b>
	2000	<b>14.7</b>	<b>18.5</b>	<b>17.5</b>	<b>11.0</b>	<b>8.5</b>	<b>6.1</b>	<b>5.7</b>	<b>8.3</b>

Red > 0.5 x ERM (1998 only); **Bold** > reference upper 95% predictive limit (1999 & 2000); \*==recurrent elevated contamination measured in ?2 surveys

## 6.5 CONCLUSIONS

Additional studies or data evaluations, such as ecological risk assessments, are necessary to determine whether statistically elevated contaminant concentrations found in the creeks are biologically meaningful. This is especially important in the case of metals and PAH, where nearly all creek concentrations were statistically elevated compared to reference conditions at two or more stations; however, very few concentrations exceeded one-half the ERM criterion when TOC-normalized. The remaining contaminants, consisting entirely of chlorinated compounds (i.e., pesticides and PCBs), were elevated in relation to reference sediments and half the corresponding ERM value at many stations. DDT was the only chlorinated contaminant that was statistically elevated in both creeks compared to reference sediments but well below the 1998 numeric criterion – a TOC-normalized value proposed by Schwartz et al. (1994) of  $100 \mu\text{g}\cdot\text{g}^{-1}$  organic carbon and accepted by the RWQCB. This discrepancy is due in part to the fact that the TOC-normalized criterion for DDT is generally 2-3 orders of magnitude greater than the corresponding ERM criterion (i.e.,  $46 \text{ ng}\cdot\text{g}^{-1}$  sediment dry weight) when applied to sediments containing 1-3% TOC. TOC-based criteria exist for many sediment contaminants and are in use in other regulatory programs (e.g., Washington Department of Ecology); however, DDT was the only contaminant compared to a numeric criterion originally based on organic carbon instead of sediment dry weight, following the primary method of evaluation used throughout the BPTCP.

## 7.0 BIOACCUMULATION IN CLAMS

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This section presents results from chemical analyses of tissues of the bentnose clam *Macoma nasuta* following 28-day laboratory exposures to surface sediments collected from each of the two creeks and reference area in April 2000. The bentnose clam was chosen as an appropriate test species for reasons presented in Section 3.3.2.4. Additionally, *Macoma nasuta* bioaccumulation tests are in the process of standardization by EPA and will become be the primary benchmark species for near coastal waters. Chemicals of potential concern (COPCs) known to bioaccumulate in the food web were examined, consisting of mercury, PCBs, Chlordane, DDT and Dieldrin. Dry weight tissue results are presented for Islais and Mission Creeks in Appendices A3 and B3, respectively. Each appendix also presents results for reference area tissue data.

Evaluation of chemical bioaccumulation in clams relies primarily on statistical comparisons of individual station results to the upper 95<sup>th</sup> predictive limit (UPL) calculated using lipid-normalized tissue concentrations for the five reference stations tested during the April 2000 survey. This method is similar to that used to identify sediment COPCs in Section 6, except tissue concentrations are compared. Tissue concentrations are associated with corresponding test sediment concentrations through the use of biota-sediment accumulation factors (BSAFs), to evaluate the biomagnification potential of COPCs in the aquatic food web.

It is important to note that there are several different ways to interpret bioaccumulation data, including: 1) an estimate of direct uptake from sediment into the test organism or a proxy; and 2) a predictive measure of transfer across trophic levels. When evaluating direct uptake from sediment, tissue concentrations are compared to various criteria that usually are established on a wet or dry weight basis, and data must be converted if they are reported in dissimilar units. In general, if tissue data are used to estimate the amount of contaminant in the test organism as a food source, wet weight concentrations are used (as they represent the unadjusted concentration per mass of prey). Dry weight tissue concentrations are often used when comparing data across species or studies. When examining transfer between organisms, results are often based on lipid weight, following the assumption that many contaminants concentrate in the fatty tissues of animals. Summary statistics for bioaccumulation data are presented on a dry weight basis, to provide consistency with raw data presented in the appendices as reported by the laboratories, and to compare with dry weight values reported in the literature. Comparisons between creek and reference tissue concentrations and evaluation of BSAFs are on a lipid weight basis following guidance from the EPA (1996). Moisture in clam tissue ranged from 87.5 to 91.6% with a mean of 90.0%; and dry weight lipids ranged from 8.4 to 12.5%, with a mean of 10.2%. Therefore, with respect to COPCs, dry weight tissue concentrations are approximately an order of magnitude (10x) greater than wet weight tissue concentrations; and lipid weight concentrations are approximately 10x greater than dry weight concentrations. There were no differences in results for creek and reference comparisons due to variances in normalization. For example, the same creek tissues were elevated compared to the reference tissue UPL, for dry weight, wet weight and lipid weight data.

## 7.1 OVERVIEW

Chemical concentrations in clam tissues varied considerably for Islais and Mission Creeks, as a function of chemical type, sample location and physical features of the exposure sediment. For all data combined, lipid-normalized tissue concentrations of chlorinated COPCs were strongly correlated with TOC-normalized sediment concentrations and grain size, suggesting that sediment physical features influence the direct uptake of these persistent compounds in biota (Table 7-1).

Mercury, the only metal examined, was not appreciably concentrated in any creek tissues, even though it was elevated in one or more sediment samples in each creek. Bioaccumulation of mercury in the aquatic food chain has been a concern since elevated levels of methylmercury in fish tissue from a highly contaminated Japanese harbor were discovered to have toxic effects in humans in 1956. Elevated fish concentrations and increasing trends in mercury in shellfish have been observed in multi-year monitoring programs conducted in San Francisco Bay, resulting in a 303(d) impaired water body listing.

PCBs and one or more of the chlorinated pesticides were elevated compared to reference tissue concentrations for the two creeks. Concentrations of chlorinated organic compounds in tissues, including PCBs, depend on many organism-related factors including, size, lipid content, trophic level, mode of ingestion, metabolism and diet. PCB concentrations found lethal to fish in laboratory experiments range from 10 to 300  $\mu\text{g}\cdot\text{g}^{-1}$  (or 10,000 to 300,000  $\text{ng}\cdot\text{g}^{-1}$ ) (Rice and O'Keefe 1995). Similar to mercury, elevated PCBs in fish also contributed to the 303(d) impaired water body listing for the bay.

The chlorinated pesticides, Dieldrin, DDT and Chlordane, were elevated in nearly all Mission Creek tissues compared to reference tissue concentrations; however, only tissues exposed to sediments collected from the west end of Islais Creek were elevated. These relict pesticides are all highly fat soluble, with a propensity to bioaccumulate in marine organisms. DDT and its metabolites have been detected in aquatic organisms from every coastal state and from nearly every estuary in the U.S., as well as from many offshore and deep-sea locations. Chlordane is a broad-spectrum poison that affects many organisms. The 1986-1987 Bioaccumulation Study of the U.S. EPA found high levels in fish and shellfish collected from 60 estuaries and coastal marine sites in the U.S., ranging from 6910 to 409,000  $\text{ng}\cdot\text{g}^{-1}$  and 7500 to 42,500  $\text{ng}\cdot\text{g}^{-1}$  wet weight, respectively (Kennish 1997). Results from the National Status and Trends Program from 1984 and 1990 suggest that Dieldrin is less ubiquitous than DDT and Chlordane, but persists on the west coast at high part-per-billion levels, and is commonly found in shellfish at the low part-per-million level (O'Conner and Ehler 1991).

**Table 7-1. Correlation results for COPC concentrations in lipid-normalized tissue vs. normalized sediment – all data combined (n=27).**

	<b>Tissue – lipid normalized</b>				
	<b>Mercury</b>	<b>Total DDT</b>	<b>Dieldrin</b>	<b>Total PCB</b>	<b>Total Chlordane</b>
<b><u>Sediment – percent fines normalized</u></b>					
Correlation coefficient (r)	0.09	<b>0.62</b>	<b>0.80</b>	<b>0.77</b>	<b>0.68</b>
Probability (p)	0.66	<0.001	<0.001	<0.001	<0.001
<b><u>Sediment - TOC normalized</u></b>					
Correlation coefficient (r)	-0.14	<b>0.75</b>	<b>0.94</b>	<b>0.93</b>	<b>0.92</b>
Probability (p)	0.39	<0.001	<0.001	<0.001	<0.001

**bold**=significant positive correlation at  $p < 0.05$ .

## 7.2 REFERENCE AREA

COPC concentrations in tissues exposed to reference area sediments were very low, commensurate with concentrations in marine organisms from pristine coastal environments (Table 7-2). Total Chlordane was below detection limits (i.e.,  $< 0.7 \text{ ng g}^{-1}$  dry weight) in two of the five reference tissues. Only total DDT and total PCBs exceeded  $10 \text{ ng g}^{-1}$  dry weight for the organic contaminants. Mean total DDT and total PCB dry weight concentrations of  $10.3$  and  $27 \text{ ng g}^{-1}$  correspond to wet weight concentrations of approximately  $1.0$  and  $2.7 \text{ ng g}^{-1}$ , respectively, for these tissues. Mercury was extremely low, with the maximum dry weight concentration of  $0.24 \text{ } \mu\text{g g}^{-1}$  recorded for Tubbs Island. Table 7-2 summarizes Reference Area results and includes UPL values on a dry weight basis for COPCs measured in tissues. Actual comparisons between creek and reference tissues were made using dry-lipid weight normalized data; results of these comparisons are shown in Figures 7-1 and 7-2.

Table 7-3 shows BSAFs for reference area data. Although BSAFs provide insight on bioaccumulation in tissues exposed to contaminated sediments, they are less meaningful when chemical concentrations are extremely low in tissues and sediments, such as those reported for the reference area.

**Table 7-2. Reference Area – summary statistics for COPC tissue dry weight concentrations (5 stations)**

COPC	Minimum	Maximum	Maximum Station	Mean	95 <sup>TH</sup> UPL
Mercury ( $\mu\text{g g}^{-1}$ )	0.18	0.24	Tubbs Island	0.20	0.26
<b><u>Chlorinated COPCs (<math>\text{ng g}^{-1}</math>)</u></b>					
Dieldrin	0.8	2.4	North Site	1.3	2.7
Total DDT	2.2	14.1	Paradise	10.3	21.3
Total Chlordane	$<0.7$	3.0	North Site	1.1	4.0
Total PCB	10.5	36.7	Paradise	27.0	51.0

UPL=upper predictive limit

**Table 7-3. Reference Area – summary statistics for biota-sediment accumulation factors (BSAFs).**

COPC	Minimum	Maximum	Maximum Station	Mean	Standard Deviation
Mercury	0.056	0.28	Tubbs Island	0.14	0.08
Dieldrin	0.69	1.33	North Site	1.01	0.25
Total DDT	0.10	2.00	Paradise	0.64	0.77
Total Chlordane	0.17	0.84	North Site	0.42	0.30
Total PCBs	0.42	4.13	Paradise	1.49	1.51

## 7.3 ISLAIS CREEK

Tissues exposed to sediments collected from Stations 1N and 1S accumulated the highest chemical concentrations in Islais Creek. Sediments from these stations were coarser-grained and lower in chemical concentrations relative to the remaining sediments. Stations 1N and 1S had 28.1 and 86.8% sand, respectively, compared with remaining sediments that ranged from 0.9 to 3%. Four chlorinated COPCs (total PCBs, total Chlordane, total DDT, & Dieldrin) that were elevated in sediments were

significantly elevated in clam tissues at Stations 1N and 1S relative to reference tissues. The remaining chlorinated pesticides were not detected at the sub-part-per-billion level in tissue ( $<1 \text{ ng g}^{-1}$ ). Relative differences observed for total Chlordane primarily were due to reference tissue concentrations that were below the detection limit for two stations. Summary statistics for bioaccumulating chemicals measured in tissues are shown in Table 7-4. Mercury, the only metal measured, was detected at sub-part-per-million dry weight concentrations in both creek and reference tissues. Sediment COPC concentrations normalized to percent fines (silt + clay sediment fraction) and TOC strongly correlate to tissue concentrations of the chlorinated COPCs (correlation coefficient [ $r^2$ ] ranging from 0.68 to 0.96), supporting the contention that sediment physical features influence the uptake of these persistent chemicals. Tissue mercury, however, only very weakly associates with percent fines and TOC-normalized sediment mercury ( $r^2 = 0.031$  and  $0.053$ , respectively). Figure 7-1 graphically presents the strongest associations (linear regression) between tissue concentrations and sediment chemical concentrations normalized to either percent fines or TOC.

BSAFs were less than unity (one) for all COPCs, indicating that these chemicals do not readily biomagnify at the bottom of the food web at Islais Creek (Table 7-5).

**Table 7-4. Islais Creek – summary statistics for COPC tissue dry weight concentrations (6 stations).**

COPC	Minimum	Maximum	Maximum Station	Mean	Standard Deviation	Reference Mean
Mercury ( $\mu\text{g g}^{-1}$ )	0.09	0.63	1N	0.24	0.19	0.20
<b><u>Chlorinated COPCs (<math>\text{ng g}^{-1}</math>)</u></b>						
Dieldrin	1.4	8.2	1S	3.8	2.9	1.3
Total DDT	13.6	52.6	1S	26.5	14.3	10.3
Total Chlordane	2.4	29.3	1S	12.9	12.2	1.1
Total PCB	36	248	1S	98	78	27

**Table 7-5. Islais Creek – summary statistics for biota-sediment accumulation factors (BSAFs).**

COPC	Minimum	Maximum	Maximum Station	Mean	Standard Deviation
Mercury	0.07	0.40	1N	0.17	0.12
Dieldrin	0.11	0.78	1S	0.42	0.29
Total DDT	0.20	0.40	1S	0.31	0.07
Total Chlordane	0.05	0.46	1N	0.22	0.15
Total PCBs	0.16	0.39	3N	0.25	0.08

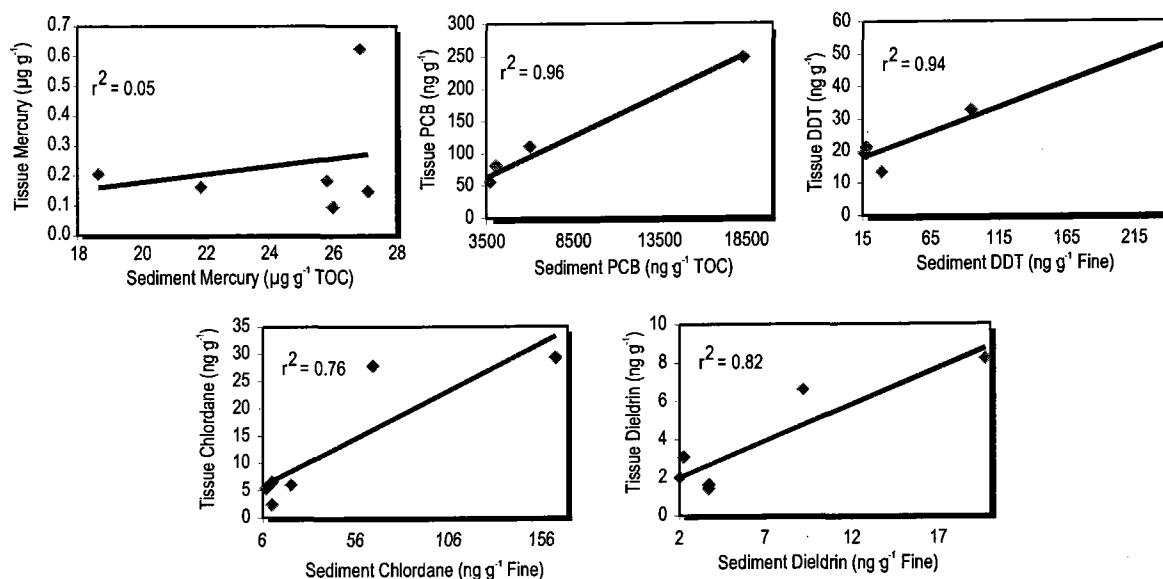


Figure 7-1. *Macoma* tissue concentration compared to sediment concentration normalized to TOC or percent fines. Blue line represents best linear regression.

### 7.3.1 Mercury

None of the tissues, except those exposed to Station 1N sediments, exceeded the lipid-normalized reference UPL for mercury of  $5.15 \text{ } \mu\text{g g}^{-1}$  lipid (Figure 7-2).

Average dry weight concentrations of mercury in creek tissues at  $0.24 \text{ } \mu\text{g g}^{-1}$  were comparable to both in-bay reference tissues and to average concentrations recorded in the related bivalve species *Mytilus* (i.e.,  $0.24 \pm 0.08 \text{ } \mu\text{g g}^{-1}$ ), collected from relatively clean sediments from U.S. and Canadian waters (Fowler 1990). Only tissue exposed to Station 1N sediment had a significantly higher concentration at  $0.63 \text{ } \mu\text{g g}^{-1}$  dry weight. Sediment collected from Station 1N had the second highest mercury concentration ( $1.19 \text{ } \mu\text{g g}^{-1}$ ) and the highest TOC concentration (4.43%) measured in the April 2000 survey. However, this station also had a BSAF of 0.40, indicating that mercury in sediment was not readily bioavailable to exposed clams. Remaining BSAFs were even lower, averaging 0.12. The average BSAF for all stations was 0.17 (Table 7-5).

There was no significant correlation between tissue and sediment mercury concentrations for Islais Creek; however, this is not unexpected as all concentrations were low and fairly uniform, except those observed for Station 1N (tissue & sediment) and Station 1S (sediment only).

### 7.3.2 Polychlorinated Biphenyls (PCBs)

Total PCB concentrations in creek tissue ranged from 36 to  $248 \text{ ng g}^{-1}$  (Table 7-4), with an average concentration of  $98 \text{ ng g}^{-1}$  (dry weight). Tissue concentrations for five out of six stations exceeded the

lipid-normalized reference UPL of 858 ng $\text{g}^{-1}$  lipid (or 51.0 ng $\text{g}^{-1}$  dry weight) (Figure 7-2), with the highest levels observed at Station 1S. Although elevated compared to in-bay reference tissues, creek tissue concentrations were significantly lower than many concentrations reported for bivalves collected from other populated shorelines (Table 7-6). Total PCB concentrations in tissues correlated well with TOC normalized sediment total PCB concentrations (Figure 7-1); however, the mean BSAF of 0.25 indicates that these contaminants are not biomagnifying.

The relative abundances of individual congeners in tissue samples generally were consistent with abundances in the sediments. The lower and higher chlorinated congeners typically were undetected, or present at low part-per-billion concentrations, whereas, the pentachloro- and hexachloro-biphenyls (especially congener numbers 101, 118, 138, and 158) were detected the most frequently. The two more toxic coplanar congeners measured, PCB 77 and PCB 126 (3,3',4,4'-tetrachlorobiphenyl and 3,3',4,4',5-pentachlorobiphenyl, respectively), were not detected in any Islais Creek tissues.

**Table 7-6. Ranges of PCB and DDT concentrations (ng $\text{g}^{-1}$  dry weight) in the mussel *Mytilus* and closely related species (adapted from Fowler 1990).**

Study Area	Total PCBs		Total DDT	
	Minimum	Maximum	Minimum	Maximum
Baltic Sea	179	778	62	739
North Sea	106	362	15	143
Irish Sea	57	1070	92	590
English Channel	380	480	35	112
US Northwest Atlantic	10	6808	2.8	1109
US Pacific Coast	607	2052	5.4	1077
Northeast Atlantic (France)	96	1345	-	-
Mediterranean (Spain)	10.8	1264	60	288

### 7.3.3 Chlorinated Pesticides

**DDT.** Tissues exposed to sediments from Stations 1N and 1S and marginally 3S were statistically elevated compared to the lipid-normalized reference UPL of 345.7 ng $\text{g}^{-1}$  lipid (Figure 7-2). Total DDT (sum of 2,4'- and 4,4'- DDT, DDE, and DDD isomers) dry weight concentrations ranged from 13.6 to 52.6 ng $\text{g}^{-1}$ , averaging 26.5 ng $\text{g}^{-1}$  in creek tissues. Maximum dry weight concentrations reported for related species exposed to sediment from other populated areas reach concentrations 10 to 50 times higher than the maximum Islais Creek concentration (Table 7-6). Total DDT in tissue associated more strongly with sediment DDT normalized to percent fines ( $r^2 = 0.94$ ) than it did with sediment normalized to total organic carbon ( $r^2 = 0.81$ ). The BSAF was well below unity, indicating that this nationally ubiquitous contaminant likely does not biomagnify in the local food web.

In general, the breakdown products of DDT (i.e., DDD and 4,4'-DDE) displayed the highest concentrations in tissues, consistent with DDT distribution patterns in sediment.



**Chlordane.** All creek tissues, except those exposed to Station 3S sediments, exceeded the lipid-normalized reference UPL of  $71.5 \text{ ng}\cdot\text{g}^{-1}$  lipid; however, three stations (2N, 2S & 3S) only slightly exceeded this threshold (Figure 7-2).

Total Chlordane concentrations again were highest in clam tissues exposed to the coarser-grained sediments from Stations 1N and 1S. Dry weight concentrations for all tissues ranged from 2.4 to  $29.3 \text{ ng}\cdot\text{g}^{-1}$ , averaging  $12.9 \text{ ng}\cdot\text{g}^{-1}$ . Tissues exposed to sediments collected east of Transect 1 were similar to reference levels, averaging  $5.1 \text{ ng}\cdot\text{g}^{-1}$ . Alpha-, cis-, and trans-Chlordane were the dominant forms measured. Heptachlor and Heptachlor epoxide were not detected in any tissues.

**Dieldrin.** Only tissues exposed to Stations 1N and 1S sediments were significantly elevated compared with the lipid-normalized reference UPL of  $49.6 \text{ ng}\cdot\text{g}^{-1}$  lipid.

Dieldrin dry weight concentrations in creek tissues ranged from 1.4 to  $8.2 \text{ ng}\cdot\text{g}^{-1}$  (mean= $3.8 \text{ ng}\cdot\text{g}^{-1}$ ), posting the closest values to reference tissues (mean= $1.3 \text{ ng}\cdot\text{g}^{-1}$ ) of the chlorinated compounds measured.

Tissue concentrations of Dieldrin more strongly associated with sediment Dieldrin concentrations normalized to percent fines compared to TOC-normalized sediments (Figure 7-1). The BSAF for Dieldrin was highest (mean=0.42) of the chlorinated compounds measured in tissue (Table 7-5); however, all values were less than unity (1.0) indicating a low biomagnification potential for sediment-dwelling organisms exposed to Dieldrin at Islais Creek.

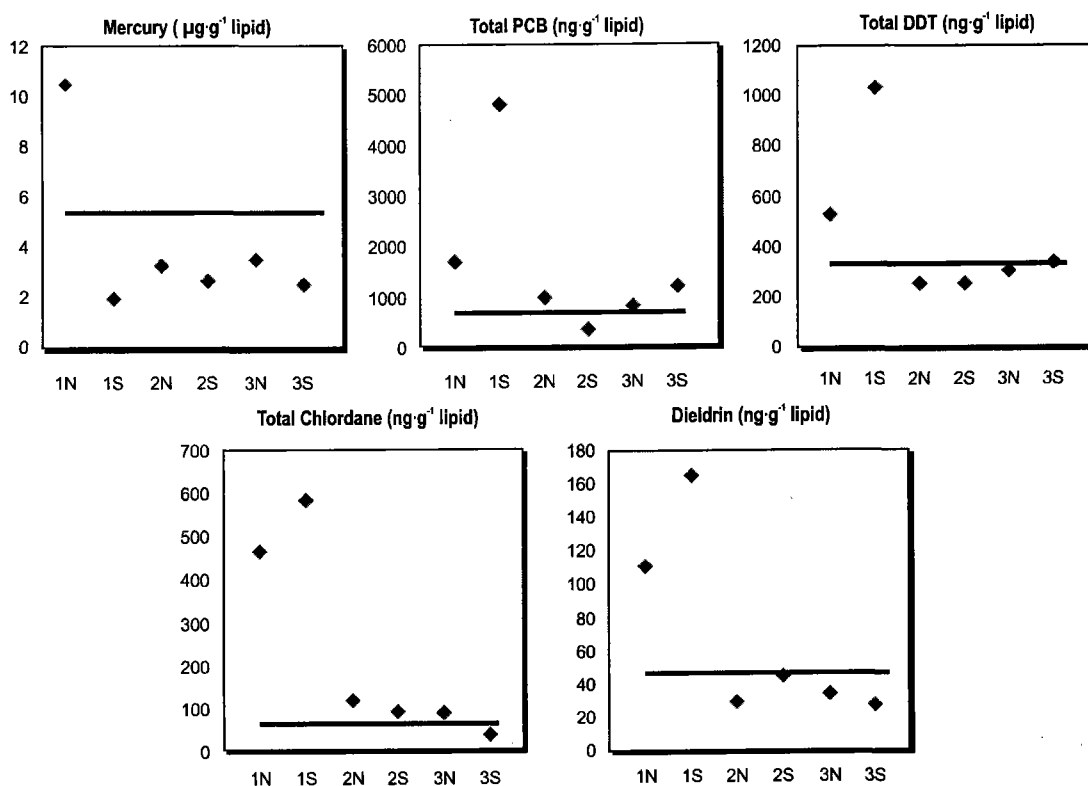


Figure 7-2. Islais Creek tissue concentrations compared to the upper 95<sup>th</sup> predictive limit (blue line) established for tissues exposed to reference area sediments (all results are lipid-normalized).

## 7.4 MISSION CREEK

In general, COPCs, excluding mercury, were elevated in Mission Creek tissues compared to tissues exposed to reference area sediments. Mercury, the only metal measured, was detected at sub-part-per-million dry weight concentrations in both creek and reference tissues. Tissues exposed to sediments collected west of Transect 4, except Station 1S, accumulated the highest concentrations (Figure 7-3). Corresponding sediments from this area were fairly coarse-grained, ranging from 10.8 to 71.2% sand. The highest tissue concentrations were consistently recorded for Station 1N, which also had the highest concentration of sandy sediment. Two chlorinated pesticides (total Chlordane and total DDT) and total PCBs that were elevated in sediments were significantly elevated in creek tissues relative to reference tissues for nearly all stations. Dieldrin was slightly elevated in tissues exposed to creek-end stations only. The remaining pesticides were not detected at the sub-part-per-billion level in tissue ( $<1 \text{ ng g}^{-1}$ ). Relative exceedances observed for total Chlordane and Dieldrin in creek tissues were due primarily to non-detect or near-detection limit values for two or more reference tissue samples. Summary statistics for bioaccumulating chemicals measured in tissues are shown in Table 7-7.

When normalized to either percent fines or TOC, only sediment Chlordane and Dieldrin exhibited strong significant correlations with dry weight tissue concentrations (Figure 7-4). BSAFs were less than unity (one) for all COPCs, indicating that these chemicals do not readily biomagnify at the bottom of the food web (Table 7-8). The highest BSAFs were recorded at stations comprised of relatively sandy sediments that were generally less contaminated than the finer-grained, TOC-enriched sediments in the creek.

**Table 7-7. Mission Creek – summary statistics for COPC dry weight tissue concentrations (6 stations).**

COPC	Minimum	Maximum	Maximum Station	Mean	Standard Deviation	Reference Mean
Mercury ( $\mu\text{g g}^{-1}$ )	0.14	0.26	4S	0.19	0.05	0.20
<b><u>Chlorinated COPCs (<math>\text{ng g}^{-1}</math>)</u></b>						
Dieldrin	2.0	9.0	1N	5.1	2.5	1.3
Total DDT	23.5	52.1	2S	36.1	11.6	10.3
Total Chlordane	12.0	62.8	1N	29.1	20.0	1.1
Total PCB	94	225	1N	138	44	27

Table 7-8. Mission Creek – summary statistics for biota-sediment accumulation factors (BSAFs).

Analyte	Minimum	Maximum	Mean	Standard Deviation	Maximum Station	Reference 95 <sup>th</sup> UPL
Mercury	0.02	0.10	0.05	0.03	3S	0.46
Total PCBs	0.06	0.31	0.18	0.09	4N	1.54
Total Chlordane	0.05	0.37	0.19	0.11	2S	0.64
Total DDT	0.06	0.37	0.23	0.10	2S	0.86
Dieldrin	0.12	0.60	0.28	0.14	2S	1.68

#### 7.4.1 Mercury

None of the tissues exposed to Mission Creek sediments exceeded the lipid-normalized reference UPL of  $5.15 \text{ } \mu\text{g} \cdot \text{g}^{-1}$  lipid (or  $0.26 \text{ } \mu\text{g} \cdot \text{g}^{-1}$  dry weight) (Figure 7-3).

Average dry weight concentrations of mercury in creek tissues (mean =  $0.19 \text{ } \mu\text{g} \cdot \text{g}^{-1}$ ) were comparable to both in-bay reference tissues and to average concentrations recorded in the related bivalve species *Mytilus* (i.e.,  $0.24 \pm 0.08 \text{ } \mu\text{g} \cdot \text{g}^{-1}$ ), collected from relatively clean sediments from U.S. and Canadian waters (Fowler 1990). The extremely low BSAF mean value of 0.05 provides further confirmation that mercury does not bioaccumulate in benthic organisms exposed to Mission Creek sediments (Table 7-8).

#### 7.4.2 Polychlorinated Biphenyls (PCBs)

Total PCB concentrations in creek tissue ranged from 94 to  $225 \text{ ng} \cdot \text{g}^{-1}$  (Table 7-7), with an average concentration of  $138 \text{ ng} \cdot \text{g}^{-1}$  (dry weight). Tissue concentrations exceeded the lipid-normalized reference UPL of  $858 \text{ ng} \cdot \text{g}^{-1}$  lipid (or  $51.0 \text{ ng} \cdot \text{g}^{-1}$  dry weight) (Figure 7-3) at all stations, with the greatest exceedance observed at Station 1N.

The relative abundances of individual congeners in tissue samples generally were consistent with abundances in the sediments. The lower and higher chlorinated congeners typically were undetected, or present at low part-per-billion concentrations, whereas, the pentachloro- and hexachloro-biphenyls (especially congener numbers 101, 118, 138, and 158) were detected the most frequently. The two more toxic coplanar PCB congeners measured, PCB 77 and 126 (or 3,3',4,4'-tetrachlorobiphenyl and 3,3',4,4',5-pentachlorobiphenyl, respectively), were not detected in any tissues.

#### 7.4.2 Chlorinated Pesticides

**DDT.** Tissues exposed to sediments from all stations, except 1S, were statistically elevated compared to the lipid-normalized reference UPL of  $345.7 \text{ ng} \cdot \text{g}^{-1}$  (or  $21.3 \text{ ng} \cdot \text{g}^{-1}$  dry weight) (Figure 7-3). Total DDT (sum of 2,4'- and 4,4'- DDT, DDE, and DDD isomers) dry weight concentrations ranged from 23.5 to  $52.1 \text{ ng} \cdot \text{g}^{-1}$ , averaging  $36.1 \text{ ng} \cdot \text{g}^{-1}$  in creek tissues. Maximum dry weight concentrations reported for related species exposed to sediment from other populated areas often reach concentrations 10 to 50 times higher than the maximum Mission Creek concentration (Table 7-6).

In general, the breakdown products of DDT displayed the highest concentrations in tissues, consistent with DDT distribution patterns in sediment.

**Chlordane.** All creek tissues exceeded the lipid-normalized reference UPL of  $71.5 \text{ ng g}^{-1} \text{ lipid}$  (Figure 7-2), corresponding to an extremely low reference dry weight concentration of  $4.0 \text{ ng g}^{-1}$ . Total Chlordane concentrations again were highest in tissues exposed to coarser-grained sediments from Stations 1N and 2S located at the creek end. Dry weight concentrations ranged from  $12.0$  to  $62.8 \text{ ng g}^{-1}$ , averaging  $29.1 \text{ ng g}^{-1}$ . The BSAF for total Chlordane was extremely low (mean=0.19), indicating a low biomagnification potential for Chlordane in creek biota.

**Dieldrin.** All tissues except for those exposed to sediment from Stations 3S and 4S were significantly elevated compared with the lipid-normalized reference UPL of  $49.6 \text{ ng g}^{-1} \text{ lipid}$ . This value corresponds to an extremely low reference dry weight concentration of  $2.7 \text{ ng g}^{-1}$ .

Dieldrin dry weight concentrations in creek tissues were low, ranging from  $2.0$  to  $9.0 \text{ ng g}^{-1}$  (mean= $5.1 \text{ ng g}^{-1}$ ). Dieldrin in tissues strongly associated Dieldrin measured in fine particle normalized sediments (Figure 7-4). The BSAF for Dieldrin was extremely low (mean=0.28), indicating a low biomagnification potential for Dieldrin in creek biota.

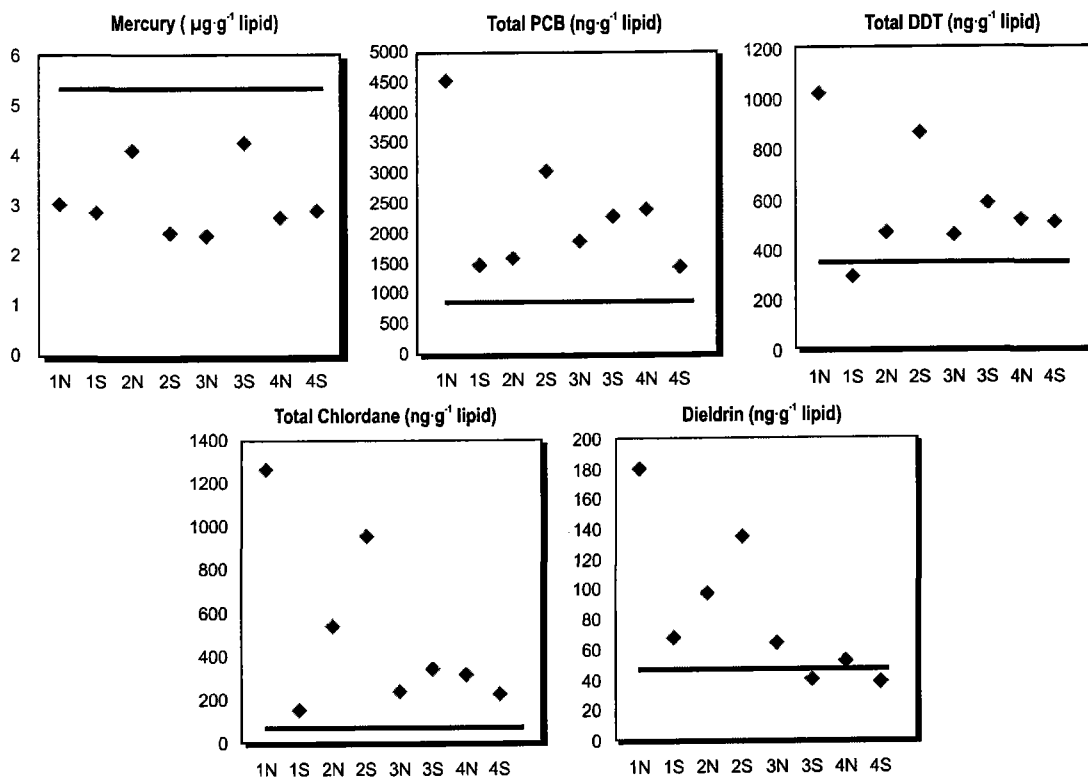
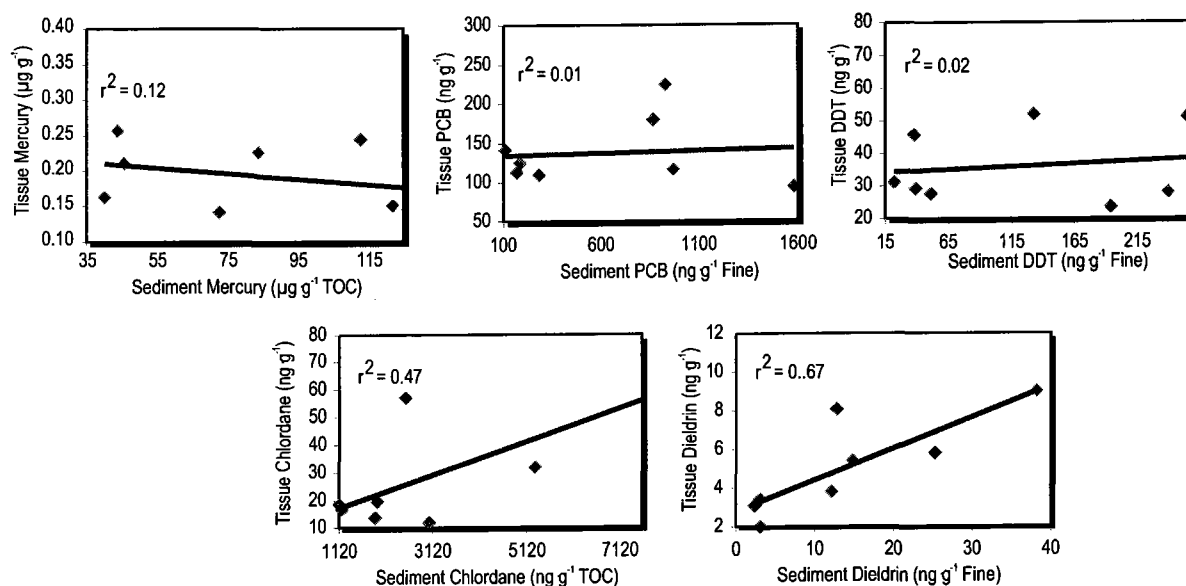


Figure 7-3. Mission Creek tissue concentrations compared to the upper 95<sup>th</sup> predictive limit (blue line) established for tissues exposed to reference area sediments (all results are lipid-normalized).



**Figure 7-4. *Macoma* tissue concentration compared to sediment concentration normalized to TOC or percent fines. Blue line represents strongest linear regression.**

## 7.4 CONCLUSIONS

Chlorinated pesticides and PCBs were statistically elevated in tissues exposed to creek end sediments compared to tissues exposed to in-bay reference sediments. However, sediment-tissue bioaccumulation factors were significantly less than unity (one) for all samples, indicating that these chemicals may not biomagnify through the local food web. Mercury tissue concentrations were lower than reference tissue concentrations for all creek stations except 1N, located at the west end of Islais Creek. The limited area of impact at the creek end (< 1 acre), coupled with strong evidence that contaminant concentrations are decreasing and have minimal biomagnification potential, make Islais Creek an ideal candidate for natural recovery or localized remediation.

Similar to Islais Creek, tissues exposed to west end Mission Creek sediments displayed statistically elevated concentrations of chlorinated compounds compared to reference tissues. However, biota-sediment accumulation factors were far less than one, again indicating a low biomagnification potential for these contaminants in the local food web.

## 8.0 SOURCE IDENTIFICATION OF SELECTED COPCs

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The section explores sources of selected chemicals of potential concern (COPCs), identified in Section 6 for Islais Creek, the only study area that had any sediment with recurrent toxicity and elevated chemistry (see Section 9). Only those COPCs that had physical or chemical distribution patterns suggesting contaminant sources are evaluated – certain metals and PAHs. Methods used to identify potential physical and/or chemical sources are discussed in Section 8-1. Physical source refers to point locations of contaminant entry to the creeks that are linked to past or present activities; chemical source refers to the type of contamination, which can be inferred from a chemical signature (e.g., PAH from creosote) or relationship (e.g., correlations between metals). Chemical source identification was not performed for chlorinated pesticides or PCBs. Pesticides are largely comprised of single compounds or limited compound mixtures, and are therefore, not conducive to forensic chemical techniques. PCBs originate from Aroclors™, a limited suite of seven man-made ubiquitous mixtures that both complicate and reduce the importance of source identification.

Historical sources of COPCs to creek sediments are discussed in Section 2. These sources are of interest for several reasons: 1) to see if sources have changed over time; 2) to see if variations in contaminant distributions in sediment are related to different source terms; and 3) to see if source identification can be used to support future preventative measures or remedial alternatives analysis.

### 8.1 OVERVIEW OF SOURCE IDENTIFICATION METHODS

#### 8.1.1 Metals

Concentrations of trace metals in Islais Creek sediments varied considerably as a function of location, sediment type and contaminant loading. In some cases, concentrations were 10 to >30 times greater than typical background concentrations (see Section 6, Table 6-1). To better understand differences in metal concentrations that may result from variations in grain size and mineralogy and to identify creek locations where metal content may be influenced by anthropogenic inputs, metal concentrations were normalized to (divided by) iron and aluminum concentrations. Aluminum and iron are major constituents of sediment minerals and usually exhibit a positive relationship with trace metals. Aluminum is mostly present as a structural component of aluminosilicate minerals, whereas iron may occur as a structural component of aluminosilicates as well as an oxide coating on mineral grains. In general, when concentrations of aluminum or iron are higher in a sediment sample, concentrations of trace metals also are higher naturally. Lower concentrations of aluminum, iron, and metals are found for sediments composed primarily of quartz sand or shell carbonates, whereas higher values of a metal are common to more clay-rich, fine-grained sediments, such those found throughout much of Islais Creek. Thus, plots of metals versus aluminum or iron from a given area with little or no pollutant inputs often show a strong linear relationship. Positive deviations from this linear trend of a metal versus aluminum or iron help identify anthropogenic inputs of that metal to the sediment. The iron versus aluminum relationship for sediments sampled in 1998 in Islais Creek was strong and statistically significant

( $r^2=0.87$ ;  $p<0.001$ ). Thus, either aluminum or iron is appropriate to normalize trace metal concentrations (i.e., remove natural variability). The normalized metal data can then be used to identify sediment locations with anthropogenic inputs of metals as well as to target sources for these inputs through the subsequent identification of transport pathways. However, these data cannot be used to identify elevated metals associated with natural deviations in mineralogy, since *a priori* assumptions exclude metal concentrations that are significantly elevated from background from the initial analysis.

### 8.1.2 Polycyclic aromatic hydrocarbons (PAH)

Source identification of petroleum hydrocarbons was performed using sediment results from the analysis of total petroleum hydrocarbons (TPH) and polycyclic aromatic hydrocarbons (PAH). This analysis was limited to data collected in 1998, the only year in which TPH was analyzed and samples were collected throughout the creek. A total of eight TPH and 41 PAH "compounds" were analyzed using laboratory methods described in Section 3. These results were used to support chemical forensic methods to identify potential hydrocarbon sources.

PAH in surface and subsurface sediments were first evaluated using principal component analysis (PCA). This multivariate statistical technique has been used to identify petroleum source in marine sediments (Kennicut et al. 1994; Maxon et al. 1997) and is not detailed here. PCA was used to provide insight to potential PAH sources for Islais Creek sediments. It is a useful technique as it removes investigator bias, and can evaluate large data sets with multiple analytes simultaneously. Log transformed concentrations of PAH, total petroleum hydrocarbons (8 range classes) and five linear alkylbenzenes (LAB) were analyzed together in the PCA. LABs were analyzed in 1998 only, as potential tracers of contamination associated with sewage discharges. For this investigation, PCA was used only as an exploratory tool to reveal sample relationships and to support other forensic methods. Other interpretive tools such as GC/FID chromatogram pattern recognition (Douglas, et al. 1992) and/or source ratio analysis (Douglas et al. 1996) also were used to identify potential hydrocarbon sources.

## 8.2 METALS

Sediment concentrations of COPC metals, lead and zinc, were most elevated at the west end of the creek (see Section 6, Table 6-5). Overall, the highest concentrations were found at Stations 1N, 1C (sampled only in 1998) and 1S. These elevated metal concentrations were distinguished by large positive deviations from the natural metal/aluminum relationship (solid line) as shown in Figure 8-1. Each of the anomalous data points suggests an anthropogenic input of lead or zinc at the specified station. Anthropogenic inputs of lead and zinc were greatest at Transect 1, somewhat less at Transects 2 and 3, and not identifiable at Transects 4, 5 and 6 (Figure 8-1; see Figure 3-1, Section 3 for station map).

To help identify metal sources and transport pathways, actual concentrations of anthropogenic lead and zinc were determined by subtracting natural concentrations (based on the solid line on each graph) from total metal levels. The solid line was generated through linear regression of unelevated metal creek concentrations with corresponding aluminum concentrations. A strong linear relationship was found for anthropogenic zinc versus anthropogenic lead with an overall ratio of the two components of  $1.5 \pm 0.4$

(Figure 8-1). This strong linear relationship supports either a common source or a common partitioning mechanism (e.g., organic carbon adsorption) for the elevated sediment concentrations of lead and zinc. Based on higher concentrations at the creek end (Transect 1), and decreased concentrations at Transects 2 and 3, inputs of both metals can be traced to the CSO Weir and/or runoff from the Interstate 280 overpass. Interstate 280 is a likely co-contributor, as considerable storm water runoff from the overpass to the creek end was observed during the two wet-weather surveys. Additionally, the highest concentrations of COPC metals were observed in sediments directly below Interstate 280 and not at either end of the CSO Weir (Transects 2 and 3).

Sediments with the highest concentrations of COPC metals also had the highest concentrations of TOC (Figure 8-2). These results confound identification of metal sources because TOC adsorbs and concentrates metals (including dissolved metals from the water column). Therefore, elevated metals observed in the TOC-enriched sediments at the creek end may have been transported by a nearby source with the TOC, or they could have been partitioned from the water column into TOC accumulated at the creek end. Both processes most likely contribute to elevated metals concentrations observed at the end of the creek.



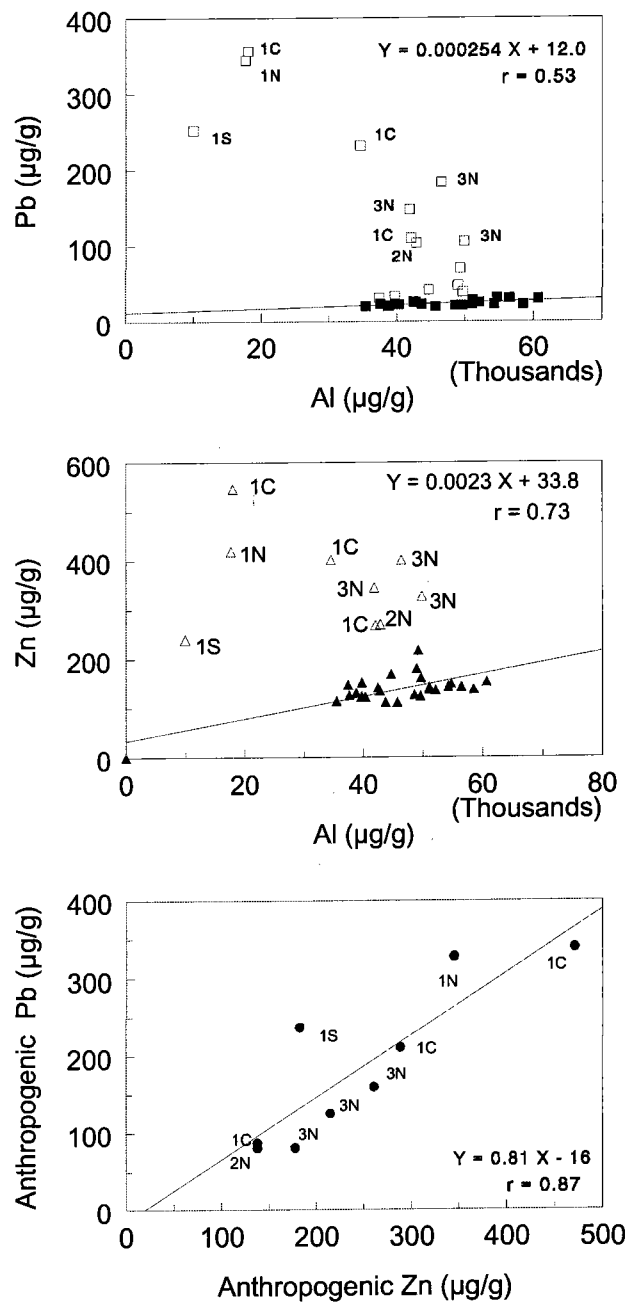


Figure 8-1. Concentrations of lead, zinc and mercury vs. aluminum, and elevated zinc vs. elevated lead in Islais Creek surface sediments.

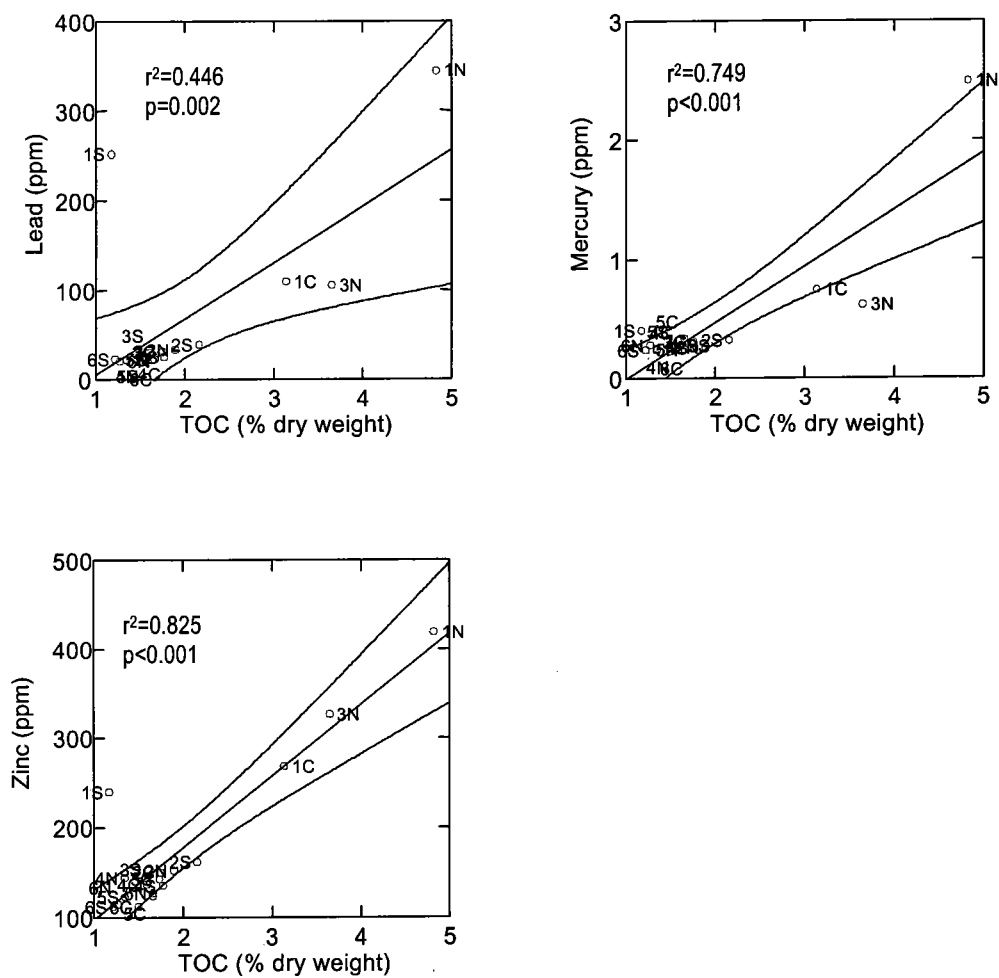


Figure 8-2. Lead, mercury and zinc concentrations vs. TOC in Islais Creek surface sediments.

## 8.3 HYDROCARBONS (PAH, TPH & LAB)

### 8.3.1 Results from Principal Components Analysis.

Principal components analysis was performed on 1998 samples only, including all creek surface and subsurface cores and the single reference site sampled at Paradise Cove. Results shown in Figure 8-3 indicate several likely sources of hydrocarbons to creek sediments, namely petroleum-based oils (e.g., lube oils) at Transect 1 and common combustion-related (pyrogenic) compounds associated with urban runoff and aerial fallout at all other stations, including the in-bay reference site at Paradise Cove. Minor inputs from either creosote or coal tar are seen in sediments located near the creek mouth (Transects 5 & 6); however, total concentrations at these stations were very low – close to reference concentrations of  $<2 \text{ ng g}^{-1}$  (ppm) total PAH. Figure 8-3 is a plot of the first two principal component vector scores (i.e., PCA1 and PCA2), which accounted for 65% of the total data variance. Figure 8-4 plots the sample "loadings" for the same two vectors. The scores describe how sediment samples are related and the loadings explain why the samples are similar or dissimilar. The farther the loadings are from the origin, the larger the impact of that compound on the variance. It is also possible that one compound class will drive the separation (e.g., combustion PAH); however the sample may also contain compounds that are not described in the loadings. For example, the sample set may be driven by combustion PAH versus PAH compounds associated with heavy oil. Although the sample relationship is driven by these two loadings, the heavy oil samples may also contain combustion PAH (e.g., phenanthrene, fluoranthene, and pyrene) and it is the lack of heavy oil in the samples with combustion PAH that separates the samples.

Principal component 2 (PCA 2), plotted on the y-axis of Figure 8-3, identifies additional information on chemical differences in the sample set, most importantly the source type of the combustion and petroleum inputs. The distribution of samples impacted by combustion sources along the y-axis is driven primarily by PAH sample differences due to atmospheric deposition or urban runoff (higher concentrations of benzo(b)fluoranthene [BBF]) versus coal tar or creosote (higher in fluoranthene [F], phenanthrene [P], pyrene [PY]). Weathering or biodegradation of lighter more labile PAH (e.g., 2-4 ring PAH) relative to the more refractory PAH (e.g., benzo[b]fluoranthene) could result in the relative enrichment of high molecular weight (HMW) PAH. Samples that fall between the extremes are likely due to mixing of these two sources. Additional samples from potential contaminant sources to Islais Creek are required to further delineate HMW PAH sources; however, this effort is not recommended as total HMW PAH concentrations were not significantly elevated in creek sediments (e.g., HMW is not a COPC).

The results of the PCA analysis clearly indicate multiple sources of pyrogenic and petrogenic related contamination to Islais Creek. Petroleum sources dominate the distribution in the upper creek with hydrocarbons characteristic of lubricating oils present. Combustion products dominate contamination in the middle and lower creek with hydrocarbon inputs from atmospheric deposition, and coal tar or creosote sources.

In Figure 8-3, the difference between pyrogenic (combustion related) and petrogenic (petroleum origin) hydrocarbons in the sample set drive PCA1, graphed on the x-axis. Stations names coupled with 0, 1 or

2 indicate PCA results for that station at the surface, 0-1 ft or 1-2 ft, respectively. Samples from Transect 1 (surface and core samples) plot on the right side of the graph due to heavy petroleum in the samples (e.g., Station 1C, depth 1-2 ft. shown as 1C2). Transect 3 and 4 stations plot toward the center of the graph because they have inputs from combustion-related and petroleum-related hydrocarbons. Even these samples separate across the x-axis depending on the relative proportion of hydrocarbons. For example, the surface sample at Station 3C (3C0), which is dominated by pyrogenic PAH, plots further to the left than the 1-2 ft subsurface sample from Station 3S (3S2), which is a mixture of petrogenic and pyrogenic sources. Subsurface sample 3S2 provides an excellent example of containing primarily combustion-related hydrocarbons with minimal input from oil. Sediments from transects 4, 5 and 6 have PAH distribution patterns and concentrations similar to background sediments from impacts from atmospheric fallout. These samples cluster on the plot with the Paradise Cove reference sample in the upper right quadrant.

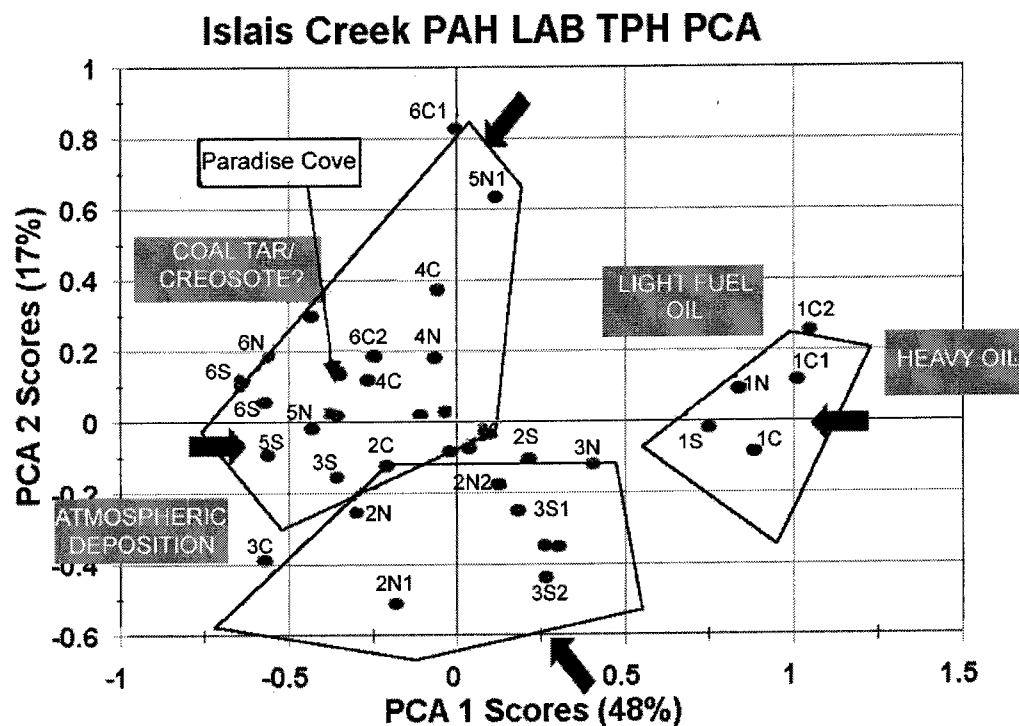
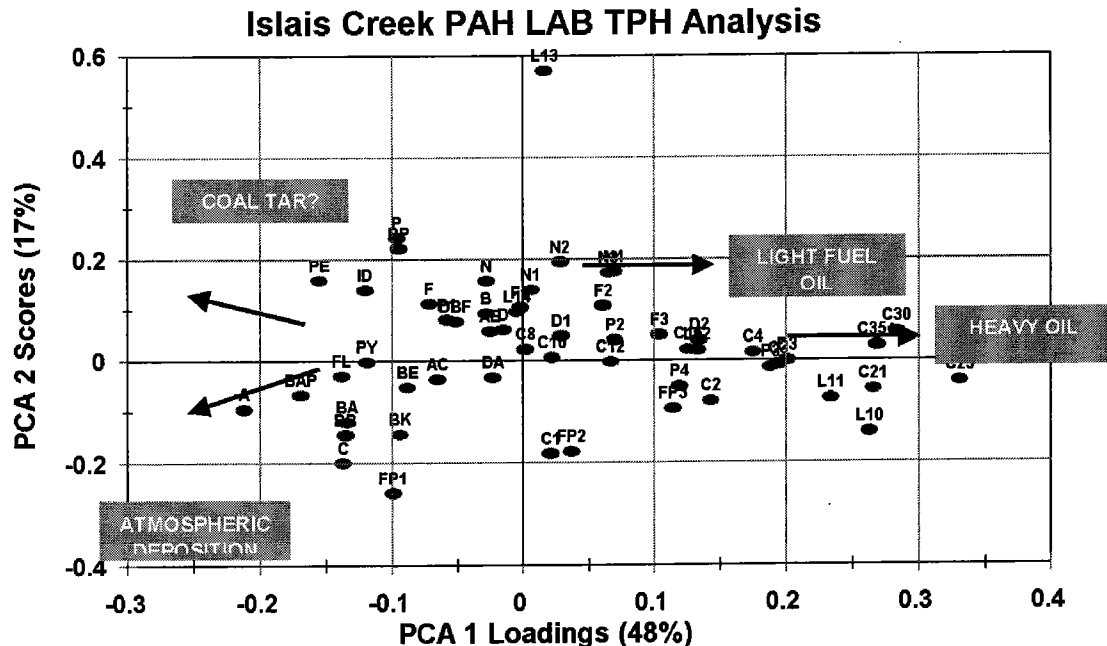


Figure 8-3. Plot of PCA Vector 1 vs. PCA Vector 2 scores showing different PAH sources between creek end (1), mid-section (2 & 3), and creek mouth (5 & 6) and Paradise Cove sediments.



**Figure 8-4. Plot of CA loadings showing individual PAH compounds (abbreviated) associated with different sources.**

As noted in Section 8.1.2, PCA was utilized in an attempt to determine those PAH compounds that were responsible for the greatest differences observed between creek sediments. From this screening analysis, a number of PAH compounds were identified for further delineation of potential hydrocarbon sources to Islais Creek. Sediment samples were compared, using a series of double ratio plots, with a diverse grouping of reference standards, including petrogenic (i.e., oil derived), pyrogenic (manufactured gas residues, wastes and distillates), pyrogenic dominated reference sediments (urban runoff reference standards), natural background reference sediments and in-bay reference sediments to determine whether any creek PAHs were associated with specific contaminant types.

### 8.3.2 Diagnostic Ratios

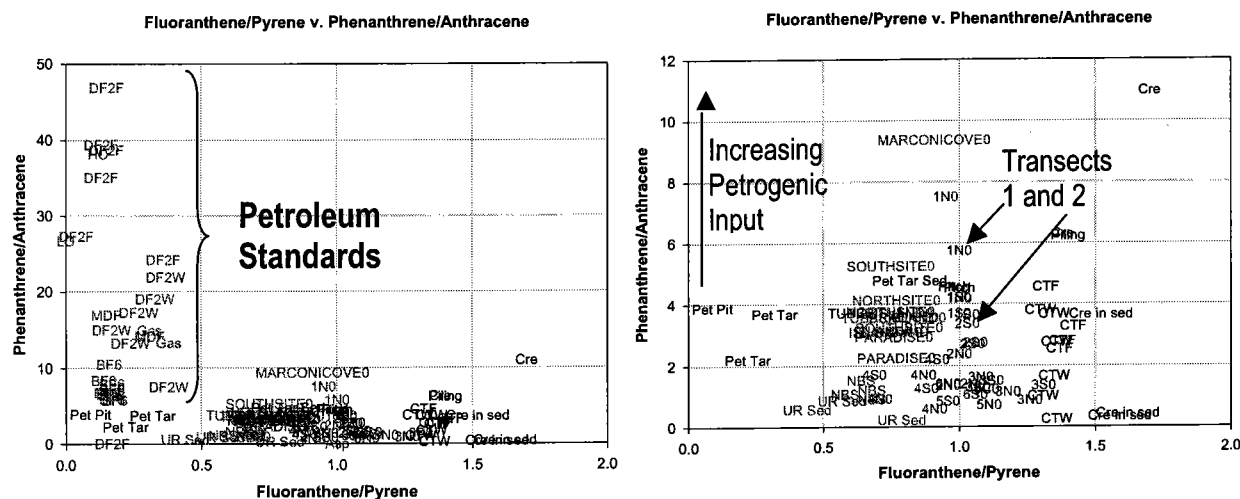
Previous investigators have used C4-phenanthrenes in double ratio plots to identify co-occurring PAH-like compounds present from natural sources. An aromatic diterpane, retene, is commonly found within the C4-phenanthrene isomer pattern. Retene is derived from specific plant resins, and is commonly found in west coast sediments. Many of the sediment samples contained elevated C4-phenanthrenes, and while the extracted ion profiles were not available to review, it is assumed that the elevated C4-phenanthrene patterns seen in the sediments are partially due to retene.

Other PAH compounds such as benzo[*b*]fluoranthene, fluoranthene, pyrene, phenanthrene and anthracene also were useful in identifying various pyrogenic (via fossil fuel combustion) and pyrogenic dominated (via natural processes) input to sediments.

Fluoranthene/pyrene versus phenanthrene/anthracene are plotted in the double ratio plots shown in Figure 8-5. These ratios, when used together are effective in discerning petrogenic versus pyrogenic signatures. In the plot on the left, a suite of petroleum standards (samples in PINK) are plotted with the field samples (RED), pyrogenic (via fuel combustion processes - BLUE), pyrogenic dominated (via natural processes - GREEN) and in-bay reference sediments (GREEN). It is clear from this plot that there is little correlation (if any) to specific petroleum references in most of the sediments in Islais Creek. Transects 1 and 2 (and the Marconi Cove reference site) appear to contain more petrogenic input (plot on right, which is the same plot as on the left, with the petroleum standards removed) than the other sediments in the creek, as evidenced by the elevated phenanthrene/anthracene ratio, which is suggestive of petroleum input. Additionally, the field samples do not appear to have a significant input from any manufactured gas plant (MGP) residue or waste.

As noted above, most of the sediments contained elevated C4-phenanthrene concentrations, likely due to input from a naturally occurring biomarker compound, retene. Another compound that was present at elevated levels in most sediments was benzo[b]fluoranthene. When these two diagnostic compounds are plotted (as percent of total PAH), the pyrogenic signature is revealed. It is apparent that the petroleum (PINK) and coal derived standards (BLUE) that are flush against the Y axis (thus containing small percentages of C4-Phenanthrene relative to total PAH) are unrelated to the field samples, reference sediments (GREEN), Pet Tar (petroleum tars - BLUE) and Cre in sed (Creosote in sediment from Eagle Harbor, Washington - BLUE). This plot illustrates again that there are no specific petroleum or MGP residue/waste/distillate that are uniquely responsible for the contamination present in the sediments.

Based on evidence of retene and the results of double ratio plots using diagnostic PAH compounds, it appears that there is a single, overwhelming 'source' contaminant signature present in Islais Creek sediments, urban runoff. Since urban runoff is not a specific fuel or waste type, a brief description of its general makeup is presented below.



**Figure 8-5. Diagnostic ratio plots showing increased petrogenic (combustion) inputs at the creek end (Transects 1 and 2).**

Urban sediments can receive PAH derived from both ‘point’ and ‘non-point’ sources. Among the latter, urban background (derived from both runoff and atmospheric fallout) is considered ubiquitous in most urban water bodies. Numerous studies of PAH in urban runoff and atmospheric particulates have been conducted around the U.S. over the last two decades. Although ‘non-point’ sources of PAH in urban environmental vary, the most common sources are 1) urban dust containing combustion-related PAH (principally arising from internal combustion engines, especially diesel-based [e.g., Harrison et al. 1996]); 2) street runoff containing traces of lubricating oils (principally arising from releases from automobiles); and 3) illegal or unintentional discharging of waste oil and petroleum products into storm drain systems. In spite of the presence of a petroleum component in urban runoff, PAHs associated with urban runoff and, in turn, in receiving urban sediments, are typically dominated by pyrogenic PAH (Eganhouse et al. 1982). This is because 1) the PAH in storm water run-off often have a pyrogenic PAH signature to begin with; and 2) the 2- and 3- ring PAH are more water-soluble and, therefore, degrade faster than the HMW PAH.

### 8.3.3 Chromatographic pattern recognition

From the results of the PCA, four “extreme” samples (samples at the outer boundaries of Figure 8-3) are examined to further identify product sources and possible sediment mixing. Sediments from the following stations were further examined: Station 1C, 0-1 ft (1C1), Station 3S, 1-2 ft (3S2), Station 6S, surface (6S0) and Station 6C, 0-1 ft (6C1).

Station 1C, 0-1 ft. The GC/FID chromatogram of this sample (Figure 8-6a) is dominated by the presence an unresolved complex mixture (UCM), which appears as a hump, in the lubricating oil range (approximately  $n\text{-C}_{20}$  to  $n\text{-C}_{30}$ ). Resolved hydrocarbons at low levels are also observed in the diesel

range (approximately  $n\text{-C}_{10}$  to  $n\text{-C}_{20}$ ). The PAH distribution profile for this sample (Figure 8-6b) exhibits a distribution pattern characteristic of a petroleum product that has been degraded. The relatively lower proportion of PAH and LAB to that of TPH is a characteristic of petroleum-derived sources. Alkylated PAH groups are higher than the corresponding parent PAH also indicating impacts from petroleum. The presence of these types of petrogenic distributions of PAH, from naphthalenes (2-ring PAH) through chrysenes (4-ring PAH), suggest possible fuel oil (e.g., diesel or #2 fuel oil) and lube oil sources. The sample also contains combustion-related PAH, indicating pyrogenic inputs.

Note that the distribution within each homologous PAH series has been altered from the initial "bell" shape typical of petroleum, to one dominated by higher alkylation and declining concentration with declining alkylation. This alteration is due to physical processes described as weathering, which includes volatilization, water dissolution and hydrolysis, as well as biological processes described as biodegradation (e.g., bacterial degradation). In petroleum derived sources, such as crude oil, PAH typically comprise a substantially lower proportion of total petroleum hydrocarbons (i.e., 1-5%) compared to those found in combustion derived hydrocarbon products such as coal tars which are enriched in PAH. PAH in coal tars typically comprise as much as 50% of the petroleum hydrocarbons. In this sample total PAH comprise 0.6% of the TPH. The hydrocarbon distribution in this sample indicates primary input from fuel oil and secondary inputs from lubricating oil.

Station 3S, 1-2 ft. The GC/FID chromatogram of this sample (Figure 8-7a) is also dominated by a UCM in the lubricating oil range. Compared to Station 1C, 01 ft., there are fewer resolved hydrocarbons in the diesel range. In fact, the diesel-range normal alkanes targeted in the analysis are below detectable levels. The PAH distribution profile of this sample (Figure 8-7b) is dominated by pyrogenic PAH, specifically pyrene and benzo[b]fluoranthene. Lower molecular weight PAH (i.e., 2-3 ring) are present but at relatively low levels. The bell-shape of the distribution of the phenanthrene/anthracene series indicates influence from a light-range petroleum. The overall sample distribution displays characteristics similar to that of typical atmospheric dust with the exception of the relative depletion of fluoranthene which is most often found at concentrations comparable to pyrene. Total PAH comprise a greater proportion (0.9%) of the TPH than the previous sample, but are lower than most combustion products such as coal tar and creosote. The hydrocarbon distribution in this sample suggests a mixture of lubricating oil and a combustion-related source.

Station 6S, surface. The GC/FID chromatogram of this sample (Figure 8-8a) is dominated by resolved and hydrocarbons in the lubricating oil range. The PAH distribution profile in Figure 8-8b is dominated by pyrogenic PAH, especially fluoranthene, pyrene, and benzo(b)fluoranthene. The distribution within several homologous series, especially phenanthrene and chrysene, exhibit characteristics typical of pyrogenic derived sources. In combustion related products the parent compound dominates each homologous series and concentration declines with greater alkylation.

The greatest difference between this sample and the Station 3S, 1-2 ft sample are that PAH (and LAB) comprise a greater percentage of the measured TPH. Total PAH comprise 2.6% of the TPH concentration in this sample. In addition, fluoranthene is not depleted as observed in the previous sample. The PAH distribution observed is characteristic of atmospheric deposition without a great deal of impact from lubricating oils, typically associated with highway runoff. This is indicative of the mixed sources contributing to the hydrocarbon distribution at this location.



Station 6C, 0-1 ft. The GC/FID chromatogram of this sample (Figure 8-9a) is dominated by resolved and unresolved hydrocarbons in the lubricating oil range. Overall levels of TPH and PAH are lower than the other samples discussed. The PAH distribution profile (Figure 8-9b) is dominated by pyrogenic derived PAH, especially fluoranthene, pyrene, and benzo(b)fluoranthene. The distribution within several homologous series, especially phenanthrene and chrysene, exhibit characteristics typical of pyrogenic derived sources. The proportion of total PAH to TPH is in the mid-range of those discussed (1.2%), which is why the PCA plot positions this sample near the x-axis origin. However, PCA 2 separates this sample from the main cluster of samples, which are from a mixture from pyrogenic and petrogenic sources due to the elevated levels of linear alkylbenzenes (LAB-13 in particular).

As previously discussed, the primary signature observed in the sediment samples is a mixture of a heavy oil, such as lubricating oil, and combustion-related hydrocarbons. This mixture is similar to that of urban runoff, which would include the lubricating oils from engine crankcases that are released to road surfaces and the atmospheric deposition of the combustion products of gasoline and other fuel oils. This contamination is ubiquitous in waterways in urban areas, especially those that drain highways. The relative concentrations of the lubricating oil component and the PAH content in any particular location is a function of localized inputs, weather and the energy of the depositional environment. Low energy depositional environments, such as Islais Creek, may accumulate substantial concentrations of these contaminants, which are associated with fine-grained particles and organic matter.

Figure 8-10 compares PAH distributions for a sediment from a pond that received direct drainage from a four lane urban highway and a representative sediment from the end of Islais Creek (Station 2S, surface). There were no CSO inputs associated with the pond sample, which has a PAH distribution that is typical of urban runoff. The ellipses on the plot indicate the presence of PAH generally associated with petroleum sources. Note that alkylated naphthalenes and alkylated dibenzothiophenes (associated with light fuels such as diesel) are present and of similar distribution in both samples. Alkylated chrysenes (associated with heavy oils) are present in both samples but relative concentrations are higher in the pond sample. The pyrogenic PAH distributions are remarkably similar in both samples. As discussed, these PAH, which are generated by combustion of gasoline and fuel oil (as well as other pyrogenic sources such as wood stoves and forest fires) are ubiquitous and enter near coastal sediment environments primarily through atmospheric fallout and storm runoff.

PAH concentrations in the pond sediment "comparison" sample are significantly higher than PAH concentrations measured in Islais Creek sediments. The fact that highway runoff and atmospheric fallout were the primary sources of elevated PAH concentrations in a typical non-CSO road-side sediment, indicates that similar sources could contribute substantially to the PAH contamination measured at Islais Creek. Characterization of similar sediment environments in San Francisco Bay are required to substantiate this conjecture.

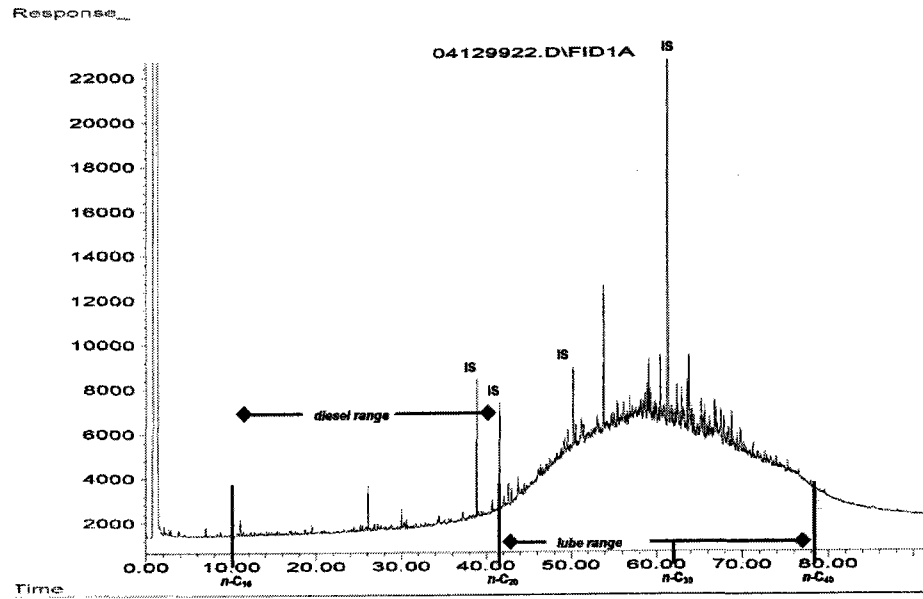


Figure 8-6a. TPH GC/FID chromatogram for Station 1C, 0-1 ft.

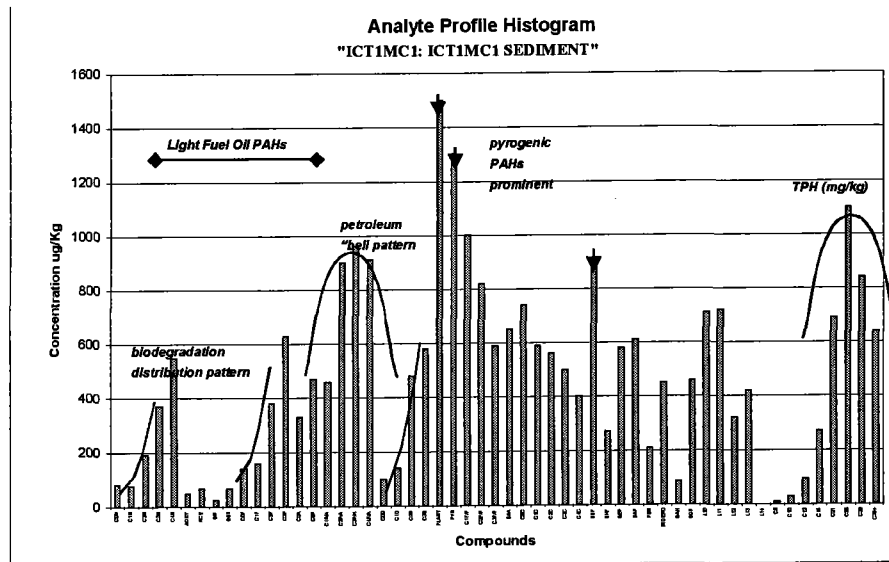


Figure 8-6b. PAH, LAB and TPH distribution for Station 1C, 0-1 ft.

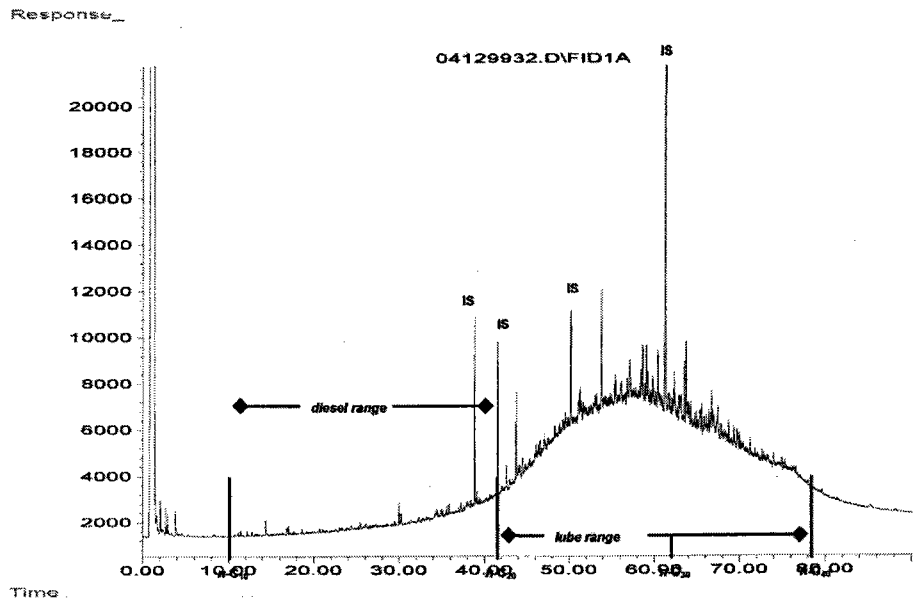


Figure 8-7a. TPH GC/FID chromatogram for Station 3S, 1-2 ft.

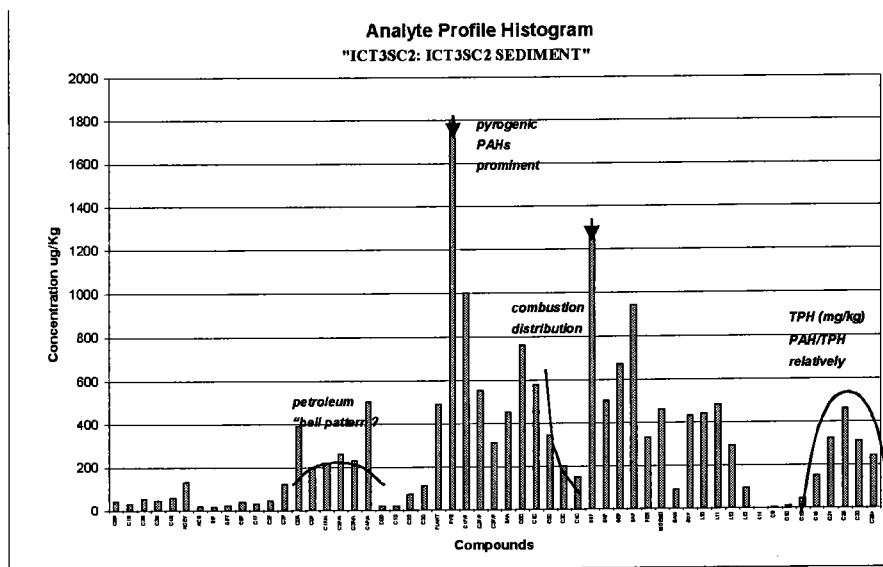


Figure 8-7b. PAH, LAB, TPH distribution for Station 3S, 1-2 ft.

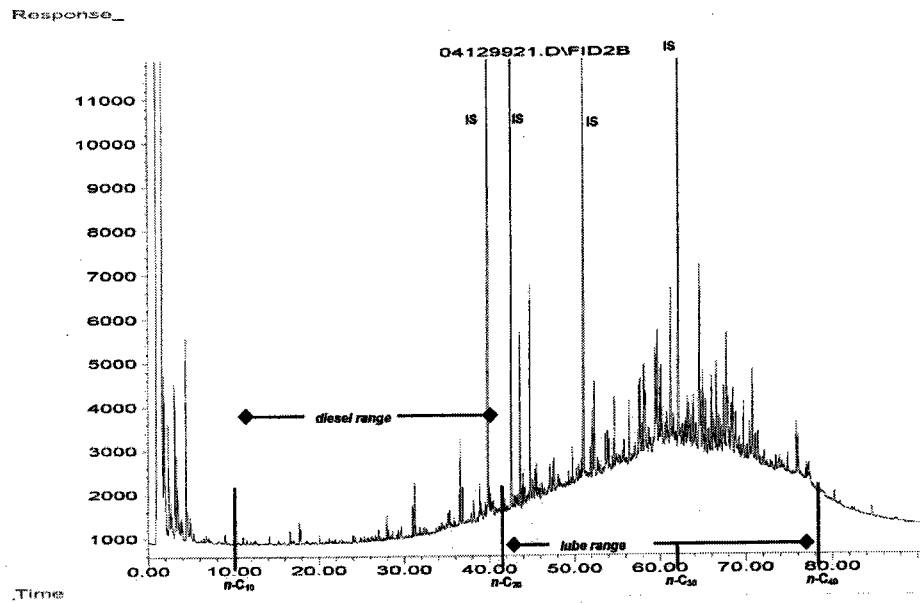


Figure 8-8a. TPH GC/FID chromatogram for Station 6S, surface.

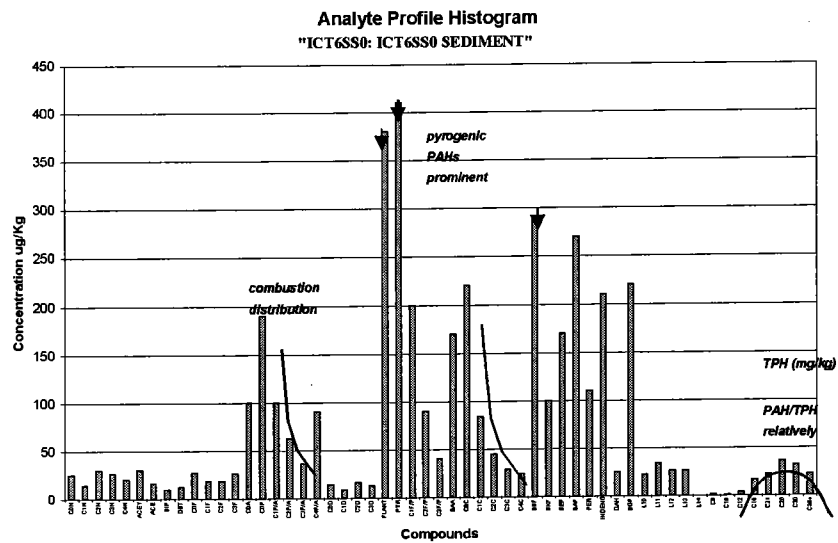


Figure 8-8b. PAH, LAB, TPH distribution for Station 6S, surface.

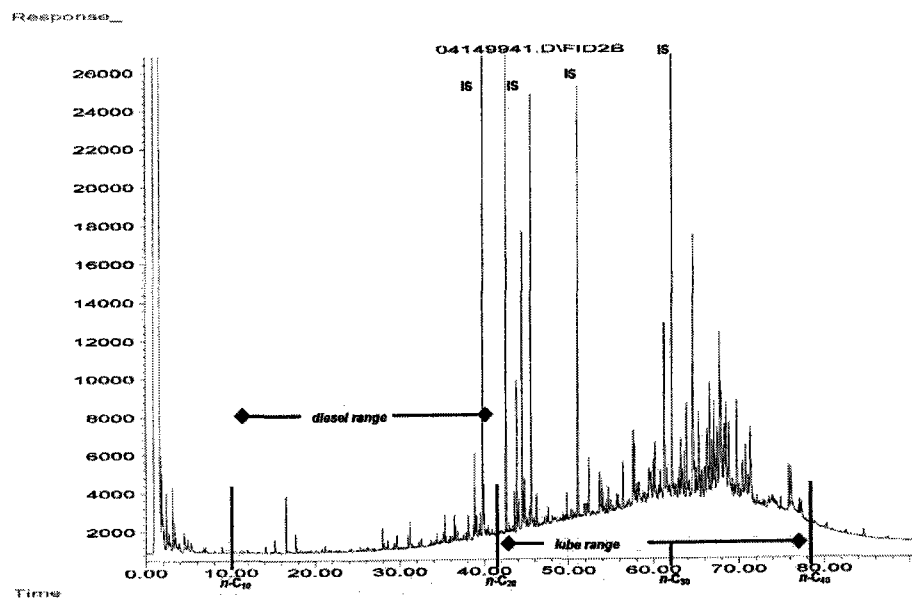


Figure 8-9a. TPH GC/FID chromatogram for Station 6C, 0-1 ft.

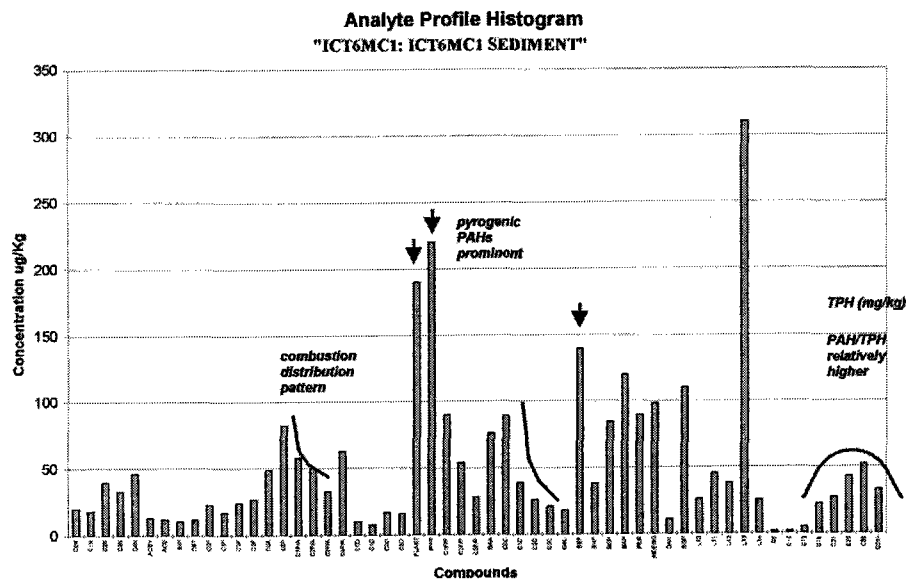


Figure 8-9b. PAH, LAB, TPH distribution for Station 6C, 0-1 ft.

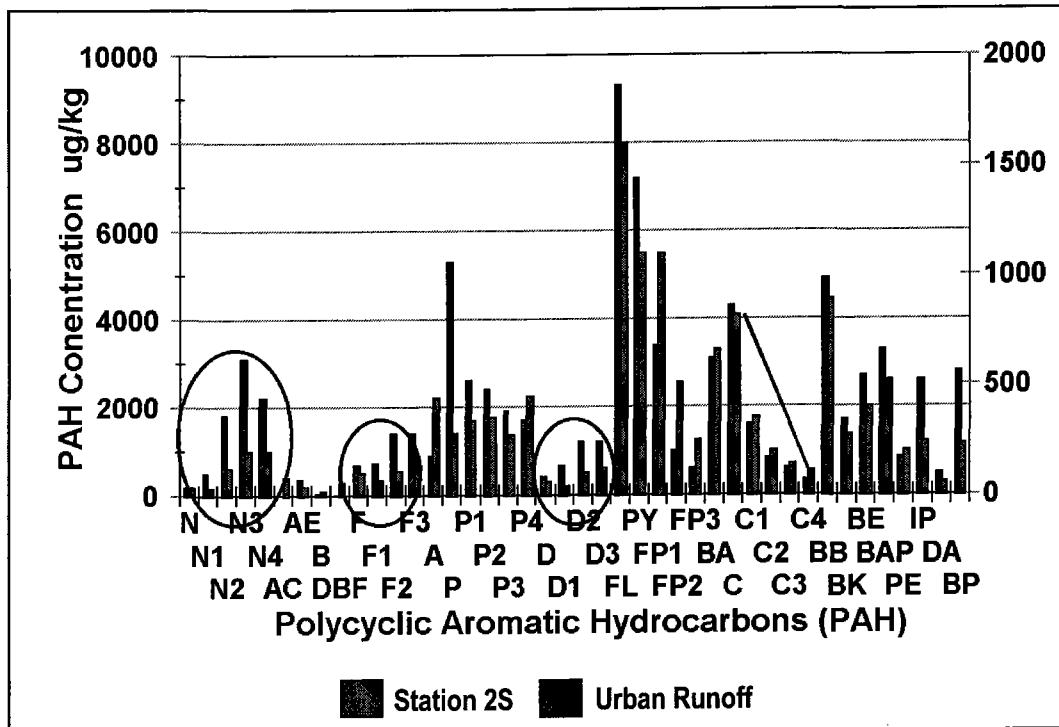


Figure 8-10. Pond sediment with highway drainage compared to Islais Creek surface sediment from Station 2S.

## 9.0 APPLICATION OF DECISION RULES AND RECOMMENDATIONS

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The decision matrix initially introduced in Section 3 (Table 3-1) is applied to sediment chemistry, toxicity, and bioaccumulation results in this section. This approach identifies creek sediment locations that qualify as “toxic hot spots” (as defined in the BPTCP), as well as locations that show no significant toxicity but are potential sources of contaminant transfer to the food web. Also presented are recommendations for possible further study, including remedial alternatives analysis for localized creek areas.

The decision matrix proposes action based on three lines of evidence that are assumed to be proportional to ecological impact to creek sediments. These actions address not only the conditions necessary for toxic hot spot designation, but other potential outcomes based on the lines of evidence. One of these evidentiary lines, bioaccumulation, was investigated by SFPUC even though it was not used in the BPTCP. This was done because certain COPCs, confined to the western end of both creeks, are known to bioaccumulate in marine food webs even though they rarely cause toxicity in 10-day acute amphipod tests. SFPUC is not suggesting that these data be used to redefine a toxic hot spot, rather that they are considered ancillary information in the likely event that BPTCP data are used in other regulatory programs (e.g., 303[d] listing) for which they were not originally intended.

### 9.1 ISLAIS CREEK

Table 9-1 shows results for sediment chemistry, toxicity and clam bioaccumulation tests applied to the decision matrix, and where creek stations were statistically elevated compared to in-bay reference stations. The first row identifies only two stations (2N & 3S) that meet the BPTCP definition of a toxic hot spot using data from the three SFPUC surveys (1998, 1999 & 2000). These results refute previous BPTCP assertions that the entire creek is toxic, in that impacts are confined to a small localized area (<1 acre) at the west end of the creek. The BPTCP toxic hot spot listing relied on data collected from a total of three stations from which only a single location was sampled twice (in 1994 and 1997). SFPUC findings are based on a total of 18 stations, six of which were sampled in three consecutive surveys. Further, the BPTCP 1994 amphipod toxicity results for the confirmed hot spot remain suspect because unionized ammonia concentrations exceeded the test threshold subsequently required for the 10-day amphipod test (EPA/USACE 1998). Even though this requirement was established after the BPTCP tests were conducted, bioassay laboratories routinely took precautionary measures to reduce high ammonia and/or hydrogen sulfide levels, which were known to confound test results (pers. communication with F. Charles Newton, former Director of MEC Analytical Systems Bioassay Laboratories, 8/98; pers. communication with Jeff Cotsifas, Pacific EcoRisk Laboratories, 6/99). Requirements for reducing ammonia to acceptable test levels (e.g.,  $0.8 \text{ mg}\cdot\text{L}^{-1}$  for *Eohaustorius estuarius*) were established by the EPA/USACE in 1999 (in PN-99-3). The laboratory (Granite Canyon) that performed toxicity testing for the BPTCP in San Francisco Bay did not reduce ammonia or hydrogen sulfide to acceptable levels, nor did they remove potential predators from the test chambers prior to testing. Any of these factors could have contributed to the complete mortality observed in the 1997 confirmation testing of Islais Creek; and the subsequent listing of this site as a confirmed toxic hot spot.

The COPCs for Islais Creek were similar to those identified in the BPTCP; however, they were confined to the west end of the creek near the CSO Weir and the Interstate 280 overpass. Extensive sampling of 18 surface sediment stations in 1998 confirmed that COPC creek concentrations were commensurate with in-bay reference levels east of the 3<sup>rd</sup> Street Bridge (see Figure 3-1, Section 3). Laboratory detection limits were typically at or below those used in the BPTCP. Analyte lists were essentially the same between the BPTCP and SFPUC investigations, except for the 1998 SFPUC study, which expanded the number of analytes to support contaminant source identification.

**Table 9-1. Islais Creek - results of creek and reference station comparisons applied to decision matrix. Stations statistically elevated compared to in-bay reference stations are shown.**

Chemistry	Toxicity	Bioaccumulation	Results	Action
2N, 3S	2N, 3S	2N, 3S	Toxic Hot Spot – as defined by RWQCB	Candidate for remedial analysis or preventative action
1N, 1S, 2S, 3N	-	1N, 1S, 2S, 3N	Not a toxic hot spot; possible contaminant transport to food web	Possible studies to determine potential food web effects (ecological risk)

Minuses (-) denote no significant differences between creek and reference sediments for 2 or more years.

The ancillary bioaccumulation data showed elevated COPC levels in clams exposed to sediments confined to the west end of the creek. PCB and chlorinated pesticide tissue concentrations were elevated compared to in-bay reference tissues. Biological impact cannot be inferred from these statistical comparisons alone; however, the action presented in the decision matrix calls for “possible studies to determine food web effects.” An ecological risk assessment would be an appropriate study to determine if higher organisms exposed to creek COPCs through trophic transfer are at risk. For example, the bioaccumulation data coupled with conservative assumptions of prey exposure, contaminant bioavailability, and site use could be used to determine if creek sediments pose unacceptable risk to invertebrate-eating birds (e.g., diving ducks). Existing site data collected in the SFPUC surveys could be used to perform this analysis without collecting additional field data. However, based on the very small area of localized impact (site use) and the fact that bioaccumulation factors were less than unity (one) for all samples, it is unlikely that these sediments would pose unacceptable risk through trophic transfer.

The SFPUC confirmation of two impacted locations (Stations 2N & 3S) identifies “remedial analysis or preventative action” as an appropriate follow up action (Table 9-1). Any further action must be negotiated between the SFPUC and RWQCB outside of the scope of this report. However, SFPUC strongly urges that data presented in this report are considered in any subsequent remedial alternatives analysis. Subsurface data, which demonstrate that concentrations increase with depth and decrease with distance from the creek end, are of particular importance. These results strongly suggest that creek conditions are improving with time, and that buried, in place contaminants are not being transported to overlying waters or the greater San Francisco Bay.



## 9.2 MISSION CREEK

Table 9-2 shows results for sediment chemistry, toxicity and clam bioaccumulation tests applied to the decision matrix, and where creek stations were statistically elevated (compared to in-bay reference stations). Notably absent are any toxic hot spots from the 21 Mission Creek stations sampled during the three SFPUC surveys. These results are driven by the high, uniform survivals demonstrated in the 10-day amphipod test conducted throughout the creek, which are in stark contrast to the BPTCP results (Hunt et al. 1998a). The BPTCP identified a substantial portion of Mission Creek (18 acres) as a toxic hot spot based on the confirmatory sampling of a single station sampled in 1995 and 1997 at the west end of the creek. Subsequent sampling by SFPUC in 1998, 1999 and 2000 of eight creek-end stations (located west of the 4<sup>th</sup> Street Bridge) failed to confirm a single toxic hot spot. Stations east of 4<sup>th</sup> Street were sampled only in 1998, as results showed that these sediments were consistent with in-bay reference conditions and did not warrant further studies. SFPUC results refute previous BPTCP assertions that the creek is toxic, in that sediment toxicity was at or below that measured at in-bay reference stations at all 22 samples tested during the three surveys. These results were achieved even when samples were split and tested by two laboratories (Pacific EcoRisk and SFPUC Oceanside) in 1998.

**Table 9-2. Results of creek and reference station comparisons applied to decision matrix for Mission Creek. Stations statistically elevated compared to in-bay reference stations are shown.**

Chemistry	Toxicity	Bioaccumulation <sup>1</sup>	Action
1N, 1S, 2N, 2S, 3N, 3S, 4N, 4S	–	1N, 1S, 2N, 2S, 3N, 3S, 4N, 4S	Possible studies to determine potential food web effects (ecological risk)

Minuses (-) denote no significant differences between creek and reference sediments for 2 or more years.

Similar to Islais Creek, clam tissues exposed to sediments collected at the west end of Mission Creek displayed elevated concentrations of selected COPCs compared to reference tissues. These COPCs consisted of PCBs and chlorinated pesticides. Mercury in Mission Creek tissues was below reference tissue concentrations in all samples (see Section 7). The action dictated in Table 9-2 for the chlorinated organic COPC results calls for possible studies to determine potential food web effects. Following the above discussion for Islais Creek, these studies should first make use of existing data to conduct a screening ecological risk assessment, focusing primarily on impacts to higher organisms via the food web. However, the limited area of impact at the creek end (< 5 acres) coupled with strong evidence that contaminant concentrations are decreasing over time make Mission Creek an ideal candidate for natural recovery.

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