FINAL REPORT

MARINA DEL REY HARBOR SEDIMENT STRESSOR IDENTIFICATION STUDY






Prepared for:
Los Angeles County Department of Public Works
Los Angeles County Department of Beaches and Harbors
City of Los Angeles
City of Culver City
Caltrans

Final Report

Marina Del Rey Harbor Sediment Stressor Identification Study

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Technical Report ##

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EXECUTIVE SUMMARY

A Toxics Total Maximum Daily Load (TMDL) was instituted for Marina del Rey Harbor (MdRH) in 2006 in order to protect and restore fish tissue, water, and sediment quality in the Harbor. The 2015 Amendment to the TMDL required a stressor identification study to determine the cause of the impaired sediment quality by December 2016. This report describes the results of a stressor identification study conducted in 2016 to meet the TMDL requirement. The objectives of this stressor identification study are to:

- Determine current sediment quality conditions in MdRH
- Identify cause of sediment toxicity
- Characterize the stressors responsible for benthic community impacts

Sediment was collected from MdRH in two sampling periods: January 2016 and July 2016. In January, 5 stations were sampled to screen for toxicity. Based on the results of the screening, toxicity identification evaluations (TIEs) and chemistry analysis were performed at two stations. In July, 10 stations were sampled, 2 of which were repeated stations from the January sampling. No sediment toxicity to amphipods was present in July. The July samples were used to conduct additional TIE analyses, and perform a Sediment Quality Objectives (SQO) assessment, which integrated the results of toxicity, chemistry, and benthic community analyses.

The principal results of the stressor identification study are:

- **Sediment quality in MdRH does not meet the SQO for protection of the benthic community.**

  The SQO assessment found low levels of toxicity, high chemical exposure, and moderate to high benthic community impairment at the stations. Each station was classified as Likely or Clearly Impacted, indicating nonattainment of the benthic community SQO. TMDL sediment chemistry targets were not met at all locations sampled.

- **Contaminants of potential concern listed in the TMDL are not the cause of sediment toxicity.**

  TIE treatments known to inactivate the specific toxics listed in the TMDL did not significantly affect sediment toxicity, indicating that the contaminants of potential concern are not the principal cause of sediment toxicity. Additional chemical and statistical analyses confirmed the TIE characterization results. Concentrations of dissolved copper, lead, and zinc in sediment pore water were below CTR water quality objectives protective of chronic toxicity. EPA-developed equilibrium partitioning analyses confirmed that sediment metals, DDTs, PCBs, and chlordanes had low bioavailability and presented minimal risk of toxicity. Additional analyses using spiked sediment toxicity thresholds confirmed the conclusion that the TMDL listed trace organics are not a likely cause of MdRH sediment toxicity.

- **TMDL sediment targets show little correspondence to sediment toxicity occurrence.**

  Comparison of sediment chemistry and toxicity in MdRH and other bays in Southern California shows that the TMDL targets bear little relationship to the occurrence of sediment

toxicity. Sediment concentrations of total chlordanes, DDTs, PCBs, copper, lead, and zinc showed little meaningful association with the incidence or magnitude of toxicity. These results indicate that other constituents are primarily responsible for sediment toxicity and/or that the chemical quantification methods used in monitoring programs have little utility as an indicator of sediment toxicity potential.

- **No specific stressor could be identified as a cause for sediment toxicity.**

  Sediment toxicity to amphipods was only observed in the samples collected in January, and two stations were selected for TIE characterization analyses. These sediments were amended with treatments that target specific chemical classes such as organic contaminants, metals, ammonia, and pyrethroids. These TIE characterization treatments did not reduce the sediment toxicity and were unable to identify a specific chemical class as the cause of toxicity.

- **Benthic community impacts in MdRH are likely due to exposure to sediment-associated contaminants, but the specific cause is unknown.**

  The benthic community of MdRH has low abundance and diversity, especially for crustaceans, which is the group that includes the types of animals (amphipods) sensitive to sediment toxicity in the Harbor. Amphipods are known to be highly sensitive to pyrethroids and other pesticides currently in use throughout Los Angeles watersheds. The stressor characterization analyses included evaluation of MdRH community composition, presence of indicator species, and comparisons to sediment chemistry and community characteristics at other Southern California embayments. Each of the analyses identified exposure to chemical toxics as a likely stressor, with a minor indication for low dissolved oxygen stress. Methods are not available to determine which specific sediment constituents are responsible for the impacts to the benthic community. It is likely that the constituents causing sediment toxicity to amphipods in MdRH are also important causes of benthic community impacts.

- **Numeric targets for metals do not consider important site-specific conditions.**

  The greatest disparities between TMDL targets and MdRH conditions were identified for metals. Reference element normalization analyses show that TMDL targets for copper and zinc are at or below background levels expected to occur in MdRH in the absence of any anthropogenic input. Consequently, attainment of these TMDL targets is not likely to be possible under any management scenario. Data and analytical methods are available to support derivation of alternative TMDL targets that take into account natural background conditions in southern California.

- **Sediment quality in MdRH shows no evidence of improvement in the last 10 years.**

  Regional monitoring studies show that most of MdRH did not meet the SQO for benthic community protection in 2008 and 2013. Similar levels of impact on sediment toxicity, chemistry, and benthic community condition were observed for each time period.

- **Wet weather runoff from the surrounding watershed is the likely source of MdRH sediment toxicity.**

  A seasonal pattern of sediment toxicity is evident in MdRH, with a greater frequency of toxicity occurring in winter/spring. This pattern provides strong evidence that sediment toxicity in the Harbor is associated with seasonally variable inputs of unidentified toxics, likely from wet weather runoff. Additional analysis of stormwater monitoring data is suggested as a method to identify likely toxic constituents in runoff. Potential contributors to the sediment toxicity include pyrethroid pesticides, PAHs, or unmeasured toxics from runoff inputs.

- **Multiple types of management actions may be required to address sediment quality impacts.**

  A variety of management actions to restore MdRH sediment quality have been implemented or are planned. Implementation planning that considers all the toxics-related impairments in an integrated manner is recommended in order to allow for the evaluation and prioritization of management efforts that have the greatest benefit to overall water quality.

ACRONYMNS AND ABBREVIATIONS

>	greater than
<	less than
≤	less than or equal to
µg/g	microgram(s) per gram (parts per million)
µg/L	microgram(s) per liter
µm	micron
%	percent
±	plus or minus
AVS	acid volatile sulfides
Bight ’08	Southern California Bight 2008 Regional Monitoring Program
Bight ’13	Southern California Bight 2013 Regional Monitoring Program
BRI	Benthic Response Index
CA LRM	California Logistic Regression Model
CEE	carboxylesterase
cm	centimeter(s)
CMP	coordinated monitoring program
CSI	Chemical Score Index
CTR	California Toxics Rule
DDT	dichlorodiphenyltrichloroethane
DQO	Data quality objective
dw	dry weight
EBE	enclosed bays and estuaries
EI	electron ionization
ERM	effects range median
ESB	equilibrium sediment benchmark
foc	fraction of carbon
g OC	grams organic carbon
hr	hours
GCMS	gas chromatograph mass spectrometry
ICPMS	inductively coupled plasma mass spectrometry
IBI	Index of Biotic Integrity
ID	identification
m	meters
LA	load allocation

ACRONYMNS AND ABBREVIATIONS (Cont.)

LOE	line of evidence
MdRH	Marina del Rey Harbor
MLOE	multiple lines of evidence
mm	millimeters
NCI	negative chemical ionization
NIST	National Institute for Standards and Testing
NA	not applicable to the specific sampling effort
ng/g	nanogram(s) per gram (parts per billion)
PAH	polycyclic aromatic hydrocarbon
PBDE	polybrominated diphenyl ether
PBO	piperonyl butoxide
PCB	polychlorinated biphenyl
psu	practical salinity units
QA	quality assurance
QC	quality control
RBI	Relative Benthic Index
RIVPACS	River Invertebrate Prediction and Classification System
RL	reporting limit
RWQCB	Regional Water Quality Control Board
SCCWRP	Southern California Coastal Water Research Project
SEM	simultaneously extracted metals
sp.	species
SQO	sediment quality objective
SWI	sediment-water interface
SWRCB	State Water Resources Control Board
TIE	toxicity identification evaluation
TMDL	Total Maximum Daily Load
TN	Total Nitrogen
TOC	total organic carbon
TU	toxic unit
USEPA	United States Environmental Protection Agency
WLA	waste load allocation
wt	weight
ww	wet weight

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1. INTRODUCTION

The Marina del Rey Harbor (MdRH) Toxics Total Maximum Daily Load (Toxics TMDL) was instituted in 2006 to protect and restore fish tissue, water, and sediment quality in the Harbor through the remediation of contaminated sediments and control of ongoing sediment loadings. The 2015 revision of the MdRH Toxics TMDL required a special study to identify the chemical stressors responsible for impaired sediment quality (stressor identification study) by December 2016 (Attachment A to Resolution No. R14-004; Los Angeles RWQCB 2014). This report describes the results of the sediment stressor identification study as well as potential management actions.

1.1 Study Objectives

The overall goal of the stressor identification study was to obtain key information needed to develop effective management strategies to improve benthic community health in Marina del Rey Harbor. There are three primary reasons why this study was needed:

- It is a required element of the revised MdRH Toxics TMDL and State's Water Quality Control Plan for Enclosed Bays and Estuaries – Part 1 Sediment Quality (EBE Plan Part 1, (SWRCB 2009))
- Current sediment quality conditions in MdRH must be determined in order to evaluate attainment status of TMDL targets
- Identification of the cause of impacts to benthic organisms is needed in order to develop chemical-specific management strategies for sediment toxics, if necessary

The outcome of this study will be used to inform the development of management strategies for MdRH sediment contamination. The study was designed to accomplish four study objectives:

- Determine the current sediment quality condition of MdRH using the SQO assessment framework
- Identify the chemical stressors responsible for sediment toxicity
- Determine the relative impact of chemical contamination on benthic community condition
- Update information on the sources of key chemical stressors and the thresholds needed to minimize sediment quality impacts

1.2 Report Organization

This report is organized into nine main sections:

1. **Introduction**. Background information on regulatory context, sediment quality, stressor identification, and development of management actions.
2. **Study Design and Methods**. Describes the overall study design and chemistry, toxicity, and benthic analysis methods.

3. **Sediment Quality Survey.** Describes methods and data analysis specific to the SQO assessment framework. Provides SQO assessment results and comparison to previous studies.
4. **Toxicity Identification Evaluation**. Describes the TIE specific methods and detailed data analysis used for toxicity confirmation. Summarizes and discusses results from the toxicity screening and characterization studies. Discusses toxicant confirmation in the context of bioavailable contaminants, toxicity thresholds, and sediment contaminant-toxicity associations.
5. **Benthic Community Stressor Identification.** Describes the data compilation and analysis methods. Evaluates the relative importance of contaminant and non-contaminant factors on benthic community stressors.
6. **Management Actions.** Describes potential management actions based on the study results.
7. **Key Findings.** Summary of study results.
8. **References**
9. **Appendices**

1.3 Regulatory Background

This section provides a brief overview of the regulatory background for MdRH that led to the inclusion of a stressor identification study in the revised Toxics TMDL and also discusses the importance of this study in developing scientifically defensible management actions.

Based on prior 303(d) listings, the MdRH Toxics TMDL was approved in 2006 to address impairments associated with sediment for copper, lead, zinc, chlordane, PCBs, and toxicity, and with fish tissue for DDT, dieldrin, chlordane, and PCBs (Los Angles Regional Water Quality Control Board 2013). Monitoring and special studies conducted in support of the Toxics TMDL have since provided additional information regarding the spatial extent and magnitude of the impairments. The results have shown the presence of sediment toxicity.

The Toxics TMDL was revised and adopted by the Los Angeles Regional Water Quality Control Board (RWQCB) in February 2014, approved by the State Water Resources Control Board (SWRCB) in September of 2014, and approved by the United States Environmental Protection Agency in October of 2015.

The MdRH Toxics TMDL encourages collaboration and coordination of monitoring, reporting, and implementation efforts. Named responsible parties with sediment waste load allocations (WLAs) in MdRH include:

- County of Los Angeles
- Los Angeles County Flood Control District
- California Department of Transportation
- City of Los Angeles

- City of Culver City

Named responsible party with in-harbor sediment load allocations (LAs):

- County of Los Angeles

1.3.1 TMDL Compliance

The MdRH Toxics TMDL compliance metrics are based on achieving WLAs in MdRH, which have been developed to limit sediment-bound pollutant loadings from upstream and on-land sources. In addition, the TMDL set LAs in MdRH to limit concentrations in bed sediments believed to impact marine benthos and fish tissue (human health effects). Mass-based limits for chemical constituents are provided in Attachment A to Resolution No. R14-004, 004 (Los Angeles RWQCB 2014).

As specified in the TMDL, compliance with sediment WLAs for metals, chlordane, p,p'-DDE, and total DDTs may be demonstrated via any one of three different means:

1. The categorical sediment condition of "Unimpacted" or "Likely Unimpacted", by interpreting and integrating multiple lines of evidence (MLOE) as defined in the State's Water Quality Control Plan for Enclosed Bays and Estuaries – Part 1 Sediment Quality (EBE Plan Part 1, (SWRCB 2009)).
2. Sediment numeric targets are met in bed sediments (Table 1-1).
3. Final sediment WLAs are met.

Table 1-1. TMDL numeric targets for sediment contaminants.

Contaminant	Target
Copper	34 mg/kg
Lead	46.7 mg/kg
Zinc	150 mg/kg
Chlordane	0.5 µg/kg
Total PCBs	3.2 µg/kg
Total DDTs	1.58 µg/kg
p,p' DDT	2.2 µg/kg

For PCBs, compliance with sediment WLAs may be demonstrated via any of four different means:

1. Fish tissue targets are met in species resident to the waterbody.
2. Final sediment WLAs are met.
3. Sediment numeric targets to protect fish tissue are met in bed sediments.
4. Demonstration that the sediment quality condition protective of fish tissue is achieved per the EBE Plan Part 1, as amended to address contaminants in resident finfish and wildlife.

Guidance for evaluating sediment condition under the SQO program is provided in the EBE Plan Part 1. A SQO and assessment framework for protection of aquatic life (i.e., benthic invertebrate community) from sediment contamination has been developed for bays and estuaries in California based on an approach that incorporates MLOE. These MLOE include sediment chemistry, sediment toxicity, and benthic community condition. A SQO for protection of human health, has also been adopted by SWRCB, and an associated assessment framework and implementation guidance is under development; however, preliminary guidance is currently available.

1.3.2 TMDL Implementation

Meeting goals and targets in complicated TMDLs requires a holistic approach that includes source identification and control from sources including the watershed, water column, and in-place (bed) sediments. Components of a holistic approach include:

- Monitoring
- Watershed management plans
- Sediment management plans
- Special studies, such as stressor identification (Stressor ID), source identification, BMP effectiveness, and chemical fate and transport processes

The Stressor ID special study is only one component in a larger effort to meet the goals of a TMDL concerned with legacy pollutants in harbor sediments. As required by the revised Toxics TMDL, this Stressor ID study only focuses on impacts to benthic organisms (e.g., sediment toxicity), not human health risk resulting from ingestion of fish.

1.3.3 Stressor Identification Study Requirement

The EBE Plan Part 1 (SWRCB 2009) requires additional investigations to be conducted to confirm impacts and identify causative agents when sediment contamination impairment to benthic community is evident. This stressor identification study includes various types of analyses: toxicity identification evaluation, chemical bioavailability analysis, and comparative analysis.

Previous investigations have found that MdRH sediments failed to meet the SQO for benthic community in accordance with the EBE plan. Consequently, the revised MdRH Toxics TMDL requires a stressor identification study, compliant with the requirements of the EBE Plan Part 1, be conducted to identify causative agents that impair or impact benthic organisms (e.g., sediment toxicity or benthic community stress). In addition, the TMDL also requires the evaluation of MdRH sediment quality using SQO MLOE assessment methodology every five years.

1.4 Marina del Rey Harbor Sediment Quality Background

Multiple special studies have investigated the quality of Marina del Rey Harbor sediment and other water quality factors as part of the toxics TMDL. The scope of the previous studies has been quite variable, some being comprehensive with toxicity, chemistry, and benthic community assessments, while others were focused on fewer parameters or a more limited number of stations. The subsections below summarize these studies and their key findings.

1.4.1 2007 Sediment Characterization Study

The first comprehensive study of Marina del Rey sediments following establishment of the TMDL was conducted by Weston Solutions in September of 2007 (Weston Solutions 2008). This study included the collection of both surficial sediment grabs and subsurface vibracore samples. Samples were taken from at least one station within each basin with an additional five stations located in the main channel (Figure 1-1). Sediment toxicity was tested using the amphipod (*Eohaustorius estuarius*) 10-day survival test. Sediment chemistry and benthic community condition were measured at each of the grab stations. Results from the toxicity, chemistry and benthic community measurements were used to evaluate sediment quality assessments using the assessment framework established for determining attainment of the Water Board's SQO for protection of the benthic community in sediments (SWRCB 2009).

The results of this study found that only two of 16 stations, one in Basin H and one in the main channel, had acceptable sediment quality, with a classification of Likely Unimpacted (Figure 1-1). Of the other 14 stations, nine fell into the Clearly Impacted category, which is the most severe category of impact. The five remaining stations were classified as either Likely Impacted or Possibly Impacted. Those samples having impacted sediment quality usually showed evidence of elevated sediment chemistry, disturbed benthic communities, and sediment toxicity. With the exception of stations in Basins A and H, and one main channel station, the stations contained moderate to high sediment toxicity.

The surficial sediment chemistry results from the Weston study were used in the 2014 TMDL reconsideration as part of the justification for both the continued inclusion of specific chemicals in the TMDL and expansion of the TMDL to the entire harbor. The characterization study found than multiple front basin stations exceeded the ERMs for chlordane and copper, meeting the requirements for 303d listing.

Figure 1-1. Sediment quality assessment category results for 2007 sediment characterization study. Colored symbols indicate assessment results for surface sediment collection stations. Sediment core stations were not evaluated for sediment quality assessment category. Figure adapted from Weston 2008.

1.4.2 Coordinated Monitoring Program

The Coordinated Monitoring Program (CMP) has conducted periodic monitoring for the MdRH Toxics TMDL. During the Ambient Monitoring Phase, the CMP included monthly water column chemical measurements as well as less frequent sediment toxicity and chemistry analysis in the back basins of the Harbor. Sediment samples were collected and analyzed quarterly from September 2010 through June 2011 and then every six months from September 2011 through 2016. During the Effectiveness Monitoring Phase, the sediment chemistry analysis was increased

to monthly, while the monthly water column sampling was no longer required. The toxicity testing included the *Eohaustorius estuarius* 10-day amphipod survival test as used in the sediment characterization study, but also included a 28-day chronic test using another amphipod species, *Leptocheirus plumulosus*, with survival, growth and reproduction test endpoints. In addition, toxicity testing at the sediment-water interface was conducted using embryos of the mussel, *Mytilus galloprovincialis*; another toxicity test specified for use in SQO assessment. Sediment chemistry was limited to constituents listed in the TMDL. Bioaccumulation monitoring was conducted annually.

Toxicity results from the CMP indicated less toxicity to *E. estuarius* than observed in the 2007 characterization study, and the toxicity was not consistent on a spatial or temporal basis (County of Los Angeles Department of Public Works 2012b, 2012a, 2013). Testing with the *L. plumulosus* 28-day test found fairly consistent strong toxic effects on amphipod survival, growth, and reproduction.

Sediment chemistry results for the CMP found that the concentrations of copper, lead, and zinc did not meet the TMDL targets for all samples analyzed. PCBs did not meet the TMDL target for most samples. Chlordane was never found above the TMDL target, but the reporting and detection limits were more than an order of magnitude higher than the TMDL target, making interpretation of the results inconclusive.

1.4.3 Southern California Bight Regional Monitoring Program

The Southern California Bight Regional Monitoring Program is a multi-agency survey which seeks to determine sediment quality conditions throughout the Bight. Samples have been collected and analyzed approximately every five years since 1994. In 2008, the Bight Program (Bight '08) modified its methods to be compliant with the SQO program, using both the *Eohaustorius estuarius* 10-day survival and mussel (*Mytilus galloprovincialis*) embryo sediment-water interface tests for sediment toxicity (SWRCB 2009). Methods for chemistry and benthic community analysis were also SQO compliant. These SQO compliant methods were also used in Bight '13. Bight '08 sampled five stations located within MdRH, one each in Basins C, E, and G, one in the main channel near the lower basin and one in the main channel near the inlet (Figure 1-2). In Bight '13, four stations were sampled in MdRH, three in the main channel and one in Basin E (Figure 1-3).

The SQO assessment for the Bight'08 program found that only the station near the channel opening met the SQO, with a category of Likely Unimpacted (Southern California Bight 2008 Regional Monitoring Program Coastal Ecology Committee 2012). The other main channel station and the station in Basin C were classified as Possibly Impacted. The stations in Basins E and G were in the Likely and Clearly Impacted categories respectively. The Bight '13 results indicated that the Basin E station was in the Likely Impacted category, while the main channel stations ranged from Likely Unimpacted to Likely Impacted (Southern California Bight 2013 Regional Monitoring Program Coastal Ecology Committee, In Prep).

For Bight '08, none of the stations were found to be toxic to the mussel embryo test. The stations in Basins C and E and in the main channel near the inlet were all classified as Nontoxic to the amphipod test. The lower basin main channel station was found to have Low Toxicity to the amphipod test, while the station in Basin G had High toxicity. A Toxicity Identification

Evaluation (TIE) was conducted on sediment and pore water from the Basin G station (Section 1.5.2). The Bight '13 survey found all of the main channel stations to be Nontoxic to amphipods and only one station had Low Toxicity to the mussel test. The Basin E station had Low Toxicity to the amphipod test and was Nontoxic to mussel embryos.

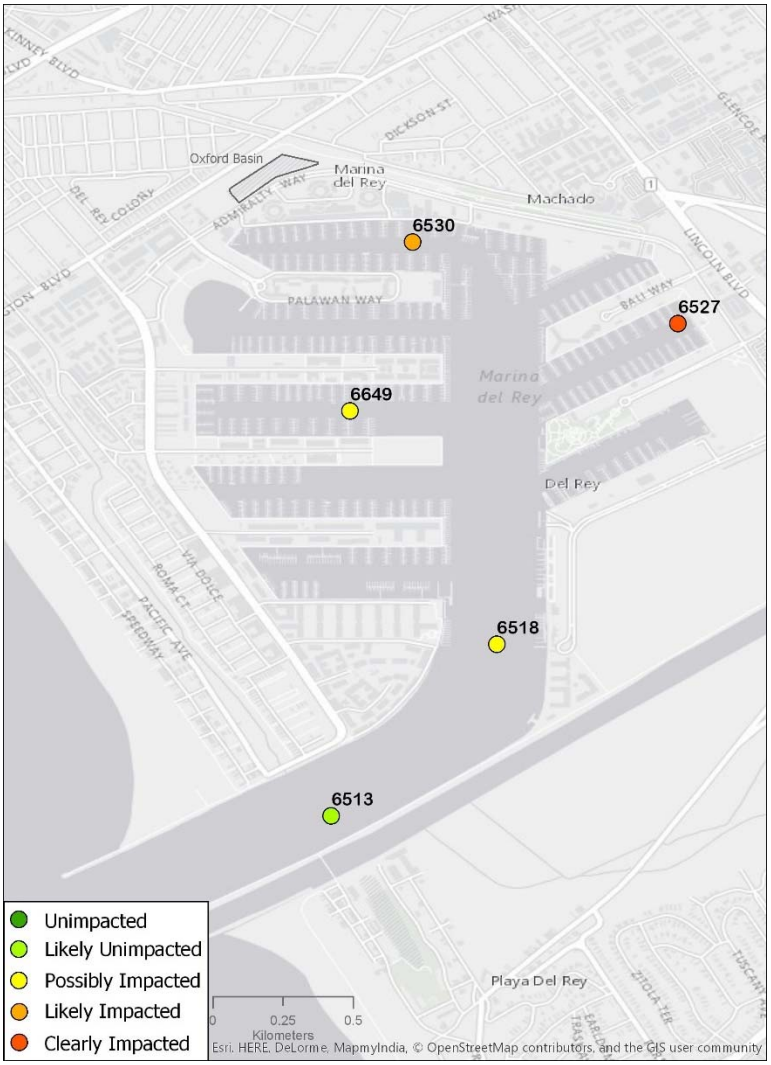

Figure 1-2. Sediment quality assessment category results from Bight'08 regional monitoring survey.

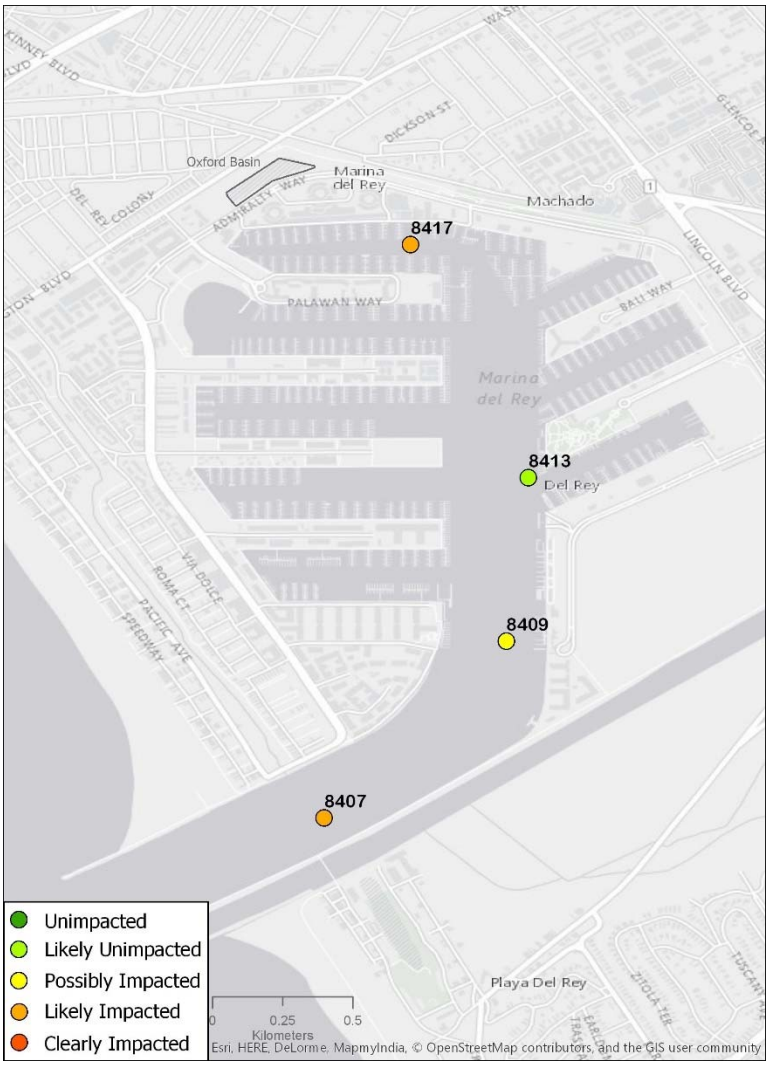

Figure 1-3. Sediment quality assessment category results from Bight'13 regional monitoring survey.

The other two SQO lines of evidence showed greater impacts than did toxicity in both the 2008 and 2013 surveys. The sediment chemistry line of evidence identified the station in Basin C as being in the Moderate chemical exposure category, while the remaining stations were all in the High Exposure category in 2008. In 2013, the Basin E station and two of the main channel stations were in the High Exposure category while the remaining main channel station was in the Moderate Exposure category. Sediment contaminant concentrations were above their respective TMDL target values for all stations. Regarding benthic community condition, the three basin stations were identified as having Moderate Disturbance, while the two channel stations were in the Low Disturbance category for Bight '08. In Bight '13, the station in the lower main channel was found to be in Reference condition while the remaining three stations were all in the Moderate Disturbance category.

Bight surveys in 1998 and 2003 also detected toxicity in MdRH sediments using the same *E. estuarius* amphipod survival test. Seven stations in MdRH were sampled in the 1998 survey.

Three stations in the main channel and one in Basin H were found to be Nontoxic (Bay et al. 2005). One additional station in the main channel and stations in Basins C and E had Moderate Toxicity. For the 2003 survey, six stations in MdRH were analyzed. Two stations in the main channel and one in Basin F were Nontoxic (Bay et al. 2011). One main channel station and two stations in Basins E and B had Moderate Toxicity in 2003.

1.5 Stressor Identification

Stressor identification is a broad term that encompasses a suite of analyses designed to determine the cause of biological impacts at a site. In some applications, the analyses are focused on the compilation and synthesis of existing information about the site or adjacent areas (USEPA 2000). In other cases, as with the MdRH study, additional site-specific analyses are conducted to support more detailed and robust conclusions.

Multiple types of biological impacts can be investigated using stressor identification, including sediment toxicity and degraded benthic communities. The scope of stressor identification is also not limited to chemical pollutants, as other types of stressors (e.g., nutrients, habitat alterations) can also cause biological impairments. However, the stressor identification process described for the SQO program in the EBE Plan Part 1 has a greater emphasis on chemical contamination impacts than the approach used for other programs. The SQO stressor identification process consists of three types of studies (SWRCB 2009):

1. Confirmation and Characterization of Pollutant Related Impacts
2. Pollutant Identification
3. Source Identification and Management Actions

The goal of the first step is to verify that the cause is chemical in nature, as opposed to something not related to toxics, such as sediment grain size or excessive nutrient loading. The outcome of this step can have an important influence on the interpretation of the SQO outcome and need for additional studies. If the adverse biological effects are determined to be caused solely by non-chemical contaminant stressors, then there is the possibility that the sediment may be reclassified as meeting the SQO, altering the need for subsequent stressor identification studies. This step is particularly important for investigating impacts to the benthic community, which is more susceptible to non-contaminant stressors in the environment than are laboratory sediment toxicity tests.

Confirmation that biological impacts at the site are likely due to chemical contaminants leads to the second step of stressor identification: identifying the pollutants responsible. The specific studies conducted under this step vary, depending upon the nature of the effect (e.g., sediment toxicity or benthic community impact). In most cases, the adverse effect being studied is toxicity and the aim is to determine the specific chemicals responsible so that management actions can be taken to reduce or eliminate the impact. Identifying the cause of toxicity includes application of various methods and analyses that are collectively known as a Toxicity Identification Evaluation (TIE). Standardized sediment TIE methods are available that include toxicity tests of sediment following chemical or physical manipulation in the laboratory, as well as statistical analysis of chemistry and toxicity data from multiple stations (USEPA 2007).

The source identification and management actions identified in the EBE Plan Part 1 are the same as the steps followed in developing a TMDL. The SQO plan recommends conducting source identification and load allocation studies following toxic pollutant identification in order to support the development of management actions likely to be most effective in restoring sediment quality.

1.5.1 Toxicity Identification Evaluation Background

Standardized methods used for TIEs in the aqueous environment have been available for the past two decades (USEPA 1991, 1996). Methods for use in whole sediments have been available for a shorter time, and due to the complexities of the sediment matrix, there are fewer methods available than for water (USEPA 2007). TIE methods are grouped into three phases. In Phase I (characterization), generalized treatments are used to identify broad classes of chemical toxicants (e.g. non-polar organics). In Phase II (identification) more specific methods are used to narrow down or identify the likely chemicals causing toxicity. In Phase III (verification), additional approaches are applied to verify the results found in the previous phases. The verification methods may include sediment spiking and statistical analyses. These methods have been previously applied on a limited basis in MdRH and in a more extensive study in neighboring Ballona Creek.

1.5.2 Previous TIE Studies

Bight '08

In conjunction with the 2008 Southern California Bight Regional Monitoring Program, a sediment TIE was performed on one station in MdRH Basin G (Bay et al. 2011). Initial testing measured high toxicity at this station, with only 20% survival of *E. estuarius* in the 10-day test. When the TIE was conducted on the same sample several weeks later, the baseline survival had risen to 73%, indicating the toxicity had decreased during sediment storage. The change in toxicity suggested that the toxicant was subject to relatively rapid chemical/biological degradation, as is characteristic of some current use pesticides or biological toxins. The reduction in toxicity during storage complicated interpretation of the TIE, with most of the treatments yielding inconclusive results. However, TIE tests of the sediment pore water from this station indicated that pyrethroid pesticides were present.

Ballona Creek Estuary

A sediment TIE study was conducted in the Ballona Creek Estuary in 2007-2009. The objective of this study was to determine the cause of toxicity in Estuary sediments to the amphipod *E. estuarius* (Greenstein et al. 2014). This investigation included analysis of samples collected at different times and different locations within the Estuary. All three TIE phases were included in the study. Additional analyses were conducted to improve confidence in the results, including: 1) deployment of passive samplers in the field to determine the bioavailable concentration of metals and organic compounds in the sediment pore water; 2) toxicity tests of sediment spiked with TMDL target chemicals to determine toxicity thresholds; and 3) comparison to toxicity thresholds to sediment contaminant concentrations in the Estuary.

Results of this study indicated that the magnitude of toxicity was highly variable both spatially and temporally. This was likely due to the highly dynamic nature of the system where high flow rates during storm events can redistribute sediment and contaminants. Similar temporal

variability would not be expected in MdRH due to its protected location and relatively small amount of stormwater input, as compared to Ballona Creek Estuary.

The TIE results for Ballona Creek Estuary indicated that none of the chemicals for which TMDL targets had been established were the likely cause of the observed sediment toxicity. Instead, multiple lines of evidence gathered during the study indicated the more likely cause of toxicity was pyrethroid pesticides. The lines of evidence supporting this conclusion included the TIE sediment manipulations, and comparisons of chemical concentrations in the sediment to toxicity thresholds derived from spiked sediment toxicity tests.

1.6 Role of Stressor Identification in Development of Management Actions

The MdRH Toxics TMDL addresses multiple classes of chemicals and types of resource level impacts: water column quality impacts (e.g., aquatic toxicity), sediment quality impacts (e.g., sediment toxicity, benthic community impairments), and fish tissue impacts (e.g., human health and wildlife impacts from ingestion of fish). From the TMDL implementation plan, attainment of water, sediment, and tissue quality will be achieved through management actions such as source reduction, source control, and sediment remediation.

A generalized management approach for sediment quality impairments to benthic organisms is depicted in Figure 1-4. The approach is initiated with a sediment quality assessment that includes evaluation of TMDL numeric chemical targets (load allocations) and a multiple lines of evidence SQO assessment (EBE Plan Part 1, (SWRCB 2009)). If sediment chemical levels are below targets (NOAA effects range low (ERL)) or the SQO assessment result is “Unimpacted” or “Likely Unimpacted”, then the site is in compliance with the water quality objectives and no further special study is required. If the outcome of the SQO assessment is “Possibly, Likely, or Clearly Impacted”, the site may advance to development of management actions. If the cause of the impairment is not known, then a stressor identification study, as proposed here, may be conducted to confirm the linkage between toxics in the sediment (stressors) and impairment.

The stressor identification process can facilitate the identification of effective management strategies. The EBE Plan Part 1 provides recommendations for additional investigations to be conducted to confirm impairment and identify causative agents. Potential studies/tools may include statistical procedures, toxicity identification evaluations, bioavailability studies, and dose/response spiking studies as proposed in this plan. The outcome of the stressor identification study may benefit development of sediment management actions in three ways:

- Identify areas not impacted by chemical contamination
- Determine stressors of highest priority for remediation
- Suggest alternative numeric targets for compliance

The results of the stressor identification study are used to inform the development of management alternatives (USEPA 2005). Alternatives considered will range from passive actions (e.g., monitored natural recovery and source control) to active remedial actions (e.g., treatment, capping, and/or dredging). For each potential management alternative, the following should be considered:

- Technical, logistical, and economic feasibility
- Social and environmental impacts
- Estimated cost
- Estimated time to complete
- Predicted load reduction to water, sediment and fish

Once an area is designated for management and available management alternatives are summarized, the relevant stakeholder group can select the appropriate action. Once all parties agree to the selected management approach and funding mechanisms are secured, the management action can be scheduled and implemented.

The MdRH stressor identification study will assist in identifying management strategies to improve benthic community health; however, it does not address other TMDL impairments related to water column copper and fish tissue chemical contamination.

TMDL implementation planning may be most efficient if the development of management alternatives considers all of the toxics-related impairments in an integrated manner, including benthic community health, fish tissue contamination, and water column copper. Such an approach will allow for the evaluation and prioritization of management efforts that have the greatest impact to the overall water quality. This approach is recommended to ensure that management actions are ecologically beneficial, and logistically and economically feasible.

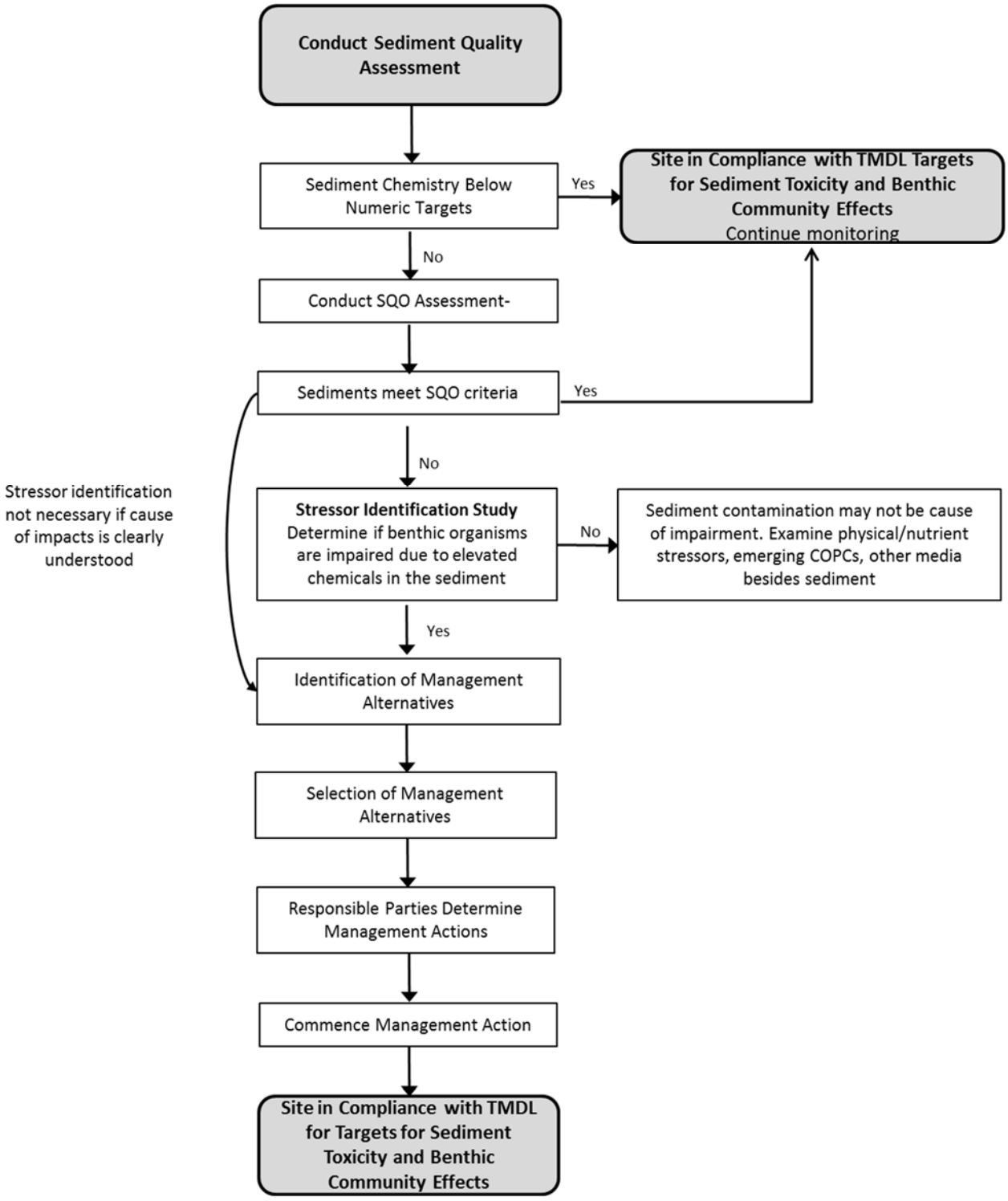

Figure 1-4. Conceptual sediment management process for benthic community impairment.

2. STUDY DESIGN AND METHODS

2.1 Chemical Analysis

Sediment samples were analyzed for trace metals and organic contaminants following USEPA SW-846 for extraction, clean-up, and analytical methods (USEPA 2008b). The chemistry analyte list included all of those measured for the Southern California Bight Regional Monitoring Program with the addition of selected current use pesticides (Table 2-1). For the January 2016 sampling event, two sediment samples were measured for each station: the chemical analysis sediment sample prepared on the day of sampling and the TIE sample prepared in the laboratory. Analyzing the chemical contaminants in both sample types allows for direct comparison to the sediments used in the toxicity screening and TIE exposures. The samples were collected and handled as described in the sampling methods.

2.1.1 Sediment Trace Organics

An aliquot of thawed wet sediment was freeze dried to constant weight prior to extraction of target organic analytes with dichloromethane (DCM) at 100°C and 1500 psi using a Dionex Accelerated Solvent Extraction (ASE) 300 system (Sunnyvale, CA, USA). The dried sediment aliquot was pre-spiked with a solution of dibromooctafluorobiphenyl (DBOFB), PCB-208, BDE-172, naphthalene-d8, acenaphthylene-d10, phenanthrene-d10, chrysene-d12, perylene-d12, and benzo[g,h,i]perylene-d12 as surrogates to track target analyte recovery. The ASE extracts were concentrated on a TurboVap 500 evaporator (Zymark, Hopkinton, MA, USA), exchanged to hexane and kept in the dark overnight at room temperature after adding activated copper powder to remove elemental sulfur. After splitting the Cu-treated sample extract into two equal portions (50:50, v/v), one split (for PAH, PCBs, OCPs and PBDEs) was chromatographed on a 30 cm length ×10 mm i.d. glass column packed, from the top to bottom, with sodium sulfate (1 cm), neutral alumina (6 cm, 3% deactivated) and silica gel (12 cm, 3% deactivated). After sample loading, two fractions (15 ml hexane followed by 60ml of hexane/DCM (70:30, v/v) were eluted from the column. The final extract volume was reduced to 0.5 ml under a gentle nitrogen stream, internal standards PCB-30, PCB-205, 2-Fluorobiphenyl, and p-terphenyl-d14 added, and stored at -20°C until analysis.

The second sample split (for pyrethroids, fipronil and its degradates) was chromatographed on a 30 cm length ×10 mm i.d. glass column packed with 10 g of 6% water deactivated Florisil. After sample loading, a single fraction of 60 ml hexane/ethyl ether (7:3, v/v) was eluted from the column. The final extract volume was reduced under a gentle nitrogen stream to 0.5 ml PCB-205 added as an internal standard, and stored at -20˚C until analysis (Lao et al. 2010).

Extracts were analyzed on an Agilent 7890 gas chromatograph (GC) coupled to a 5975C quadrupole mass selective detector (MSD) (Wilmington, DE, USA) operated either in the electron ionization (EI) mode (70 eV) or negative chemical ionization (NCI) mode, depending on the class of target analytes. Low molecular-weight PCB congeners (PCB8, 18, 28, 37, 44, 49, and 52) and PAHs were analyzed by GC/EI-MS. All remaining target PCB congeners, OCPs, PBDEs, pyrethroids, and fipronil and its three degradates were analyzed by GC/NCI-MS. An Agilent J&W DB-XLB (30 m × 0.25 mm × 0.25 μm) column was used to separate target analytes. The GC inlet was held isothermal at 300ºC in splitless mode. The carrier gas was ultrahigh purity (>99.999%) helium with a constant flow rate of 1 ml/min (GC/EI-MS) or 1.9

ml/min (GC/NCI-MS). For GC/EI-MS analyses, the oven temperature was programmed from 80°C (1 min hold), ramp to 190°C at 5°C/min, ramp to 260 °C at 4°C/min, ramp to 290°C at 20°C/min, and ramp to 300°C at 50°C/min (20 min hold). The transfer line, ion source and quadrupole were maintained at 280, 230, and 150 °C, respectively. For GC/NCI-MS analyses, methane (99.97% purity) was the reagent gas at 40% flow rate. The oven temperature was programmed from 90°C (1 min hold), ramp to 150°C at 5°C/min, ramp to 260 °C at 3°C/min, and ramp to 320°C at 20°C/min (5 min hold). The transfer line, ion source and quadrupole were maintained at 280, 150, and 150 °C, respectively. Mass spectral data was collected using the selected ion monitoring (SIM) mode, and quantitation of target analytes employed multi-point internal standard calibration curves based on dilutions of authentic (high purity) standards.

2.1.2 Sediment Trace Metals

Samples were digested using nitric and hydrochloric acid in sealed Teflon vessels heated with a microwave system. Digestates were filtered, then diluted to a final volume of 250 ml, and an aliquot was spiked with internal standard and analyzed using an Agilent 7700 ICPMS equipped with a reaction/collision cell. All calibration curves were based on NIST traceable commercial standards using a minimum of 5 points.

2.1.3 Acid Volatile Sulfides (AVS) and Simultaneously Extracted Metals (SEM)

Samples were analyzed by acidification in a sealed container and then centrifuged. An aliquot of the water was then analyzed for total sulfides using a spectrophotometer. The calibration curve was based on a commercial standard using a minimum of 3 points. Simultaneously extracted metals were analyzed by taking an aliquot of the water which was spiked with internal standard and analyzed using an Agilent 7700 ICPMS equipped with a reaction/collision cell. All calibration curves were based on NIST traceable commercial standards using a minimum of 5 points.

2.1.4 Pore Water Metals

Pore water was extracted from the sediment by centrifuging the sample and removing the supernatant. The supernatant was then passed through a 0.45 µm nucleopore filter for measurement of dissolved metals. This sample was spiked with internal standard and analyzed using an Agilent 7700 ICPMS equipped with a reaction/collision cell. All calibration curves were based on NIST traceable commercial standards using a minimum of 5 points.

2.1.5 Sediment Total Organic Carbon

A pre-weighed aliquot of freeze dried sediment was treated with concentrated hydrochloric acid to remove inorganic carbon prior to analysis using a Shimadzu TOC-VCPH/CPN Total Organic Carbon (TOC) analyzer with SSM-5000A Solid Sample Module (Columbia, MD, USA). The CO_2 generated from the combustion of TOC was measured by an infrared detector (non-dispersive infrared analysis, NDIR). Quantitation of TOC was performed using a multi-point calibration curve with an acetanilide standard.

2.1.6 Sediment Grain Size

Samples were exposed to hydrogen peroxide to remove organic material then air dried. An aliquot of sediment was sieved using 1 mm and 2 mm screens to determine particle size in that

range and a separate aliquot of the sieved sediment was placed in a Micromeritics Particle Size Analyzer for <1 mm grain size distribution.

2.1.7 Contaminant summarization

Results of sediment analyses are reported as µg/kg dry weight (dw) for trace organics and as mg/kg for trace metals. For individual chemicals, one half of the detection limit was used for all non-detects.

Sums of organic contaminant classes (e.g., chlordanes, PAHs, PCBs, pyrethroids, and DDTs,) were calculated as the sum of all detected analytes within the class. In cases where all class components were non-detect for a sample, the sum value was represented by the highest detection limit of any of the class components.

Table 2-1. Chemical constituents measured for sediment samples.

Metals and General Constituents	Legacy and Priority Pollutants	Contaminants of Emerging Concern
Total Organic Carbon (TOC)	o.p'-DDT, p,p'-DDT	Polybrominated diphenyl ethers[c]
Grain Size	o.p'-DDD, p,p'-DDD	**Fipronil and degradates**
Aluminum (Al)	o.p'-DDE, p.p'-DDE	Fipronil
Arsenic (As)	Aldrin	Fipronil desulfinyl
Beryllium (Be)	Dieldrin	Fipronil sulfide
Cadmium (Cd)	Endrin	Fipronil sulfone
Chromium (Cr)	Chlorpyrifos	**Pyrethroid Pesticides**
Copper (Cu)	Chlordene	Bifenthrin
Iron (Fe)	DDMU	Cyfluthrin
Lead (Pb)	Heptachlor epoxide B	Cypermethrin
Manganese (Mn)	Cis-, trans-nonachlor	Deltamethrin
Mercury (Hg)	cis-, trans-chlordane	Esfenvalerate
Nickel (Ni)	oxychlordane	Fenpropathrin
Selenium (Se)	Polycyclic Aromatic Hydrocarbons (PAHs)[a]	Lamda-Cyhalothrin
Silver(Ag)	Polychlorinated Biphenyls (PCBs)[b]	Permethrin
Tin (Sn)		
Titanium (Ti)		
Vanadium (V)		
Zinc (Zn)		

Notes:
[a]Includes Acenaphthene, Acenaphthylene, Anthracene, Benz[a]anthracene, 9,10-Diphenylanthracene, Biphenyl, Chrysene, Fluoranthene, Benzo[b]fluoranthene, Benzo[k]fluoranthene, Fluorene, 11H-Benzo[b]fluorine, Naphthalene, 1-Methylnapthalene, 2-Methylnapthalene, Perylene, Benzo[g,h,i]perylene, 2,6-Dimethylnaphthalene, 2,3,5-Trimethylnaphthalene, Phenanthrene, 1-Methylphenanthrene, 2-Methylphenanthrene, 3,6-Dimethylphenanthrene, Pyrene, Benzo[a]pyrene, and Benzo[e]pyrene.

[b]Includes congeners: PCB-8, 18, 28, 37, 44, 49, 52, 66, 70, 74, 77, 81, 87, 99, 101, 105, 110, 114, 118, 119, 123, 126, 128, 138, 149, 151, 153/168, 156, 157, 158, 167, 169, 170, 177, 180, 183, 187, 189, 194, 200, 201, and 206.

[c]Includes congeners PBDE-15, 28, 33, 47, 49, 66, 75, 99, 100, 153, 154, 155, and 183.

2.2 Toxicity Analysis

2.2.1 Whole sediment

Sediment toxicity was assessed using the 10 d amphipod survival test (USEPA 1994, ASTM 2010) with *Eohaustorius estuarius*. Approximately 150 ml of sediment from each station was added to 1 L glass jars to a depth of 2 cm with approximately 800 ml of 32 g/kg salinity overlying water. After a 24 hr equilibration period, 20 juvenile amphipods were randomly added to each replicate. The test chambers were aerated and subjected to a 24 hr light cycle to encourage burrowing behavior, maximizing sediment exposure. At the end of the 10 d exposure period, the sediment from each jar was passed through a 0.5 mm sieve and the surviving amphipods were enumerated. The percentage of surviving amphipods was the test endpoint.

2.2.2 Pore water

Pore water toxicity screening was completed for the round 2 sediment samples only. Sediment pore water was extracted by centrifugation at 3,000 x g for 30 minutes and stored overnight at 4ºC in the dark. Pore water toxicity tests were conducted in 22 ml glass shell vials, containing 10 ml of sample and 5 amphipods (*E. estuarius*) for each replicate vial. Laboratory control water consisted of 45 µm-filtered seawater diluted to 32 g/kg salinity using deionized water. The exposure was conducted in the dark and did not receive aeration. At 4 d and 7 d, the amphipods were enumerated in their test chamber. At the end of the 10 d exposure period, the surviving amphipods were removed from the test chambers and enumerated. The percentage of surviving amphipods was the test endpoint.

2.2.3 Sediment-water interface

Toxicity at the sediment-water interface was evaluated with the mussel embryo development test using *Mytilus galloprovincialis* following established methods (USEPA 1995, Anderson et al. 1996). Sediment was added to 600 ml tall form beakers to a depth of 5 cm with approximately 300 ml of 32 g/kg overlying water. Gentle aeration was added through the use of capillary tubing and the system was equilibrated for approximately 24 hr at 15°C.

After equilibration, a polycarbonate tube with a 37 µm mesh screen near its bottom was placed on the sediment surface so that the screen sat just above the sediment surface (Figure 2-1). The mussels were induced to spawn by thermal shock, the gametes were collected, and the eggs were fertilized. Approximately 200 fertilized eggs were added to each screen tube. The eggs were also added to a set of positive control vials containing a range of copper concentrations as a reference toxicant. The control vials from the reference toxicant were used to determine the actual number of eggs delivered at test initiation. The embryos were given 48 hr to develop, then the screen tubes were removed, the embryos rinsed into glass shell vials, and preserved for microscopic examination. All normal embryos from each replicate were counted. The number of normal embryos from each replicate was then divided by the number added at the start to calculate the endpoint of percent normal-alive; with the assumption that any missing embryos had died and decomposed.

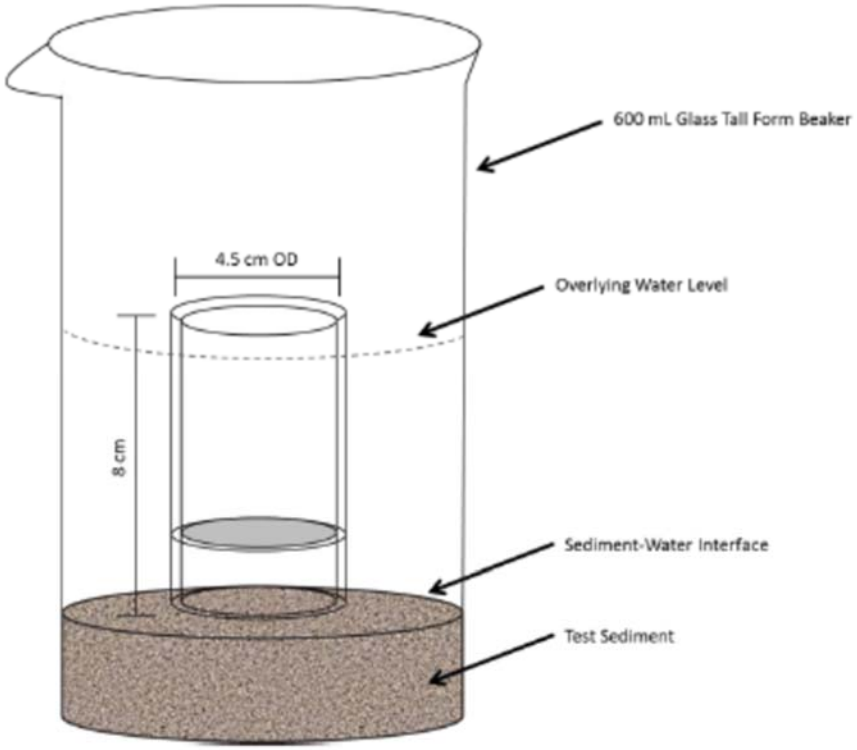

Figure 2-1. Experimental setup for the sediment-water interface test.

2.3 Benthic Analysis

2.3.1 Sample Processing

Benthic samples from each site were collected and processed following the Bight '13 Southern California Bight Regional Monitoring Survey Field Operations Manual (Bight '13 Field Sampling and Logistics Committee 2013) and Bight '13 Macrobenthic Sample Analysis Laboratory Manual (Bight '13 Benthic Committee 2013). In short, sediments were collected with a 0.1 m^2 Van Veen grab and sieved on a 1 mm screen. Material retained on the screen was placed in a chemical relaxant solution and then fixed with 10% buffered formalin. Samples were rinsed and transferred from formalin to 70% ethanol 7 days after collection. Sorting, identification, and enumeration of the fauna in the samples were done by Aquatic Bioassay and Consulting laboratories, Ventura, CA.

2.3.2 Sample QA/QC

QA/QC protocols and data quality objectives for sample sorting, identification, and enumeration are detailed in the Bight '13 Macrobenthic Sample Analysis Laboratory Manual (Bight '13 Benthic Committee 2013) and summarized in Table 2-2. All of the samples met the sorting efficiency DQO. Of the three taxonomic identification QA/QC metrics, identification accuracy (98.2%) and count accuracy (94.6%), met their DQO. The samples did not pass their taxonomic accuracy measure (81.8%), but this is not uncommon for low richness/abundance samples (QA/QC sample had 9 taxa, which was resolved to 11). As a corrective action, the appropriate taxa names were fixed throughout all of the samples where applicable. After correction, all of the data were used in subsequent SQO and Benthic Community Stressor Identification analyses.

Table 2-2. QA/QC Equations and DQOs for benthic macrofaunal samples.

QA/QC Metric	Calculation	DQO
Sorting Efficiency	100 * {#original / [#original + (#resort / aliquot fraction)]}	95%
Identification Accuracy	[1 – (# Individuals Mis-ID'd / # Individuals Resolved)] *100	90%
Taxa Discrimination	{1 – [|(# Taxa Resolved − # Taxa Original)| / # Taxa Resolved]} *100	90%
Count Accuracy	{1 – [|(# Individuals Original - #Individuals Resolved)| / # Individuals Resolved]} *100	90%

3. SEDIMENT QUALITY SURVEY

3.1 Sample Collection and Processing

Sampling occurred on July 27 and 28, 2016 at ten stations in the harbor, including four stations in the back basins, four stations in the front basins, and two stations in the main channel (Figure 3-1, Table 3-1). Two of the ten stations were repeated from the January sampling and TIE study (denoted by the purple ovals) and were not included in the SQO analysis. A field duplicate sediment sample was collected at one station (S7) and analyzed for all parameters; results for this duplicate are included in this summary.

Sediment samples were collected using a modified double Van Veen grab, with multiple grabs taken at each location to collect the necessary sediment volume. Sediment for toxicity testing and chemical analysis was taken from the top 5 cm layer of multiple grab samples. Sediment from these grabs was combined and homogenized in the field prior to distribution to the appropriate containers. For the benthic community analysis, the entire contents of a single grab were passed through a 1 mm sieve. The material trapped on the screen was preserved with formalin then transferred to alcohol prior to sorting and organism identification.

The whole sediment toxicity and sediment-water interface samples were stored at 4 °C and passed through a 2 mm sieve prior to use to remove indigenous organisms and debris. Sediment for chemical analysis was stored at -20 °C and was not sieved prior to analysis.

Sediment from a reference site offshore of Dana Point was collected on September 28, 2015. This sediment has a fine grain size composition that is similar to the sediment characteristics in Marina del Rey Harbor, and has been found to be very low in contaminants. This sediment was stored at 4 °C and tested for toxicity multiple times during holding and consistently exhibited greater than 90% survival in *E. estuarius* exposures. It was used as a second reference for toxicity assessment, in addition to the amphipod home sediment. The Dana Point sediment was collected and stored in the same manner as the MdRH sediment.

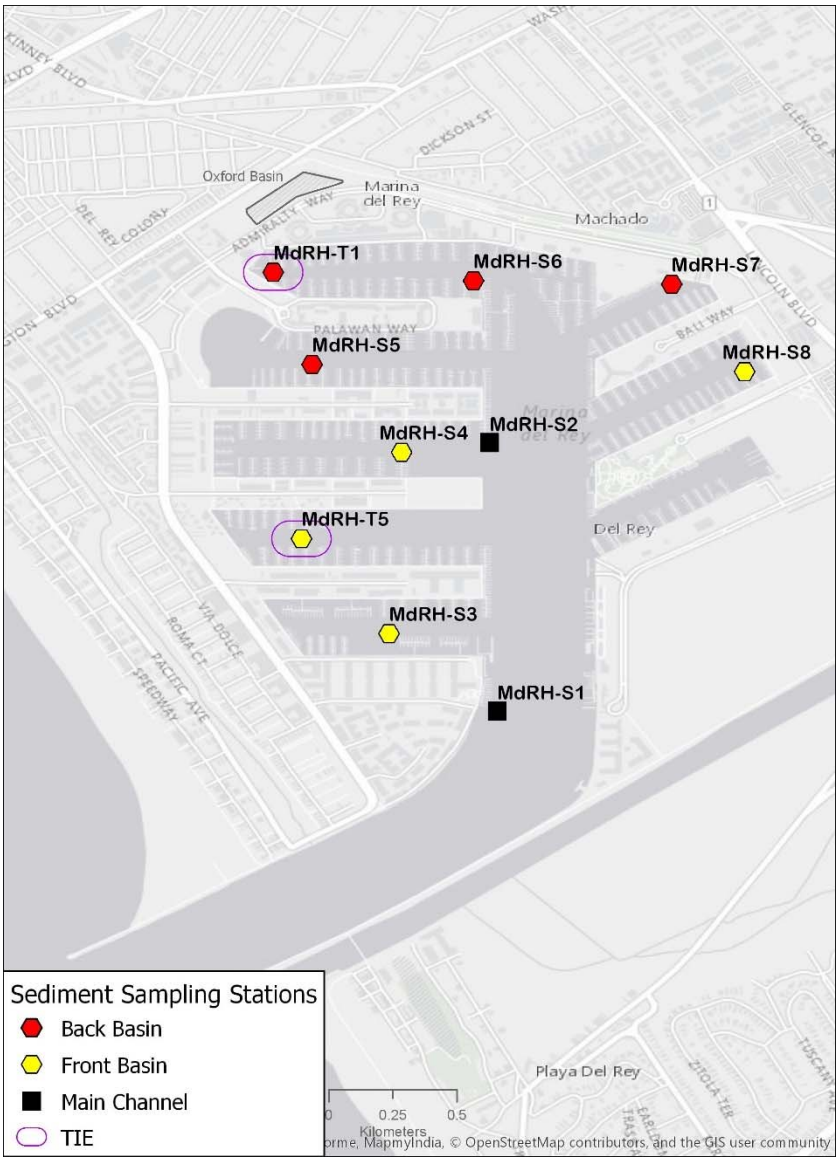

Figure 3-1. Sampling sites for sediment quality assessment.

Table 3-1. Station details for samples collected for the Marina del Rey harbor sediment quality assessment study.

Station	Date	Time	Latitude	Longitude	Depth (meters)	Distance to Target (meters)
MdRH-S1	28-Jul-16	1157	33.97002	-118.44930	6.2	22.0
MdRH-S2	27-Jul-16	1702	33.97791	-118.44953	5.8	12.0
MdRH-S3	28-Jul-16	1031	33.97229	-118.45248	4.2	16.0
MdRH-S4	28-Jul-16	0758	33.97762	-118.45211	4.3	11.0
MdRH-S5	27-Jul-16	1559	33.98020	-118.45474	4.9	2.0
MdRH-S6	27-Jul-16	1349	33.98266	-118.45000	4.0	8.0
MdRH-S7	27-Jul-16	1118	33.98256	-118.44418	4.0	6.5
MdRH-S7-Dup	27-Jul-16	1220	33.98256	-118.44418	4.0	6.5
MdRH-S8	27-Jul-16	0929	33.97999	-118.44204	4.2	2.0

3.2 Data Analysis and Interpretation

3.2.1 SQO Toxicity Assessment

Calculation of assessment categories for the toxicity line of evidence (LOE) was accomplished following SQO guidelines (Bay et al. 2014). This calculation entails control adjusting the toxicity endpoints and testing for statistical differences from the controls. Statistical differences to the control were tested using an unequal variance t-test. This information was then compared to SQO thresholds (Table 3-2). The results for each of the two toxicity tests were then integrated by averaging the category scores and rounding up to the next highest category if the average fell between two categories.

Table 3-2. Sediment toxicity response classification ranges.

Test Species/Endpoint	Nontoxic (Percent)	Low Toxicity (Percent of Control)	Moderate Toxicity (Percent of Control)	High Toxicity (Percent of Control)
Eohaustorius Survival	90 to 100	82 to 89[a]	59 to 81[b]	< 59
Mytilus Normal Development	80 to 100	77 to 79[a]	42 to 76[b]	< 42

[a] If the response is not significantly different from the negative control, then the response is classified as Nontoxic.
[b] If the response is not significantly different from the negative control, then the response is classified as Low Toxicity.

3.2.2 SQO Chemical Assessment

Calculation of assessment categories for the chemistry LOE was accomplished following California SQO guidelines (Bay et al. 2014). The chemistry LOE includes the calculation of two indices. The California Logistic Regression Model (CA LRM) is based on the relationship between chemical concentration and sediment toxicity. The index score is then compared to a series of thresholds to determine the exposure category (Table 3-3). The Chemical Score Index (CSI) is based on the relationship between chemical concentration and benthic community disturbance. The index score is then compared to a series of thresholds to determine the exposure category (Table 3-4). The two indices use a subset of the total chemical analyte list (Table 3-5). The results for each of the two chemical indices are integrated by averaging the category scores and rounding up to the next highest category if the average falls between two categories.

A set of rules for non-detected chemicals and the summation of chemical classes listed in Table 3-5 were followed, consistent with the SQO guidelines (Bay et al. 2014). For individual chemicals that were not detected, one half the detection limit was used in the index calculations. For the summations, only the individual chemicals that were detected were used in the summation. If none of the chemicals within an individual summation were detected, then the highest detection limit for any chemical in that class was used in the index calculations.

Table 3-3. Response ranges of Pmax for determination of the CA LRM category score.

Category	Range	Category Score
Minimal Exposure	<0.33	1
Low Exposure	≥0.33 - 0.49≤	2
Moderate Exposure	>0.49 - 0.66≤	3
High Exposure	>0.66	4

Table 3-4. Response ranges for CSI calculation.

Category	Range	Category Score
Minimal Exposure	<1.69	1
Low Exposure	≥1.69 - 2.33≤	2
Moderate Exposure	>2.33 - 2.99≤	3
High Exposure	>2.99	4

Table 3-5. Chemical constituents required for SQO assessment.

Metals	Low Molecular Weight PAHs
Cadmium (mg/kg)	Acenaphthene (µg /kg)
Copper (mg/kg)	Anthracene (µg /kg)
Lead (mg/kg)	Phenanthrene (µg /kg)
Mercury (mg/kg)	Biphenyl (µg /kg)
Zinc (mg/kg)	Naphthalene (µg /kg)
	2,6-dimethylnaphthalene (µg/kg)
Organochlorine Pesticides	Fluorene (µg/kg)
Alpha Chlordane (µg /kg)	1-methylnaphthalene (µg /kg)
Gamma Chlordane (µg /kg)	2-methylnaphthalene (µg /kg)
Trans Nonachlor (µg /kg)	1-methylphenanthrene (µg /kg)
Dieldrin (µg /kg)	
o,p'-DDE (µg /kg)	**High Molecular Weight PAHs**
p,p'-DDE (µg /kg)	Benzo(a)anthracene (µg /kg)
o,p'-DDD (µg /kg)	Benzo(a)pyrene (µg /kg)
p,p'-DDD (µg /kg)	Benzo(e)pyrene (µg /kg)
o,p'-DDT (µg /kg)	Chrysene (µg /kg)
p,p'-DDT (µg /kg)	Dibenz(a,h)anthracene (µg /kg)
	Fluoranthene (µg /kg)
Polychlorinated Biphenyls[1]	Perylene (µg /kg)
	Pyrene (µg /kg)

[1]Includes PCB congers 8, 18, 28, 44, 52, 66, 101, 105, 110, 118, 128, 138, 153, 180, 187, 195.

3.2.3 SQO Benthic Community Assessment

The SQO benthic community line of evidence is determined using a set of four benthic indices (Bay et al. 2014). Results from the four indices are then integrated to get the final benthic community line of evidence. Integration is achieved by calculating the median of the category scores of the four indices. If the median falls between two categories, the value is rounded up. Brief descriptions of each index are below.

Index of Biotic Integrity (IBI)

The IBI evaluates the ranges of four metrics that would be expected under reference conditions. Each metric that is outside of the reference range increases the IBI score by one (Table 3-6). If all four of the metrics are inside the reference range, the score is zero. If all four are outside of the reference range, then the IBI value is four. The index score is then compared to a series of thresholds to determine the level of community disturbance (Table 3-7).

Table 3-6. Reference ranges for IBI metrics.

Metric	Reference Range
Total Number of Taxa	13 - 99
Number of Mollusc Taxa	2 - 25
Abundance of *Notomastus sp.*	0 - 59
Percentage of Sensitive Taxa	19 - 47.1

Table 3-7. IBI category response ranges.

IBI Score	Category	Category Score
0	Reference	1
1	Low Disturbance	2
2	Moderate Disturbance	3
3 or 4	High Disturbance	4

Relative Benthic Index (RBI)

The RBI is the weighted sum of three descriptors of community health: 1) four metrics describing biodiversity (total number of taxa, number of crustacean taxa, abundance of crustacean individuals, and number of mollusk taxa); 2) the abundances of three positive indicator taxa; and 3) the presence of two negative indicator species. The calculated value for the RBI is then compared to a series of thresholds to determine the level of community disturbance (Table 3-8).

Table 3-8. RBI category response ranges.

RBI Score	Category	Category Score
>0.27	Reference	1
>0.16 to ≤0.27	Low Disturbance	2
>0.08 to ≤0.16	Moderate Disturbance	3
≤0.08	High Disturbance	4

Benthic Response Index (BRI)

The BRI is the abundance weighted pollution tolerance score for the benthic organisms present in the sample. The greater the BRI score, the more degraded the benthic community. The BRI score is compared to a series of thresholds to determine the level of benthic community impairment (Table 3-9).

Table 3-9. BRI category response ranges.

BRI Score	Category	Category Score
<39.96	Reference	1
≥39.96 to <49.15	Low Disturbance	2
≥49.15 to <73.27	Moderate Disturbance	3
≥73.27	High Disturbance	4

River Invertebrate Prediction and Classification System (RIVPACS)

The RIVPACS index represents the ratio of the number of reference taxa present compared to the number of expected reference taxa from the same habitat. Calculation of the expected number of taxa is based on the station's depth, latitude, and longitude in a complex linear discriminant function. The index value is then compared to a series of thresholds to determine the level of benthic community disturbance (Table 3-10).

Table 3-10. RIVPACS category response ranges.

RIVPACS Score	Category	Category Score
>0.90 - <1.10	Reference	1
>0.74 - ≤0.90 or ≥1.10 - <1.26	Low Disturbance	2
>0.32 - ≤0.74 or ≥1.26	Moderate Disturbance	3
≤0.32	High Disturbance	4

3.2.4 SQO Integrated Assessment

Results from the three SQO lines of evidence (toxicity, chemistry, and benthic community) are integrated by comparing their categories to a matrix of possible outcomes (Appendix B, Bay et al. 2014). The integrated assessment classifies each station into one of five categories: Unimpacted, Likely Unimpacted, Possibly Impacted, Likely Impacted, or Clearly Impacted. Stations falling into the Unimpacted or Likely Unimpacted categories are deemed to have met the State's sediment quality objective for protection of the benthic community.

3.3 Results and Discussion

3.3.1 Sampling

All samples were collected as planned with no obstructions encountered at any site. All stations were sampled within 25 m of the target coordinates (Table 3-1). All samples were transported to the laboratories and analyzed within designated holding times.

3.3.2 Sediment toxicity

No toxicity was observed at any station for the amphipod survival test. The lowest survival at any station was 99% of that observed in the control (Figure 3-2). Consequently, the amphipod test results for all samples were classified as Nontoxic (Table 3-11).

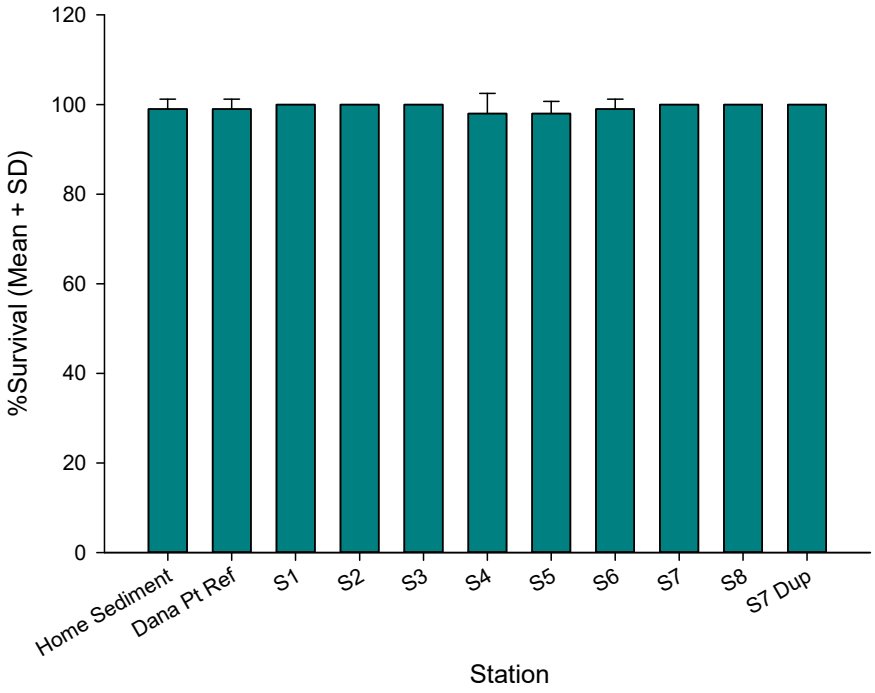

Figure 3-2. Mean amphipod survival in sediment samples collected from Marina del Rey Harbor in July 2016.

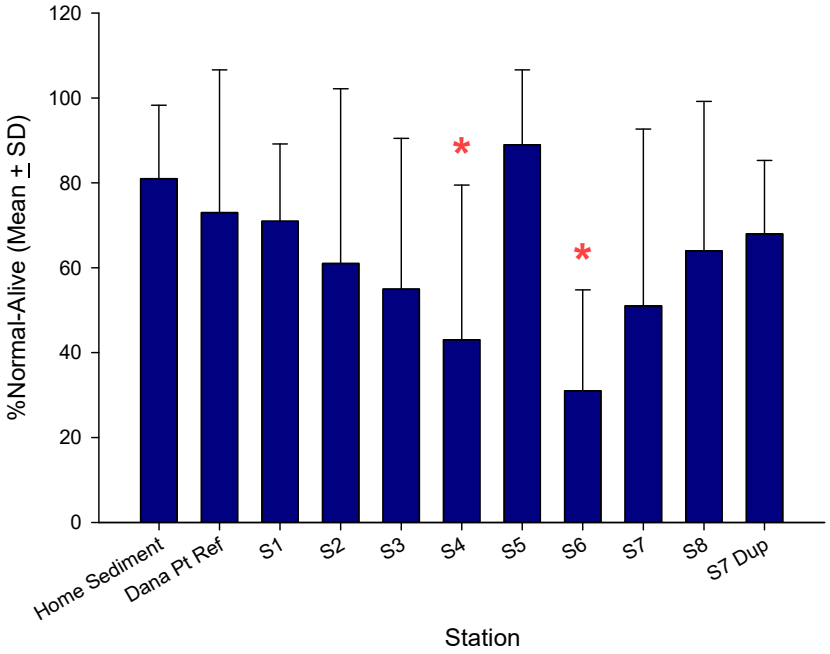

Figure 3-3. Mean mussel embryo percent normal-alive sediment-water interface samples tested from Marina del Rey Harbor in July 2016. The asterisks indicate samples that were significantly different from the control (home sediment), unequal variance t-test ($p \leq 0.05$).

Results from the sediment-water interface test indicated considerable variability between replicates for all samples. However, statistically significant toxicity relative to the controls was present at only two stations: S4 and S6 (Figure 3-3). In addition to evaluating the data for significant differences from control, the toxic response for each station was also categorized according to the SQO thresholds listed in Table 3-2. The test response was normalized as a percentage of the control response and then compared the test response thresholds. Three stations were classified as Low Toxicity (S2, S3, and S7), while one each were in the Moderate Toxicity (S4) and High Toxicity categories (S6) categories (Table 3-11).

The integrated toxicity LOE results (based on both toxicity tests) indicated that four stations were in the Low Toxicity category and one was in the Moderate Toxicity category. All of the stations that were classified as toxic were driven by the SWI test, given that the amphipod test found no toxicity.

Table 3-11. SQO categories for each of the toxicity tests and the integrated toxicity line of evidence assessment category.

Station	Amphipod SQO Category	SWI SQO Category	Integrated Toxicity SQO Category
MdRH-S1	Nontoxic	Nontoxic	Nontoxic
MdRH-S2	Nontoxic	Low Toxicity	Low Toxicity
MdRH-S3	Nontoxic	Low Toxicity	Low Toxicity
MdRH-S4	Nontoxic	Moderate Toxicity	Low Toxicity
MdRH-S5	Nontoxic	Nontoxic	Nontoxic
MdRH-S6	Nontoxic	High Toxicity	Moderate Toxicity
MdRH-S7	Nontoxic	Low Toxicity	Low Toxicity
MdRH-S7-Dup	Nontoxic	Nontoxic	Nontoxic
MdRH-S8	Nontoxic	Nontoxic	Nontoxic

3.3.3 Sediment chemistry

The concentrations of the SQO chemical analytes are listed in Table 3-12. In some cases, these values represent non–detected chemicals and follow the rules set forth in the Methods section. Concentrations for individual chemicals comprising a contaminant class (e.g., PCBs) are listed in Appendix A.

The CA LRM index classified all of the stations into the High Exposure category (Table 3-13). In all cases, this was driven by the concentration of zinc or copper observed in the sediment. The CSI index classified all but one of the stations into the Moderate Exposure category. Station S7 was classified into the High Exposure category, while the field duplicate for S7 was classified as Moderate Exposure. However, the actual index scores for the two S7 samples were very similar, 3.12 and 2.96 respectively. The CSI score for most of the stations in the Moderate Exposure category was close to the threshold between the Moderate and High categories of 3.00.

Since the CA LRM index results were all in the High Exposure category and the CSI index were all either Moderate or High Exposure, the integrated chemical line of evidence was High Exposure for all the stations (Table 3-13).

Table 3-12. Sediment contaminant concentrations for constituents used in SQO calculation.

Category	Constituent	S1	S2	S3	S4	S5	S6	S7	S8	S7-dup
Organic contaminants (µg/kg)	α Chlordane	5.82	0.953	2.5	1.2	0.641	1.1	1.37	1.05	1.26
	γ Chlordane	7.96	1.39	3.32	1.61	0.857	1.27	2.96	1.39	1.58
	Dieldrin	1.24	0.191[b]	0.678	0.191[b]	0.191[b]	0.191[b]	0.191[b]	0.191[b]	0.191[b]
	Trans Nonachlor	5.68	1.02	2.45	1.21	0.685	0.962	1.58	1.11	1.21
	ΣDDDs	20.56	6.5	7.73	6.34	3.09	11.61	37.11	30.47	10.41
	ΣDDEs	52.52	24.54	41.15	28.75	26.70	44.14	72.37	81.84	49.42
	ΣDDTs	5.1[b]	5.1[b]	5.1[b]	5.1[b]	5.1[b]	5.1[b]	5.1[b]	5.1[b]	5.1[b]
	4,4'-DDT	5.1[b]	5.1[b]	5.1[b]	5.1[b]	5.1[b]	5.1[b]	5.1[b]	5.1[b]	5.1[b]
	ΣHMW PAHs	1036.5	223.9	482.6	373.5	254.3	296.4	747.4	648.4	361.9
	ΣLMW PAHs	28.9	22.2	22.2	22.2	22.2	22.2	19.6	17.4	22.2
	ΣPCBs[a]	105.7	52.5	93.8	75.2	68.1	73.1	85.2	84.4	85.7
Metal contaminants (mg/kg)	Cd	1.08	0.38	0.42	0.36	0.40	0.51	0.38	0.39	0.42
	Cu	206	211	332	356	554	617	544	405	508
	Pb	111	54.9	109	81.6	83.1	94.8	100	108	95.6
	Hg	0.43	0.37	0.55	0.58	0.81	0.69	0.76	0.75	0.71
	Ni	36.2	25.0	38.9	36.7	38.3	44.4	42.4	43.1	40.8
	Zn	391	280	424	405	502	607	520	463	481

[a]The sum of the SQO list of PCB congeners was multiplied by 1.72 to approximate the larger NOAA list.

[b]The result was non-detect. Half of the MDL value is shown.

Table 3-13. Sediment quality objectives chemistry LOE index scores and categories.

	Station ID								
Parameter	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**	**S7-dup**
CA LRM value	0.76	0.69	0.77	0.76	0.82	0.84	0.82	0.80	0.79
CA LRM category	High	High	High	High	High	High	High	High	High
CSI value	2.91	2.54	2.96	2.86	2.91	2.91	3.12	2.93	2.96
CSI category	Mod.	Mod.	Mod.	Mod.	Mod.	Mod.	High	Mod.	Mod.
Integrated category	High	High	High	High	High	High	High	High	High

3.3.4 Benthic Community Condition

The SQO categorical scores for the benthic LOE ranged from Low to High Disturbance among the combinations of stations and indices (Table 3-14). None of the indices indicated a Reference condition at any of the stations. The IBI and BRI classified the stations as having either Low or Moderate Disturbance, while the RBI and RIVPACS indices classified most stations in the High Disturbance category.

Integration of the results from the four indices indicated that all stations were in the Moderate or High Disturbance categories (Table 3-14). Stations S1, S2, and S8, were in the Moderate category with the remainder of the stations being High.

Table 3-14. Benthic community SQO index scores and disturbance categories.

	Station ID								
Parameter	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S7-dup**	**S8**
IBI Score	2	1	1	1	2	2	2	1	1
IBI Category	Moderate	Low	Low	Low	Moderate	Moderate	Moderate	Low	Low
RBI Score	0.046	0.087	0.074	0.056	0.040	0.046	0.050	0.058	0.049
RBI Category	High	Moderate	High	High	High	High	High	High	High
BRI Score	41.72	47.71	60.73	65.83	63.54	64.71	66.92	57.35	66.83
BRI Category	Low	Low	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate	Moderate
RIVPACS Score	0.328	0.222	0.129	0.259	0.320	0.283	0.286	0.286	0.425
RIVPACS Category	Moderate	Moderate	High	High	High	High	High	High	High
Integrated category	Moderate	Moderate	High	High	High	High	High	High	Moderate

3.3.5 Integration of Lines of Evidence

Integration of all three lines of evidence indicated that with the exception of station S6, which fell into the Clearly Impacted category, all of the stations were identified as being Likely Impacted (Table 3-15). None of the stations met the SQO criterion for protection of benthic community in sediments, defined as a classification of Unimpacted or Likely Unimpacted (Appendix B, SWRCB 2009). Given that the toxicity LOE categories were mostly Nontoxic or Low Toxicity, the overall assessment is largely driven by the chemistry and benthic community LOE results.

Table 3-15. Individual SQO lines of evidence (LOE) and integrated sediment assessment categories.

Station ID	Toxicity LOE	Chemistry LOE	Benthic LOE	Assessment Category
MdRH-S1	Nontoxic	High	Moderate	Likely Impacted
MdRH-S2	Low	High	Moderate	Likely Impacted
MdRH-S3	Low	High	High	Likely Impacted
MdRH-S4	Low	High	High	Likely Impacted
MdRH-S5	Nontoxic	High	High	Likely Impacted
MdRH-S6	Moderate	High	High	Clearly Impacted
MdRH-S7	Low	High	High	Likely Impacted
MdRH-S7 Dup	Nontoxic	High	High	Likely Impacted
MdRH-S8	Nontoxic	High	Moderate	Likely Impacted

3.3.6 Comparison to Previous studies

Stations were sampled in MdRH for both the Bight'08 and Bight'13 regional monitoring surveys and assessed for sediment quality using the same methods as for the current study. Results among surveys were compared on the basis of the percentage area classified under each of the SQO assessment categories. For the 2016 survey, 100% of the area did not meet the SQO (Tables 3-16 and 3-17). In Bight '08, 20% of the area met the SQO threshold (Unimpacted and Likely Unimpacted categories), while 25% of the area met the SQO in Bight '13. Variation among surveys in the percentage area meeting the SQO may not be meaningful due to the small number of stations sampled in each survey and random variation in station locations (Table 3-17). Compilation of the results from all three SQO assessments shows that sediment quality is similar in the front and back basins of the harbor, with some indications of relatively better sediment quality in the main channel (Figure 3-4).

Table 3-16. Percent area of Marina del Rey Harbor sediment classified into each California Sediment Quality Objectives assessment category.

	Unimpacted	Likely Unimpacted	Possibly Impacted	Likely Impacted	Clearly Impacted
Survey	%Area	%Area	%Area	%Area	%Area
Current (n=8)	0	0	0	92	8
B13 (n=4)	0	25	25	50	0
B08 (n=5)	0	20	40	20	20

Table 3-17. Individual SQO lines of evidence and integrated assessment of MdRH sediment samples from Bight'08 and Bight'13.

Station	Project	Location	Toxicity	Chemistry	Benthic	Integrated
6530[a]	B08	Back basin E	Nontoxic	High	Moderate	Likely Impacted
6527	B08	Front basin G	Moderate	High	Moderate	Clearly Impacted
6649	B08	Front basin C	Nontoxic	Moderate	Moderate	Possibly Impacted
6513[b]	B08	Lower main	Nontoxic	High	Low	Likely Unimpacted
6518[c]	B08	Middle main	Low	High	Low	Possibly Impacted
8417[a]	B13	Back basin E	Low	High	Moderate	Likely Impacted
8407[b]	B13	Lower main	Nontoxic	High	Reference	Likely Unimpacted
8409[c]	B13	Middle main	Nontoxic	Moderate	Moderate	Possibly Impacted
8413	B13	Middle main	Low	High	Moderate	Likely Impacted

Stations having the same superscript are the same location sampled in both surveys.

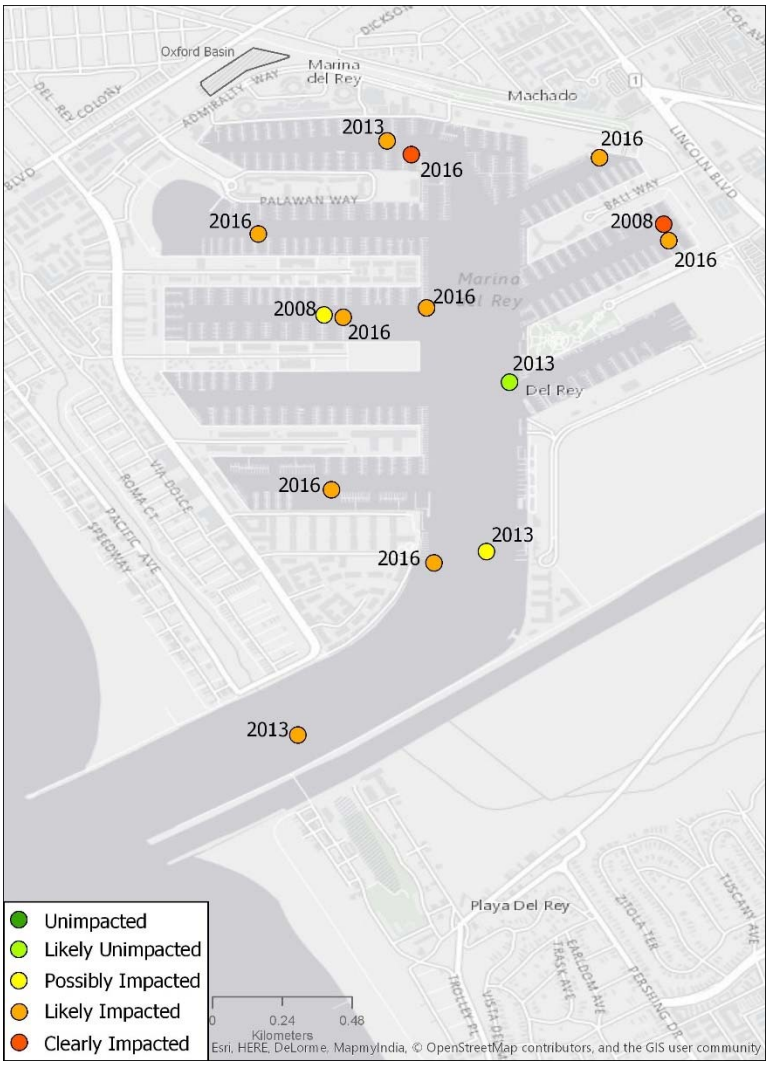

Figure 3-4. Integrated SQO Assessment categories for MdRH stations sampled in Bight'08, Bight'13, and July 2016. Data from Bight'13 are shown for the three repeated stations from Bight'08 as they are the more recent data.

3.4 Summary and Conclusions

Sediment quality in MdRH has changed little over the last eight years. For the Bight'08 stations that were resampled in 2013, the integrated assessment results were identical for both surveys. Additionally, the pattern seen in the current study was also noted for both Bight surveys where the lines of evidence largely driving the assessment categorization were chemistry which were in the Moderate and High categories, and benthic community which was in the Moderate category for all stations not located in the main channel.

4. TOXICITY IDENTIFICATION EVALUATION

Stressor identification for the MdRH sediments was conducted using a multiple lines of evidence approach. First, two toxic sediments from round 1 in January 2016 were evaluated by TIE treatment methods applied to sediments and pore water. Bioavailable metals and organics were also investigated for both sampling rounds. AVS-SEM measurements were used to evaluate the toxicity potential of metals from sediments. Sediment pore water metal concentration was used as a second method of investigating metal bioavailability in round 1 samples. Both of these methods provided direct measurements of metal bioavailability in the field samples. Equilibrium partitioning sediment benchmarks (ESB) were used to determine the toxicity potential of sediment trace organics at each station. The ESB approach uses water quality objectives to represent bioavailable pore water contaminant concentrations that are nontoxic. Partitioning relationships are then used to estimate the total sediment contaminant concentration (ESB) corresponding to the pore water objective. This value was then compared to the sediment concentrations in the field samples to determine the relative potential of trace organic contaminants to cause toxicity.

In addition to the TIE treatments and bioavailable contaminant evaluations, the MdRH sediments were compared to chemical-specific effect thresholds derived from spiked sediment studies as well as other field samples from embayments in the region to provide greater context for MdRH sediments. Spiked sediment studies are performed to evaluate the toxicity of specific contaminants, and can be used to compare the empirical LC_{50} values (sediment concentration at which 50% mortality is observed) to the sediment concentrations measured in MdRH. These values provide a sediment concentration at which toxicity is expected. In addition to laboratory studies, comparisons can be made to other field data for which sediment contaminant concentration and toxicity are known. Comparison to data from the 2013 Southern California Bight Regional Monitoring Program was used to evaluate the relative contamination level of MdRH sediments.

Lastly, reference element normalization was used to take into account the impact of local geology on the measured sediment metals. By determining the expected concentration of metals which naturally occur in the MdRH sediments, those values can be compared to the total metals measured. This provides more clarity on the extent of anthropogenic input of metals. Overall, this multiple line of evidence approach provides both direct evaluation of toxicity potential as well as expected toxicity based on other laboratory and field studies.

4.1 Sample Collection and Processing

The first round of sampling occurred on January 26, 2016 at five stations in the harbor, including three stations in the back basins and two stations in the front basins (Figure 4-1a, Table 4-1). The second round of sampling occurred on July 27 and 28, 2016 at ten stations in the harbor, including four stations in the back basins, four stations in the front basins, and two stations in the main channel (Figure 4-1b, Table 3-1). Two of the ten stations were repeated from the January sampling and TIE study (denoted by the purple ovals). One station (S7) was sampled twice as a field duplicate (S7-dup). Sediment samples were collected using a modified Van Veen grab, with multiple grabs taken at each location to collect the necessary sediment volume. Sediment samples for toxicity screening and chemical analysis was comprised of the top 5 cm layer of

multiple grab samples. These were combined and homogenized in the field prior to distribution to the appropriate containers. The toxicity screening sediment was stored at 4 ºC and passed through a 2 mm sieve prior to use to remove indigenous organisms and debris. Sediment for chemical analysis was stored at -20 ºC and was not sieved prior to analysis. Sediment collected for toxicity identification evaluation (TIE) was also comprised of the top 5 cm layer of multiple grab samples and stored at 4 ºC. Prior to use, the TIE sediment was passed through a 2 mm sieve followed by homogenization. The homogenized TIE sediment was also used for additional chemical analysis.

Table 4-1. Station details for samples collected for the Marina del Rey harbor sediment quality assessment study.

Station	Date	Time	Latitude	Longitude	Depth (meters)	Distance to Target (meters)
MdRH-T1	26-Jan-16	1025	33.98288	-118.45592	5.7	0.3
MdRH-T2	26-Jan-16	1154	33.98301	-118.45333	5.0	4.0
MdRH-T3	26-Jan-16	1345	33.98198	-118.44498	4.2	2.8
MdRH-T4	26-Jan-16	1456	33.98009	-118.44296	3.8	7.0
MdRH-T5	26-Jan-16	0822	33.97507	-118.45506	5.0	3.0

Sediment from a reference site offshore Dana Point was collected on September 28, 2015. It was collected and stored in the same manner as the MdRH sediment. Dana Point sediment has a fine grain size composition that is similar to the sediment characteristics in MdRH, and was used as a reference sediment. The amphipod home sediment control was collected in conjunction with the amphipods from Yaquina Bay, Newport, OR.

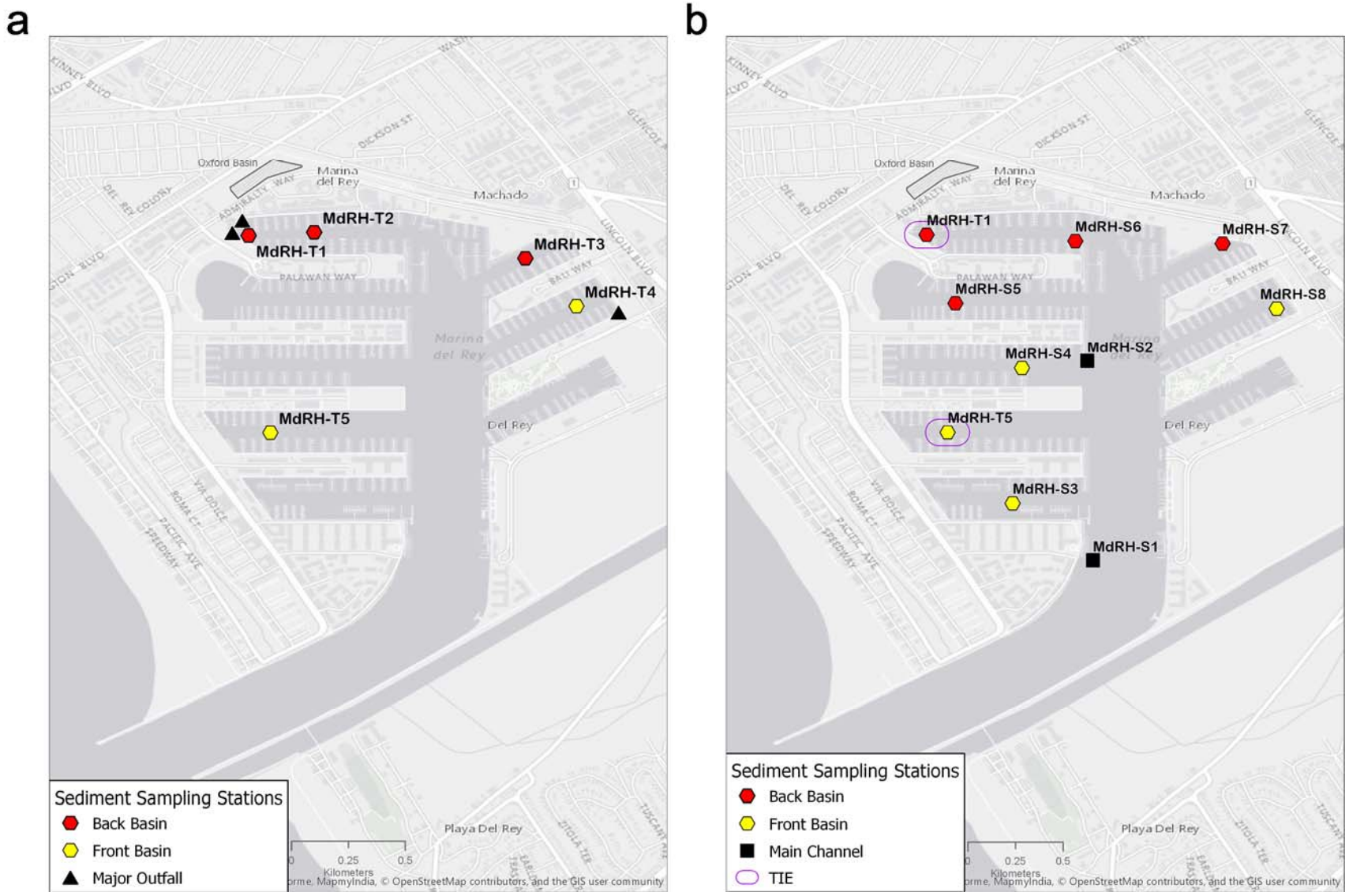

Figure 4-1. Sampling sites for January 2016 (a) and July 2016 (b) TIE events.

4.2 Characterization Treatments

Results from the amphipod survival tests in round 1 were used to select two sediment samples for TIE characterization (Stations T1 and T5), as well as the toxicity test species (*E. estuarius*). TIE characterization and identification tests were conducted on both whole sediment and pore water samples. Treatments designed to identify toxicity associated with nonpolar organic compounds, cationic metals, and ammonia were applied to each sample type (Tables 4-2 and 4-3).

4.2.1 Whole Sediment

Whole sediment TIE treatments included USEPA recommended methods to characterize the influence of ammonia, metals, and trace organics on toxicity (USEPA 2007). The treatments are manipulations of the sediment to modify toxicity in a predictable fashion (Table 4-2). The samples are then tested for toxicity after treatment and the patterns of change are diagnostic of chemical class (i.e., ammonia, metals, or organics).

TIE exposures occurred in 250 ml glass beakers with 50 ml of sediment and approximately 180 ml of overlying water. Ten amphipods were added to each beaker. The test chambers were aerated and placed under a 24 hr light cycle to encourage burrowing behavior, thereby maximizing sediment exposure. At the end of the 10 d exposure period, the sediment from each jar was passed through a 0.5 mm sieve and the surviving amphipods were enumerated. The percentage of surviving amphipods was the test endpoint.

TIE treatments included addition of charcoal to the sediment to characterize toxicity due to trace organics, addition of cation exchange resin (SIR 300) to characterize metal toxicity, and addition of zeolite to characterize toxicity due to ammonia. Additional treatments specific to pyrethroid pesticides were also applied (i.e., addition of piperonyl butoxide and carboxylesterase). The specific methods for this study followed those used for the 2013 Southern California Bight Regional Monitoring Survey. All treatments included blanks as a quality control step to demonstrate that the treatments themselves were not causing unintended toxicity. Blanks consisted of nontoxic sediment (amphipod home sediment) which was manipulated in the same manner as the test samples.

4.2.2 Pore Water

Sediment pore water extraction and test conditions were consistent with the toxicity screening methods. Pore water was subjected to various treatments (Table 4-3) in order to characterize the cause of toxicity. Characterization of toxicity caused by trace organics was accomplished by testing pore water that was passed through a C18 solid phase extraction column. Metals toxicity was characterized using two methods: addition of EDTA and addition of sodium thiosulfate. Characterization of ammonia-related toxicity was accomplished by extracting the pore water sample with a zeolite column. Additional treatments specific to pyrethroid pesticides were also applied. The specific methods for this study followed those used for the 2013 Southern California Bight Regional Monitoring Survey. All treatments included blanks. The blanks were laboratory control water that were manipulated in the same manner as the samples to ensure that the TIE treatments themselves did not cause toxicity.

Table 4-2. Treatments for whole sediment TIEs.

Treatment	Treatment Details	Purpose
Control	Home sediment	Verify test animal health
Baseline	Untreated sample	Comparison to treatments
Dilution control	20% home sediment (w/w)	Control for dilution and mixing effects associated with treatments
Carboxylesterase enzyme[1] (CEE)	1.0 Units/ml – Added every 48 hours	Breaks down pyrethroid pesticides to less toxic fractions
CEE blank	Home sediment + CEE	Identify reagent toxicity
Bovine serum albumin (BSA)	Match concentration to CEE enzyme addition	Additional CEE method control to account for effects due to nonspecific protein binding
Piperonyl butoxide (PBO)	400 µg/L	Renders organophosphorus pesticides non-toxic; increases toxicity of pyrethroid pesticides
PBO blank	Home sediment + PBO	Identify reagent toxicity
SIR 300 resin beads	20% (w/w)	Binding of cationic metals
SIR 300 blank	Home sediment + SIR 300	Identify reagent toxicity
Zeolite	20% w/w	Binding of ammonia
Zeolite blank	Home sediment + zeolite	Identify reagent toxicity
Coconut charcoal	15% (w/w)	Binding of non-polar organic contaminants
Charcoal blank	Home sediment + charcoal	Identify reagent toxicity

[1]Sigma-Aldrich brand from porcine liver (product number: E3019-20KU).

Table 4-3. Treatments for pore water TIEs.

Treatment	Treatment Details	Purpose
Control	Laboratory water	Verify test animal health
Baseline	Untreated sample	Comparison to treatments
EDTA	60 mg/L	Chelation of cationic metals (e.g. Zn, Cu)
EDTA blank	Control + EDTA	Identify reagent toxicity
Sodium thiosulfate	50 mg/L	Binding of some cationic metals
Thiosulfate blank	Control + Thiosulfate	Identify reagent toxicity
Zeolite column extraction	20 grams/column	Removal of ammonia
Zeolite blank	Extraction of control	Identify reagent toxicity
C18 column extraction[1]	6ml/1 gram column	Removal of non-polar organics
C18 column blank	Extraction of control	Identify reagent toxicity
Carboxylesterase enzyme (CEE)	1.0 Units/ml	Breaks down pyrethroid pesticides to less toxic fractions
CEE blank	Control + CEE	Identify reagent toxicity
Bovine serum albumin (BSA)	Match concentration to CEE enzyme addition	Additional CEE method control to account for effects due to nonspecific protein binding
Piperonyl butoxide (PBO)	200 µg/L	Renders organophosphorus pesticides non-toxic; increases toxicity of pyrethroid pesticides
PBO blank	Control + PBO	Identify reagent toxicity

[1]Supelco brand (Supelclean ENVI-18); 6ml/1gram capacity.

4.3 Data Analysis

4.3.1 Bioavailable Contaminant Analysis

In addition to total sediment metal concentration, the biologically available fraction of metal contaminants was determined. Although bulk sediment chemistry (C_{total}) may correspond to observed effects, a measure of the bioavailable concentration provides greater clarity in interpreting toxicity and bioaccumulation resulting from sediment contaminants.

AVS-SEM

Two methods were used to determine the bioavailability of metals in the sediments. The first involved the measurement of acid volatile sulfides (AVS) and simultaneously extracted metals (SEM) in the whole sediments. Sulfides in uncontaminated sediments are typically bound to iron or manganese. However, sulfides have a higher affinity to bind other metals such as nickel, zinc, cadmium, lead, copper, silver, and mercury and form insoluble compounds that are not biologically available to organisms (Di Toro et al. 1992). SEM is the total concentration of metals retrieved during the sulfide extraction process. When contaminated sediments contain these metals, the amount of metal that is unbound (bioavailable) is dependent on the amount of sulfides present. If the molar concentration of AVS is greater than metal, then it is unlikely that dissolved metals are causing toxicity in the sample.

Pore Water

The second method of determination of metals bioavailability was the direct measurement of dissolved metal concentrations in the round 1 sediment pore water. Pore water from centrifuged sediment was filtered to isolate the dissolved metal fraction. The concentration of dissolved metals was compared to water quality objectives listed in the California Toxics Rule (CTR) to indicate the potential for benthic community impacts. Dissolved metal concentrations below the CTR are unlikely to cause toxic impacts to benthic organisms.

Equilibrium Partitioning Sediment Benchmarks (ESB)

The equilibrium partitioning sediment benchmark (ESB) is based on EPA's ESB approach for sediment assessment and cleanup (USEPA 2008b, 2008a). These values are calculated using water quality criteria protective of aquatic life, in combination with equilibrium partitioning theory to determine the sediment contaminant concentration corresponding to a pore water concentration equivalent to the criterion. These benchmarks describe chemical-specific effect levels based on the bioavailable fraction of the contaminants.

ESBs were calculated for organic compounds using the equations as described in USEPA (2008a). The organic carbon-water partition coefficient (K_{OC}) of the compound is multiplied by a water based toxicity threshold, C_d.

$$ESB = K_{OC}\, C_d \qquad \text{Equation 1}$$

Where K_{OC} is expressed in liters per gram organic carbon and C_d equals the water based threshold expressed in micrograms per liter.

The K_{OC} values were derived from octanol-water partitioning coefficients, K_{OW} (Di Toro et al. 1991). K_{OW} values for individual compounds can be found in published scientific literature.

The K_{OC} value was calculated from:

$\text{Log } K_{OC} = 0.00028 + 0.983 \cdot (\log K_{OW})$ Equation 2

Once the K_{OC} is established, the final equation becomes:

$\text{ESB} = K_{OC}\ \text{L/kg}_{OC} * 0.001\ \text{kg}_{OC}/\text{g}_{OC} *$ Water Quality Threshold µg/L Equation 3

Where the middle term simply converts the units from a kilogram to a gram basis giving an ESB expressed as µg chemical per g organic carbon. ESBs were also expressed as a dry weight concentration for sediment with a typical TOC content of 1% for ease of comparison to monitoring data.

ESBs were calculated based on acute water quality thresholds, as these thresholds correspond to the acute basis of the 10-day amphipod survival sediment toxicity test. Acute water quality criteria were available for chlordanes, 4,4'-DDT, PAHs, total PCBs, and pyrethroids. These values and the calculated ESBs are summarized in Table 4-4. For PAHs, the acute water quality threshold was the Final Acute Value (FAV), which was calculated from the EPA Final Chronic Value (FCV) using an acute-to-chronic ratio (4.16) and dividing the result by 2. The measured sediment concentrations for each station (Appendices A and C) were normalized to organic carbon (Appendix D) and divided by the corresponding ESB to calculate the Toxic Units (TU) associated with the contaminant type. TUs indicate the relative toxic potential of a contaminant; values over 1 suggest risk to aquatic life.

For chlordanes, the measured cis-, trans-, and oxy-chlordane values were summed and compared to the ESB. For DDTs, only 4,4'-DDT was used in the TU calculation. For PAHs, the toxic units for the subset of 21 congeners listed in Table 4-3 were calculated, summed, and multiplied by a correction factor (3.596) to estimate TUs for all PAH congeners (USEPA 2003b). For PCBs, the total PCB sediment concentration was used. For pyrethroids, the toxic units were calculated for each compound and then summed.

Table 4-4. Acute water quality criteria and calculated ESB concentrations.

Class	Chemical	Acute water quality criteria (µg/L)	ESB (µg/g$_{OC}$)	ESB (µg/kg dw) at 1% TOC	Reference
Chlordanes	Chlordane	0.09	147	1,470	USEPA Aquatic Life Criteria: Saltwater CMC (chlordane and DDT)[1]
DDTs	4,4'-DDT	0.13	341	3,410	
PAHs	Acenaphthene	116	1021	10,210	USEPA 2003
	Acenaphthylene	638	940	9,400	(all PAHs)
	Anthracene	43.1	1235	12,350	
	Benzo(a)anthracene	4.63	1749	17,490	
	Benzo(a) pyrene	1.87	2012	20,120	
	Benzo(b) fluoranthene	1.41	2037	20,370	
	Benzo(k) fluoranthene	1.33	2038	20,380	
	Chrysene	4.25	1754	17,540	
	Fluoranthene	14.8	1472	14,720	
	Fluorene	81.7	1121	11,210	
	Naphthalene	402	801	8,010	
	Phenanthrene	39.8	1241	12,410	
	Pyrene	21.0	1451	14,510	
	Benzo(e) pyrene	1.87	2012	20,120	
	Benzo(ghi) perylene	0.913	2278	22,780	
	2,6-dimethylnaphthalene	53.6	1068	10,680	
	1-methylnaphthalene	157	927	9,270	
	2-methylnaphthalene	150	930	9,300	
	Perylene	1.87	2012	20,120	
	1-methylphenanthrene	15.6	1394	13,940	
	2,3,5-trimethylnaphthalene	20.4	1215	12,150	
PCBs	Total PCBs	10	15602	156,020	Oregon DEQ 2013[2]
Pyrethroids	Bifenthrin	0.8	633	6,330	USEPA OPP aquatic life benchmarks: acute invertebrates[3]
	Cyfluthrin	0.0125	8.63	86.3	
	Cypermethrin	0.21	646	6,460	(all pyrethroids)
	Deltamethrin	0.055	54.6	546	
	Esfenvalerate	0.025	0.214	2.140	
	Fenpropathrin	0.265	210	2,100	
	Lamda-Cyhalothrin	0.0035	21.2	212	
	Permethrin	0.0106	26.0	260	

[1]www.epa.gov/wqc/national-recommended-water-quality-criteria-aquatic-life-criteria-table
[2]Oregon Department of Environmental Quality, Effective Aquatic Life Criteria as of Jan 31, 2013. www.epa.gov/sites/production/files/2014-12/documents/orwqs_aquatic.pdf
[3]www.epa.gov/pesticide-science-and-assessing-pesticide-risks/aquatic-life-benchmarks-pesticide-registration

4.3.2 Spiked Sediment Toxicity Test Threshold Comparison

Many researchers have performed spiked sediment studies using *E. estuarius* and other marine amphipods, to determine the toxicity of a wide range of contaminants. The Spiked Sediment Toxicity Database (http://data.sccwrp.org/sedag) contains a compilation of these results from the literature. The lowest available marine invertebrate LC_{50} values in the database for chlordanes, DDTs, pyrethroids (bifenthrin, cyfluthrin, and permethrin), copper, lead, and zinc were used for comparison to MdRH sediment contaminant concentrations. The thresholds are summarized in Table 4-5. It should be noted that the LC_{50} for chlordane is an estimate based on a toxicity test that resulted in no toxicity at the highest concentration tested (2120 $\mu g/g_{oc}$), and therefore, is an underestimate of the actual LC_{50} value. The database did not contain sufficient data for PCBs and PAHs to enable this comparison to be conducted.

Similar to the ESB evaluation, TUs were calculated for each station/contaminant using the sediment contaminant concentrations and LC_{50} values. Organic contaminant concentrations were normalized to the organic carbon content of the sediment to minimize any variation in contaminant bioavailability based on total sediment concentrations. However, sediment metal concentrations were not TOC-normalized. These TU comparisons may not be representative of the bioavailable metal concentration and may not accurately estimate the toxicity potential if the spiked sediment and MdRH sediment had different characteristics (i.e., grain size, sulfides, organic carbon, etc.).

Table 4-5. LC_{50} values used for threshold comparison.

Class	Chemical	Spiked sediment LC_{50} ($\mu g/g_{oc}$)	Spiked sediment LC_{50} (mg/Kg)	Test Organism	Reference
Chlordanes	Chlordanes	> 2120[1]		*Eohaustorius estuarius*	Greenstein 2014
DDTs	DDTs	101		*Eohaustorius estuarius*	Weston 1996
Pyrethroids	Bifenthrin	1		*Eohaustorius estuarius*	Anderson 2008
Pyrethroids	Cyfluthrin	0.33		*Eohaustorius estuarius*	Greenstein 2014
Pyrethroids	Permethrin	18		*Eohaustorius estuarius*	Anderson 2008
Metals	Cu		439	*Eohaustorius estuarius*	Anderson 2008
Metals	Pb		1980	*Melita plumulosa*	King 2006
Metals	Zn		1790	*Melita plumulosa*	King 2006

[1]The estimated LC_{50} value for chlordanes was based on a sediment toxicity test with no observed effect.

4.3.3 Reference Element Normalization

Because of the natural abundance of metals in the environment, it is important to determine how much is an anthropogenic input and how much is naturally occurring. Reference element normalization is used to determine the background concentration of sediment metals that is associated with local geology. This normalization uses established relationships between the iron content of the sediments and other metals (Schiff and Weisberg 1999). Each metal has a specific relationship to iron describing the expected natural concentration at varying percent iron. These relationships are plotted as a reference line with 99% confidence intervals. Any metal concentration above the 99% confidence interval is considered to be anthropogenically enriched.

4.4 Results and Discussion

4.4.1 Toxicity Stressor Characterization

Sediment Toxicity Screening

Whole Sediment

For round 1, survival of *E. estuarius* in the home sediment and Dana Pt. reference sediments was 100% and 93%, respectively. Survival of *L. plumulosus* in the Dana Pt. control was 87%. The initial toxicity screening of the MdRH sediment samples showed significant reduction in amphipod survival relative to control sediments (Table 4-6, Figure 4-2). Of the two species tested (*E. estuarius* and *L. plumulosus*), both showed significant reduction in survival relative to the Dana Point reference in sediments from stations T1, T3, and T5. *E. estuarius* survival was significantly reduced in sediment from every station with respect to the Home Sediment control, and all but station T4 with respect to the Dana Point grain-size control. When comparing the response of the two amphipods, *L. plumulosus* detected a larger magnitude of toxicity for 2 of the 5 stations (T3 and T5). However, *E. estuarius* displayed a more consistent response across stations relative to both control sediments. In addition, *L. plumulosus* was more sensitive to the Dana Point control, and may not be suited for use in finer grain sediments. Overall, *E. estuarius* provided a more sensitive response, so it was chosen as the test organism for further studies.

Of the stations tested, T1, T3, and T5 had the largest reduction in survival, all of which were significantly different from controls for both organisms (Figure 4-2). Stations T1 and T5 were chosen for further study by TIE. This allowed sediments from back basins and front basins to be represented in the TIE, providing a more comprehensive characterization of toxicity in the harbor.

Table 4-6. Round 1 amphipod survival. Results are expressed as a percentage of the home sediment and Dana Point control survival.

	Eohaustorius estuarius				***Leptocheirus plumulosus***	
	Percent survival relative to Home Sediment		**Percent survival relative to Dana Pt. Sediment**		**Percent survival relative to Dana Pt. Sediment**	
Sample Name	**Mean**	**Stdev**	**Mean**	**Stdev**	**Mean**	**Stdev**
MdRH-T1	77	9	83	10	85	7
MdRH-T2	80	11	86	11	100	9
MdRH-T3	76	11	82	12	76	12
MdRH-T4	83	14	89	15	93	9
MdRH-T5	79	9	85	10	76	11

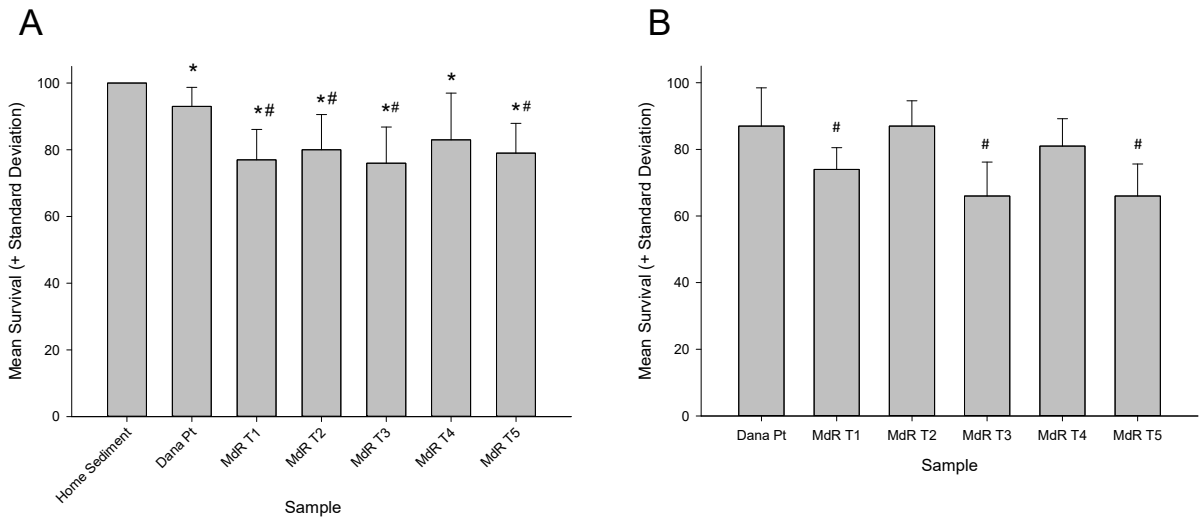

Figure 4-2. Initial whole sediment toxicity test screening with *Eohaustorius estuarius* (A) and *Leptocheirus plumulosus* (B). *Significantly less survival than the home sediment. #Significantly less survival than the Dana Point sediment (unequal variance t-test, p≤0.05).

Round 2 samples included two repeated stations from round 1 (T1 and T5) and eight samples from the sediment quality survey (Figure 4-1b). The initial toxicity screening of the sediment showed no significant reduction in amphipod survival relative to control sediments (Table 4-7). Survival in both the home sediment control and Dana Point sediment was 99%. Unlike the first round of toxicity tests in January 2016, sediment samples from stations T1 and T5 were not toxic to *E. estuarius*. This suggests there may be seasonal variation in sediment toxicity or organism response. Due to the lack of response, further toxicity identification evaluation studies were not performed on these samples.

Table 4-7. Round 2 amphipod survival for *E. estuarius*. Results are expressed as a percentage of the home sediment and Dana Point control survival.

	Percent survival relative to Home Sediment		Percent survival relative to Dana Pt. Sediment	
Sample Name	**Mean**	**Stdev**	**Mean**	**Stdev**
MdRH-T1	100	2	100	2
MdRH-T1	101	0	101	0
MdRH-S1	101	0	101	0
MdRH-S2	101	0	101	0
MdRH-S3	101	0	101	0
MdRH-S4	99	5	99	5
MdRH-S5	99	3	99	3
MdRH-S6	100	2	100	2
MdRH-S7	101	0	101	0
MdRH-S7 dup	101	0	101	0
MdRH-S8	101	0	101	0

Pore Water

Sediment pore water from round 2 samples also displayed no significant toxicity to the amphipods for any of the stations sampled (Table 4-8). These results suggest that bioavailable contaminants (metals and organics) were not present in high enough concentrations in the pore water to cause toxicity. The toxic response remained unchanged from day 7 to 10.

Table 4-8. Percent survival of *E. estuarius* in sediment pore water on days 7 and 10.

	7 day		10 day	
Sample Name	**Mean**	**Stdev**	**Mean**	**Stdev**
Lab Water Control	90	12	90	12
MdRH-T1	100	0	100	0
MdRH-T1	100	0	100	0
MdRH-S1	95	10	95	10
MdRH-S2	100	0	100	0
MdRH-S3	100	0	100	0
MdRH-S4	94	13	94	13
MdRH-S5	90	20	90	20
MdRH-S6	95	10	95	10
MdRH-S7	95	10	95	10
MdRH-S7 dup	95	10	95	10
MdRH-S8	95	10	95	10

Temporal Variation in Toxicity Response

The factor(s) responsible for the change in sediment toxicity between the two rounds of testing cannot be determined without further testing. Such variation in sediment toxicity is not unusual, and has been observed previously in MdRH through CMP monitoring, where a greater incidence of sediment toxicity occurs in samples collected in the winter or spring. Apparent seasonal variability in sediment toxicity magnitude has also been observed in Mission Bay and San Diego Bay (Brown and Bay 2011). Such variation in other studies has been attributed to seasonal stormwater inputs or small-scale spatial variability in sediment conditions. For MdRH, a likely cause of this variability is a reduction in bioavailable contaminant concentrations in summer (e.g., July), relative to winter/spring. Such changes could occur as the result of several processes: 1) geochemical degradation of contaminants to less toxic forms, 2) reduced surface concentrations of contaminants due to sediment deposition or transport by currents, or 3) change in bioavailabity due to stronger binding to sediment particles over time.

The MdRH sediment chemistry data do not provide an explanation for the seasonal variability in sediment toxicity. The total and estimated bioavailable sediment contaminant concentrations were similar between the January and July 2016 sample collections (see Section 4.4.2). However, it is possible that unmeasured stormwater-borne toxicants that are rapidly degraded are present in the system and responsible for the toxicity. For example, substantial degradation of

pyrethroid pesticides (a common toxicant in stormwater) can occur over several months in aquatic environments.

Toxicity Characterization

Whole Sediment

Overall, there were very few changes in round 1 sediment toxicity following each TIE treatment (Figures 4-3 and 4-4). A summary of the TIE survival data can be found in Appendix E. Stations T1 and T5 baseline toxicity was tested in parallel with the TIE treatments and both were found to be more toxic than in the initial screening. Initially, stations T1 and T5 exhibited a mean percent survival relative to home sediment of 77% and 79%, respectively. In the TIE study, the baseline toxicity was reduced to 62% (T1) and 70% (T5).

There was no significant change in observed toxicity when sediments were treated for organic contaminants (coconut charcoal) or ammonia (zeolite). However, sediment dilution for both T1 and T5 sediments led to significant reduction in toxicity when compared to baseline toxicity (Figure 4-3). The increase in survival observed in the SIR 300 treatment was not significantly different from the sediment baseline dilution controls or the SIR 300 blank, suggesting the effect was due to sediment dilution resulting from resin-addition, rather than metal bioavailability reduction. Sediments were also treated for organophosphates and pyrethroids. In the case of the PBO addition (pyrethroid synergist), an increase in toxicity indicates pyrethroid exposure. Amphipod survival was reduced in the PBO treatment for station T1, suggesting an elevated potential for toxic effects from pyrethroids at station T1. No effect of PBO addition was observed for station T5 (Figure 4-4).

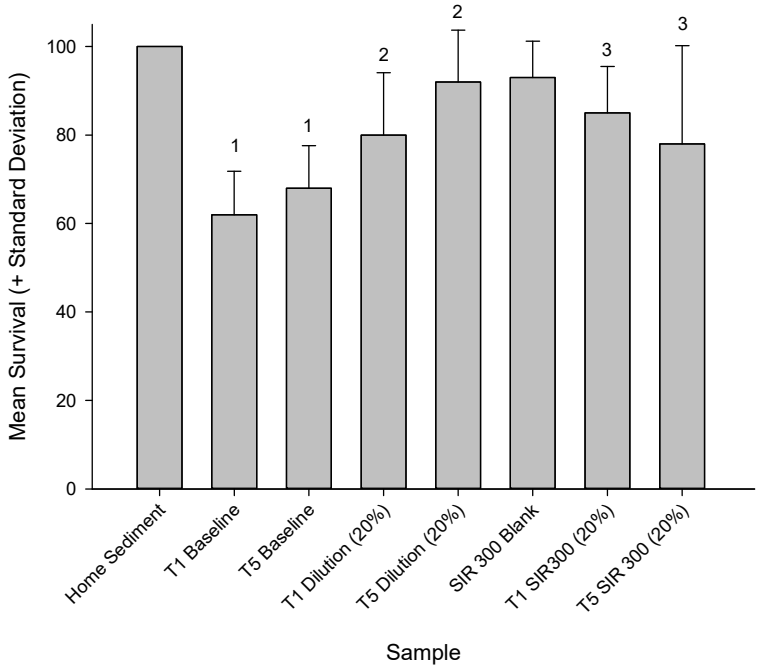

Figure 4-3. Sediment dilution and SIR300 treatment survival compared to baseline toxicity and controls. 1=Survival significantly lower than home sediment. 2=Survival significantly higher than baseline. 3=Survival not significantly different from sediment dilutions or the SIR 300 blank.

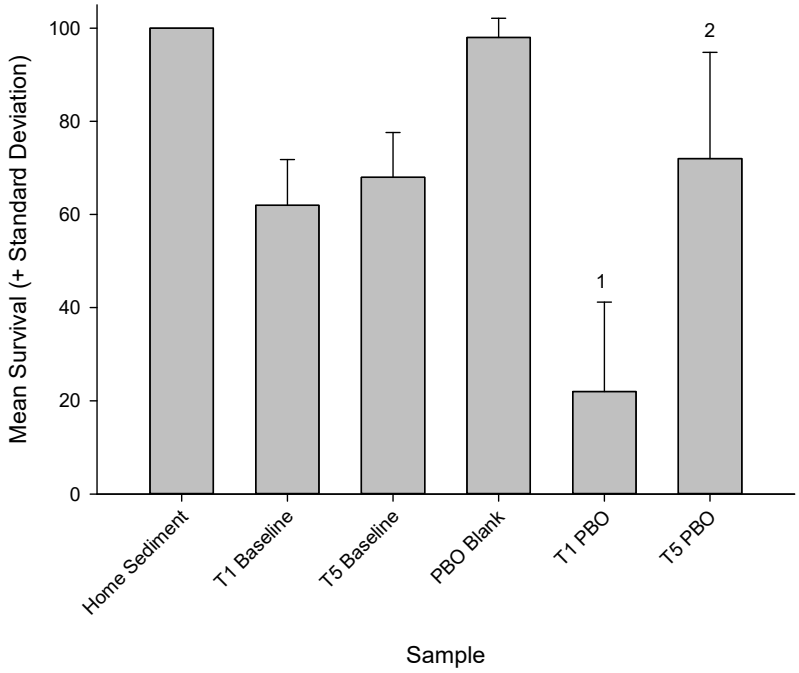

Figure 4-4. PBO treatment compared to baseline toxicity and controls. 1=T1 PBO treatment significantly lower than baseline. 2=T5 PBO treatment not significantly different from baseline.

Pore Water

Sediment pore water from stations T1 and T5 (baseline and all treatments) was not toxic as the amphipod survival did not differ from the control (Figure 4-5). The complete TIE data set is summarized in Appendix F. These results further suggest that there is not a high enough concentration of bioavailable contaminants (metals and organics) present in the pore water to cause a toxic response. These results will be further explored in the discussion of the pore water chemistry.

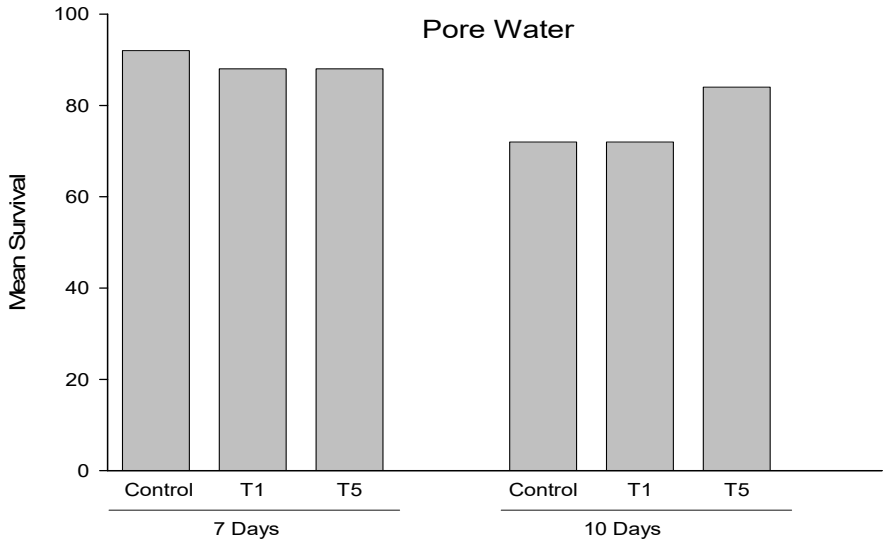

Figure 4-5. Control and baseline pore water toxicity test results.

Total Sediment Contaminants

Sediment characteristics and total contaminant concentrations for round 1 stations T1, T1-TIE, T5, and T5-TIE are summarized in Table 4-9. The full suite of chemistry data is presented in Appendix C (organic contaminants in C-1 and metals in C-2). The two stations had different sediment characteristics which may affect the bioavailability of contaminants. Station T1 sediment had less total organic carbon (TOC), silt, and clay than station T5. Overall, station T5 was a finer grain sediment compared to T1. When comparing the two samples from each station, minimal differences were observed. The sediment characteristics between the chemical analysis samples (e.g., T1) and TIE samples (e.g., T1-TIE) did not change regardless of the differences in handling mentioned previously.

The total concentrations of each organic contaminant class are summarized in Table 4-9 and compared to the TMDL targets. Station T5 had higher concentrations of chlordanes, PCBs and pyrethroids. Total concentration of DDTs was similar among sediment samples and stations. There was a higher concentration of pyrethroids in sediment from station T5 than T1. This result differed from the TIE results, which indicated greater pyrethroid exposure in T1 sediment. This may suggest that the pyrethroids in the T5 sediment are less bioavailable than those in the T1 sediment.

Concentrations of chlordanes and PCBs were greater than their respective TMDL targets. Sediments were also analyzed for total metals. Station T5 had higher concentrations of total Cu, Pb, Ni, Ag, and Zn. Sediment metal concentrations within each station (chemical analysis vs TIE samples) remained consistent. Stations T1 and T5 did not meet metal TMDL targets.

Table 4-9. Sediment chemistry results for round 1 samples.

Category	Constituent	T1	T1-TIE	T5	T5-TIE	TMDL Target
Sediment characteristics	TOC (%)	0.8	0.9	1.8	1.9	NA
	Sand (%)	37.20	33.50	5.50	4.60	NA
	Silt (%)	41.00	42.40	61.90	61.40	NA
	Clay (%)	22.40	24.00	32.50	34.10	NA
	Fines (%)	63	66	94	96	NA
Organics	Total Chlordanes (µg/kg)	1.69	1.80	2.48	2.81	0.5
	Total DDTs (µg/kg)	36.2	41.3	36.8	39.5	NA
	Total PAHs (µg/kg)	222	145	284	296	NA
	Total PCBs (µg/kg)	24.4	28.8	84.2	89.6	3.2
	Total Fipronils (µg/kg)	ND	ND	ND	ND	NA
	Total Pyrethroids (µg/kg)	1.07	1.23	5.82	7.47	NA
Metals	Cd (mg/kg)	0.46	0.54	0.43	0.44	NA
	Cu (mg/kg)	126	152	336	372	34
	Pb (mg/kg)	29.2	32.5	84.4	92.2	46.7
	Hg (mg/kg)	0.23	0.40	0.89	0.93	NA
	Ni (mg/kg)	19.8	21.7	42.0	42.4	NA
	Ag (mg/kg)	0.53	0.61	1.36	1.32	NA
	Zn (mg/kg)	194	221	414	451	150

Sediment characteristics and total contaminant concentrations for all round 2 stations are summarized in Table 4-10. The full suite of chemistry data is presented in Appendix A (organic contaminants in A-1 and metals in A-2). Most of the stations had TOC ranging from 1-2% with the lowest being station S2 (0.9%) and the highest being S1 (2.5%). There was a large range in sand (3.0-38.5%) and clay (12.7-34.4%) content but more consistent silt (47.4-67.5%) content across stations.

The total concentrations of each organic contaminant class are summarized in Table 4-10 for ease of comparison to the TMDL targets. Stations T1 or S5 typically had the lowest concentrations of organic contaminants. Station S1 had the highest values of total chlordanes (13.8 µg/kg), total PAHs (1732 µg/kg), total PCBs (99.5 µg/kg) and total pyrethroids (35.8 µg/kg). Station S7 had the highest total DDTs (110 µg/kg) and high total PAHs (1201 µg/kg) relative to the other samples. None of the stations met the TMDL targets for chlordanes and PCBs.

Round 2 sediments were also analyzed for total metals. The lowest concentrations of metals were typically at stations T1 and S2 with the exception of Cd at T1. A majority of the Cd concentrations ranged from 0.36-0.51 mg/kg with the highest concentrations measured at T1 (0.82 mg/kg) and S1 (1.08 mg/kg). The concentrations of Cu, Pb, and Zn were above the TMDL targets for all stations.

Table 4-10. Sediment chemistry results for round 2 samples.

Category	Constituent	T1	T5	S1	S2	S3	S4	S5	S6	S7	S7-dup	S8	TMDL Target
Sediment characteristics	TOC (%)	1.3	1.8	2.5	0.9	1.9	1.5	1.4	1.7	1.7	1.7	1.8	NA
	Sand (%)	25.3	5.0	14.1	38.5	11.2	14.1	10.7	3.0	7.9	11.0	3.8	NA
	Silt (%)	47.4	60.2	67.5	48.6	62.3	62.0	60.2	67.3	59.6	58.3	64.0	NA
	Clay (%)	27.4	34.4	18.5	12.7	26.6	24.0	29.2	30.0	32.3	30.6	32.7	NA
	Fines (%)	75	95	86	61	89	86	89	97	92	89	97	NA
Organics	Total Chlordanes (µg/kg)	2.24	2.91	13.8	2.34	5.82	2.81	1.50	2.37	4.33	2.44	2.84	0.5
	Total DDTs (µg/kg)	66.3	42.0	73.1	31.0	48.9	35.1	29.8	55.8	110	112	59.8	NA
	Total PAHs (µg/kg)	293	569	1732	376	783	602	513	498	1201	1016	598	NA
	Total PCBs (µg/kg)	44.1	96.9	99.5	46.9	84.6	65.8	59.8	64.3	76.5	75.9	75.9	3.2
	Total Fipronils (µg/kg)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	NA
	Total Pyrethroids (µg/kg)	2.8	4.6	35.8	7.4	9.9	6.4	4.4	6.8	9.6	9.1	6.6	NA
Metals	Cd (mg/kg)	0.82	0.38	1.08	0.38	0.42	0.36	0.40	0.51	0.38	0.42	0.39	NA
	Cu (mg/kg)	191	405	206	211	332	356	554	617	544	508	405	34
	Pb (mg/kg)	54.9	117	111	54.9	109	81.6	83.1	94.8	100	95.6	108	46.7
	Hg (mg/kg)	0.36	1.07	0.43	0.37	0.55	0.58	0.81	0.69	0.76	0.71	0.75	NA
	Ni (mg/kg)	30.7	44.6	36.2	25.0	38.9	36.7	38.3	44.4	42.4	40.8	43.1	NA
	Zn (mg/kg)	295	481	391	280	424	405	502	607	520	481	463	150

Characterization Summary

In round 1, toxicity was detected in sediment from all 5 stations sampled, of which, T1 and T5 were chosen for further analysis. TIEs conducted with *E. estuarius* at T1 and T5 did not identify a specific chemical class responsible for the observed toxicity. In the TIEs, the only significant changes in toxicity were due to sediment dilution with amphipod home sediment (Stations T1 and T5) and PBO treatment of the sediment (Station T1 only). No pore water toxicity was observed for any TIE treatment type or the baseline pore water. These results suggest there may be the potential for toxicity due to pyrethroids at Station T1, but otherwise there was an absence of chemical class treatment effects.

Sediment analyses performed on Stations T1 and T5 detected differences in grain size, TOC, and some contaminant concentrations. Overall, Station T5 sediment was finer grained and had greater TOC content. The higher TOC content may be responsible for the higher total contaminant concentration measured in the T5 sediment. The higher pyrethroids concentration measured in Station T5 sediments may not have been detected in the TIE treatment (PBO) because it was bound to the higher TOC and not bioavailable. The TMDL targets for both organic contaminants and metals were not met at either station with the exception of Pb at Station T1.

In round 2, no toxicity was detected from the sediment and pore water toxicity screening tests. Although there was no toxicity to amphipods, the total sediment concentrations of organic contaminants and metals were greater than the current TMDL targets at all stations. Total contaminant concentrations provide one piece of information, however, the bioavailable fraction is more important for evaluating potential toxic effects to organisms in the system. This will be discussed in more detail in the Toxicant Confirmation section of this report.

4.4.2 Toxicant Confirmation

The analyses in this section use multiple approaches to determine the potential for various contaminant types to cause sediment toxicity in MdRH. Two main approaches are used: 1) comparison of bioavailable contaminants to water quality-based thresholds, and 2) comparison of total sediment contaminants to spiked sediment-based thresholds.

Bioavailable Contaminants and Toxicity Thresholds

AVS-SEM

Bioavailable metals were quantified by measurement of acid volatile sulfides (AVS) and simultaneously extracted metals (SEM) in the whole sediments. Results from round 1 are reported in Table 4-11. The concentration of the individual metals used for calculating the total metals (ΣSEM) are reported in Appendix G-1. Overall, the amount of sulfides (AVS) measured in each of the sediment samples at each station were similar, approximately 3-4 µmoles sulfides/g sediment. Although ΣSEM was similar between samples from the same station, T5 had approximately double the metal concentration compared to T1.

By comparing the difference in total metals and sulfides in the sediment, one can determine the amount of unbound or potentially bioavailable metals that may cause toxicity. The ΣSEM-AVS is normalized to the fraction of organic carbon (*foc*) present in the sediment to account for the additional binding sites dependent on the amount of organic carbon present in the sediments. As

stated in the EPA/600/R-02/011 document (USEPA 2003a), a ΣSEM-AVS/*foc* < 130 µmoles/g OC poses little to no risk to aquatic life, and a value between 130 and 3000 µmoles/g OC requires further tests and/or information to assess risk. All of the values reported here are below the no risk threshold (130 µmoles/g OC).

Table 4-11. Total bioavailable metals in the round 1 sediments as measured by AVS-SEM.

Constituent	T1	T1-TIE	T5	T5-TIE
AVS (µmoles/g)	3.23	4.09	3.11	3.17
ΣSEM (µmoles/g)	2.38	2.37	4.78	5.53
ΣSEM-AVS/*foc* (µmoles/g OC)	-111.58	-191.18	90.68	126.58

Results from the AVS-SEM analysis of round 2 whole sediments is summarized in Table 4-12. The concentration of the individual metals used for calculating the total metals (ΣSEM) are reported in Appendix G-2. The amount of sulfides (AVS) ranged over one order of magnitude, from 0.43-3.75 µmoles sulfides/g sediment. Total metal concentrations varied over a factor of two from about 3-6 µmoles metals/g sediment. As with round 1, station T5 metals were approximately double those measured at station T1. Based on the risk thresholds mentioned previously, the only station classified as having little to no risk to aquatic life was T1. All other stations fell into the category which requires further tests to assess risk. The toxicity test results discussed previously fulfill this requirement and suggest no risk to aquatic life.

Table 4-12. Total bioavailable metals in the round 2 sediments as measured by AVS-SEM.

Constituent	T1	T5	S1	S2	S3	S4	S5	S6	S7	S7-dup	S8
AVS (µmoles/g)	3.75	1.35	0.73	0.43	0.85	0.85	2.37	2.55	1.88	2.46	1.13
ΣSEM (µmoles/g)	2.94	6.23	4.67	3.62	5.49	5.10	5.82	6.10	5.09	5.31	5.75
ΣSEM-AVS/*foc* (µmoles/g OC)	-62.01	270.69	157.75	354.77	199.75	283.19	246.52	209.13	189.03	167.60	256.17

Pore Water

The second method of metal bioavailability analysis was performed only for round 1 samples by measuring the dissolved metal concentration in the extracted sediment pore water (Table 4-13). This was performed on the TIE sediment samples only. Overall, the metal concentrations in the pore water were 1-2 orders of magnitude lower than the total metals in the whole sediment, suggesting limited metal bioavailability. There was no detectable Cd in the pore water, suggesting negligible bioavailability for this metal. The concentrations of Ni and Zn in the pore water were approximately double for T5 compared to T1. This is a similar relationship to the total Ni and Zn concentrations measured at each station in the whole sediments.

Pore water metal concentrations were compared to the current California water quality objectives (CTR; Table 4-13). For both stations T1 and T5, the bioavailable metals were all below the objectives, suggesting metals have very low potential to cause toxicity in the sediments. These findings agree with the AVS-SEM data and TIE characterization results.

Table 4-13. Dissolved metals in round 1 sediment pore water.

Constituent	T1-TIE	T5-TIE	Water quality objective
Cd (µg/L)	ND	ND	9.3
Cu (µg/L)	0.80	0.51	3.1
Pb (µg/L)	0.27	ND	8.1
Ni (µg/L)	0.94	2.58	8.2
Zn (µg/L)	4.30	9.97	81

Equilibrium Partitioning Sediment Benchmarks (ESB)

The toxic units for both the January and July 2016 samplings are summarized in Table 4-14. No contaminant type had TU > 1, indicating a low potential for toxicity, which is consistent with the toxicity screening and characterization results. Overall, the highest toxic units were associated with PAHs and pyrethroids, ranging from 0.06 to 0.16 and 0.04 to 0.26, respectively. Station S7 had the highest toxic units for both contaminant classes. Although these contaminants had high TUs relative to others, they are still 1-2 orders of magnitude lower than levels of concern (TU ≥ 1), and as such, no toxicity is expected.

Table 4-14. ESB-based organic contaminant toxic units for January and July 2016 MdRH stations.

	Toxic units (measured concentration/ESB)											
	January 2016		July 2016									
Chemical	**T1**	**T5**	**T1**	**T5**	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**
Chlordane	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01
DDTs	0.02	0.01	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.02	0.01
PAHs	0.12	0.06	0.09	0.09	0.15	0.14	0.10	0.11	0.11	0.09	0.16	0.09
PCBs	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01
Pyrethroids	0.05	0.05	0.05	0.04	0.13	0.09	0.06	0.07	0.05	0.08	0.26	0.05

Spiked Sediment Toxicity Test Threshold Comparison

The spiked sediment-based toxic units for both the January and July 2016 samplings are summarized in Table 4-15. For chlordanes and DDTs, the total concentrations were used for the ratio. Pyrethroid toxic units were calculated separately for three individual pyrethroid compounds and summed. Although there were only LC_{50} values available for three pyrethroids, they represented the most prevalent components of this class and accounted for 70% (on average) of the total pyrethroid concentration in the sediments.

Comparison to spiked sediment toxicity thresholds indicated a low potential for toxicity for all TMDL compounds evaluated, with an average of less than one TU for each compound. Similar to the ESB analysis, variation in TUs did not correspond with the changes in toxicity observed between the January and July samples. This result suggests that none of the measured compounds were present at sufficient concentrations individually to account for the toxicity measured in the January sediment samples.

Copper TUs for several of the July sediment samples were slightly greater than 1, indicating a greater relative potential for toxicity compared to other measured constituents (Table 4-15). However, other more reliable lines of evidence from the toxicity tests, pore water chemistry, and AVS-SEM analyses show that copper and other metals actually have very low potential to cause sediment toxicity to amphipods. The spiked-sediment estimates of the relative toxicity potential for metals in MdRH sediments are an overestimate due to likely differences in bioavailability of metals in the spiked sediment tests relative to field sediments. The equilibration times for the spiked metals exposures were relatively short (approximately two weeks (King et al. 2006, Anderson et al. 2008)) and probably did not allow for full equilibration of the spiked metals with sediment constituents that reduce bioavailability, such as sulfides and organic matter. This situation would lead to a lower LC_{50} that may not be representative of field conditions. The spiked sediment studies did not provide information on important binding factors (e.g., AVS), so it was not possible to account (normalize) for changes in metal bioavailability between the laboratory tests and MdRH sediments.

The spiked sediment threshold comparison results also indicate a somewhat higher potential for toxicity due to pyrethroids, with an average TU of 0.5 for the July samples. As for metals, these results are expected to be more conservative than the ESB results due to limitations of the spiking method. However, the pyrethroid results were normalized for TOC effects on bioavailability and are in accordance with the ESB analysis that suggest pyrethroids have greater potential to contribute to MdRH sediment toxicity than the TMDL trace organic constituents.

Table 4-15. LC_{50}-based toxic units for January and July 2016 MdRH stations.

	Toxic units (measured concentration/spiked sediment LC_{50})											
	January 2016		July 2016									
Chemical	**T1**	**T5**	**T1**	**T5**	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**
Chlordanes	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01
DDTs	0.07	0.03	0.06	0.03	0.03	0.05	0.03	0.03	0.03	0.04	0.07	0.04
Pyrethroids	0.24	0.49	0.26	0.26	1.15	0.49	0.43	0.50	0.28	0.62	0.47	0.38
Cu	0.29	0.77	0.44	0.92	0.47	0.48	0.76	0.81	1.26	1.41	1.24	0.92
Pb	0.01	0.04	0.03	0.06	0.06	0.03	0.06	0.04	0.04	0.05	0.05	0.05
Zn	0.11	0.23	0.16	0.27	0.22	0.16	0.24	0.23	0.28	0.34	0.29	0.26

4.4.3 Sediment Contamination Relationships

The analyses in this section provide context for understanding the relative degree of sediment contamination in MdRH. Two types of analyses are presented. The first analysis compares MdRH contamination and toxicity to conditions in other Southern California bays and estuaries. This analysis provides information to determine how conditions in MdRH compare to the occurrence of contamination and toxicity in similar environments. The second analysis examines the influence of MdRH sediment particle characteristics on trace metal content. This analysis uses reference element normalization to describe the relative contribution of background conditions and anthropogenic inputs on sediment metals concentration.

Contaminant-Toxicity Associations in Southern California Embayments

Contaminant concentration and toxicity have been measured in Southern California bays and estuaries the Southern California Bight Regional Monitoring program since 1998. To help put the MdRH data into perspective at a regional level, the data from the 2013 Bight Regional Monitoring Survey (Bight '13) were compared to data from the January and July 2016 samples. The concentration and toxicity data for sediment percent fines, chlordanes, DDTs, PAHs, PCBs, pyrethroids, Cu, Pb, and Zn were all investigated. The 50th, 75th, and 90th percentiles of the Bight '13 data were calculated for comparison to the MdRH data (Table 4-16).

Sediment fines and contaminant concentrations in MdRH were generally higher than in other Southern California bays and estuaries, yet there was no strong association between contaminant concentration and sediment toxicity (Figures 4-6 through 4-14). These results indicate that the measurements of specific contaminants of concern in the MdRH TMDL have little predictive value for describing the occurrence of sediment toxicity in Southern California embayments.

Sediment percent fines in the January and July sampling have some of the highest values relative to other regional sediments. Stations S3, S5, S6, S7, S8, and T5 were above the 90th percentile of all samples for percent fines (Figure 4-6). The plot of sediment fines shows a more consistent trend of increased toxicity occurrence (e.g., less than 80% survival) than for the following contaminant plots. However, the cause of this toxicity cannot be linked directly to grain size, as most of the samples with the highest percentage of fines were not significantly toxic. Rather, the trend of greater toxicity shown by this plot is likely due to the tendency for many contaminants to have higher concentrations on finer sediments and for these sediments to frequently occur in depositional areas that receive ongoing inputs of toxics from the watershed.

Plots of total chlordanes and DDTs show very little association between concentration and the occurrence of toxicity (Figures 4-7 and 4-8). While concentrations of these constituents in MdRH stations are above the 90th percentile of Bight '13 data, most of the data in this range show no sediment toxicity. The TMDL targets for these compounds are also shown on the plots and show little relationship to the occurrence of sediment toxicity, as a similar proportion of toxic sediments occur above and below the targets.

PAH and PCB concentrations in MdRH are also elevated relative to the Bight '13 data and show a similar lack of association between contaminant concentration and toxicity as noted for DDTs and chlordanes (Figures 4-9 and 4-10). For pyrethroids, the concentrations in MdRH sediments appear to be more typical of those in other embayments and range from the 75th to 95th percentile (Figure 4-11). Toxicity is also not consistently present at total pyrethroid concentrations typical

of MdRH, suggesting that other constituents or perhaps variations in bioavailability that are not represented in the monitoring data are important causes of sediment toxicity in Southern California embayments.

Concentrations of the three metals with TMDL targets in MdRH (Cu, Pb, and Zn) were elevated with respect to the Bight '13 data (80^{th} to 100^{th} percentile for Cu, 65^{th} to 99^{th} percentile for Pb, and 78^{th} to 100^{th} percentile for Zn; Figures 4-12, 4-13, and 4-14). However, there is little apparent association between metal concentration and incidence of toxicity. Overall, the MdRH data show higher survival at higher metal concentrations than many of the Bight '13 data. The TMDL targets for metals do not show any relationship to the incidence of toxicity, with similar numbers and severity of toxic stations above and below the target.

These comparisons of MdRH contaminant concentration and toxicity to other locations indicate that MdRH sediments frequently have elevated concentrations of both organics and metals. However, the data indicate that there is little association between contaminant concentration, TMDL targets, and the occurrence of sediment toxicity. The highest concentrations of total sediment contaminants were usually not toxic, or the incidence of toxicity was inconsistent, which suggests that other factors (e.g., types of toxics or variation in bioavailability) are responsible for the sediment toxicity observed in MdRH and other embayments in Southern California.

Table 4-16. Bight '13 sediment % fines and contaminant concentration percentiles.

Parameter	50^{th} Percentile	75^{th} Percentile	90^{th} Percentile
Fines (%)	66.2	80.9	87.3
Chlordanes (µg/kg dw)	NA	0.34	2.00
DDT (µg/kg dw)	0.80	3.20	12.1
PAH (µg/kg dw)	259	649	1360
PCB (µg/kg dw)	3.00	13.4	37.9
Pyrethroid (µg/kg dw)	NA	1.58	11.4
Copper (mg/kg dw)	54.3	104	187
Lead (mg/kg dw)	21.6	37.0	64.4
Zinc (mg/kg dw)	129	187	266

NA= available due to a large proportion of the data being non-detects

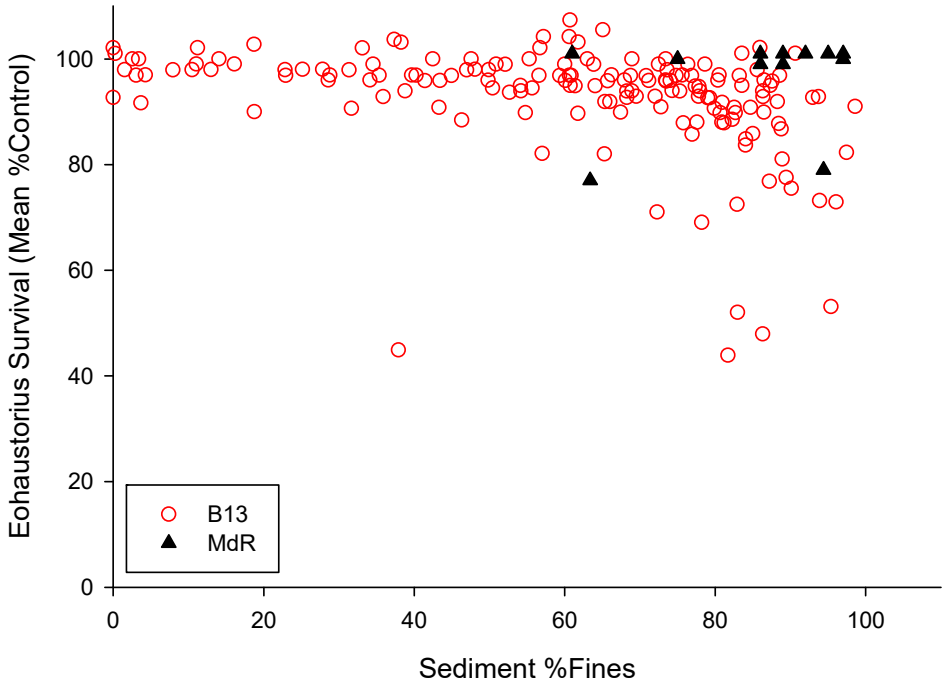

Figure 4-6. ***Eohaustorius estuarius*** **survival as a function of sediment % fines from Bight '13 (open red circles) and MdRH (closed black triangles).**

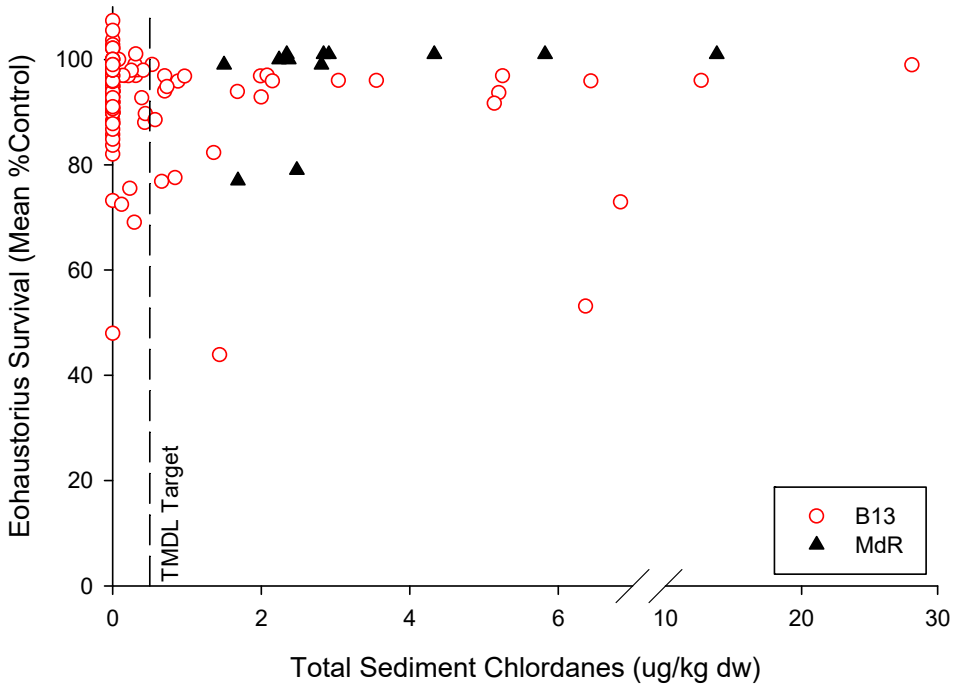

Figure 4-7. ***Eohaustorius estuarius*** **survival as a function of total sediment chlordane concentration from Bight '13 (open red circles) and MdRH (closed black triangles).**

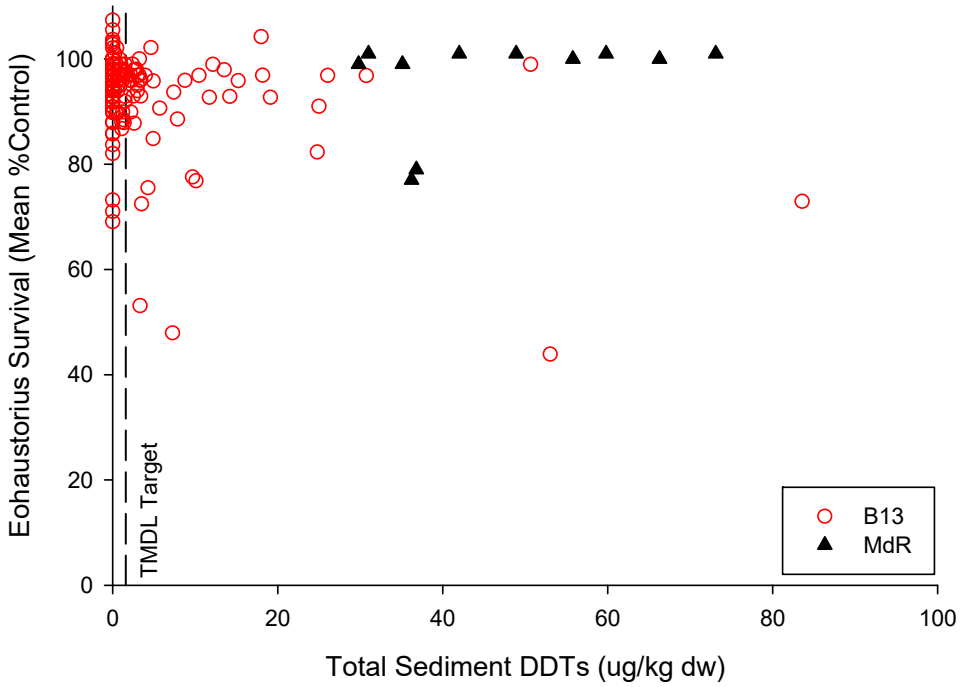

Figure 4-8. *Eohaustorius estuarius* survival as a function of total sediment DDT concentration from Bight '13 (open red circles) and MdRH (closed black triangles).

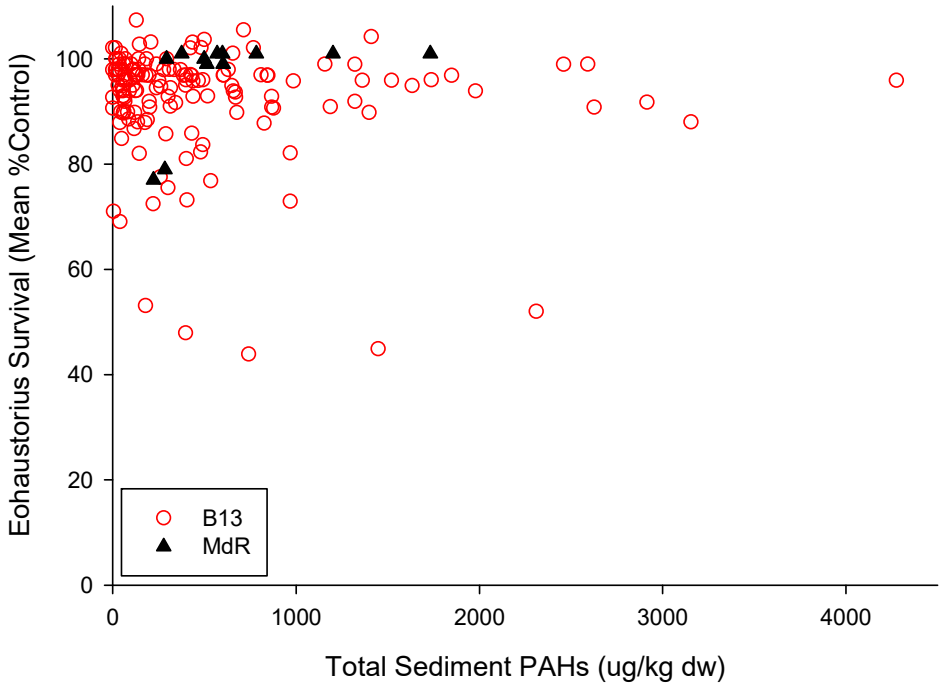

Figure 4-9. *Eohaustorius estuarius* survival as a function of total sediment PAH concentration from Bight '13 (open red circles) and MdRH (closed black triangles).

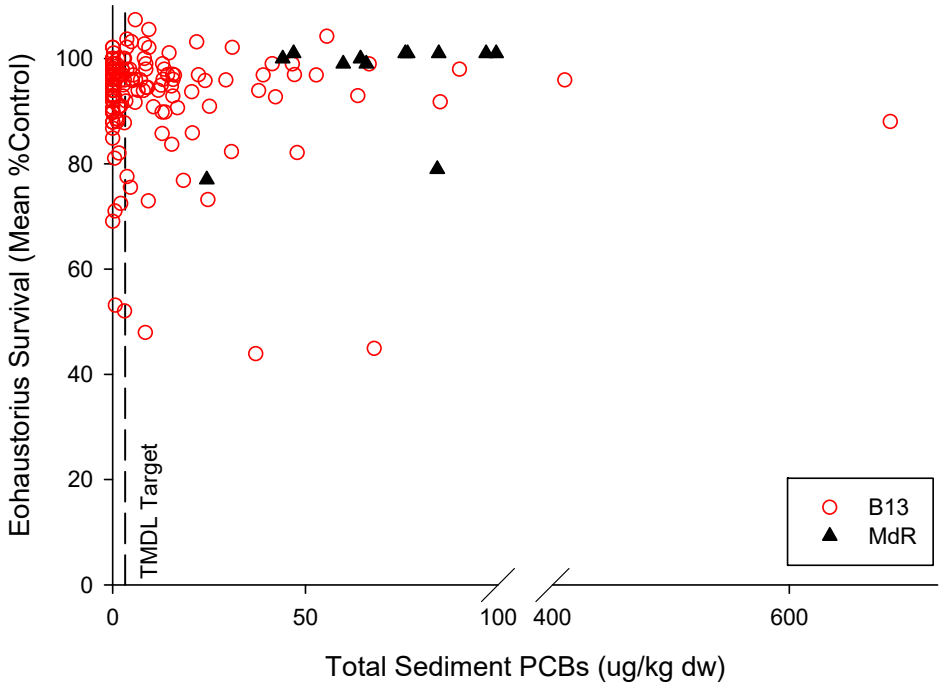

Figure 4-10. ***Eohaustorius estuarius*** **survival as a function of total sediment PCB concentration from Bight '13 (open red circles) and MdRH (closed black triangles).**

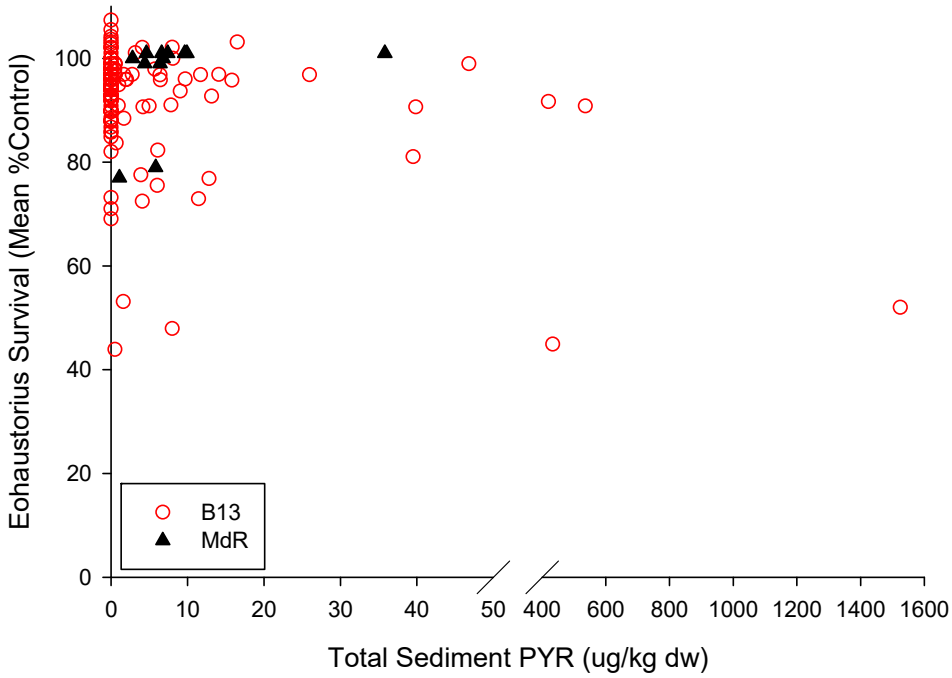

Figure 4-11. ***Eohaustorius estuarius*** **survival as a function of total sediment pyrethroid concentration from Bight '13 (open red circles) and MdRH (closed black triangles).**

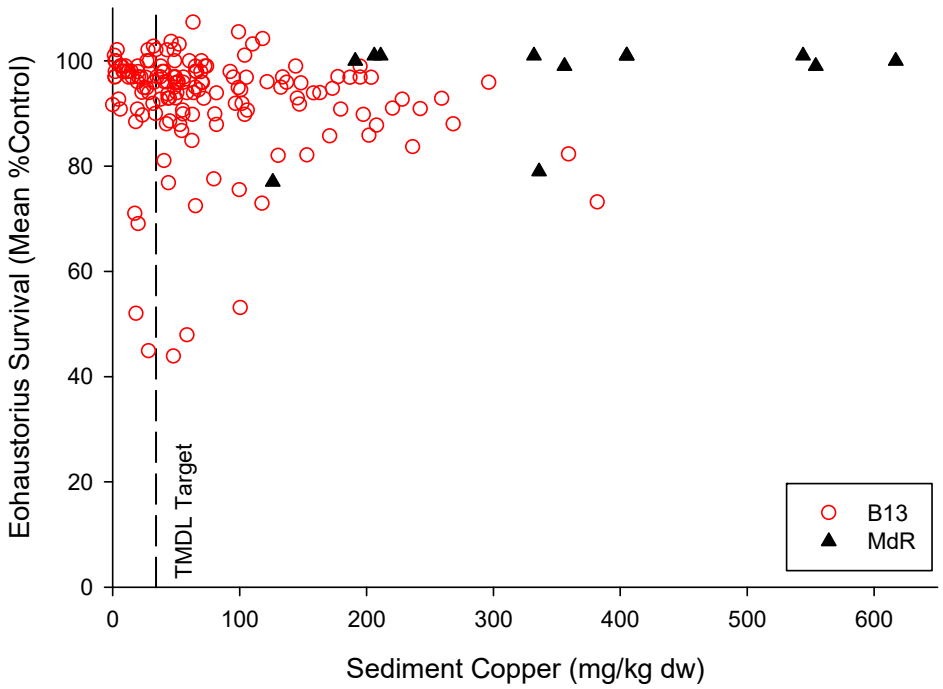

Figure 4-12. *Eohaustorius estuarius* survival as a function of sediment copper concentration from Bight '13 (open red circles) and MdRH (closed black triangles).

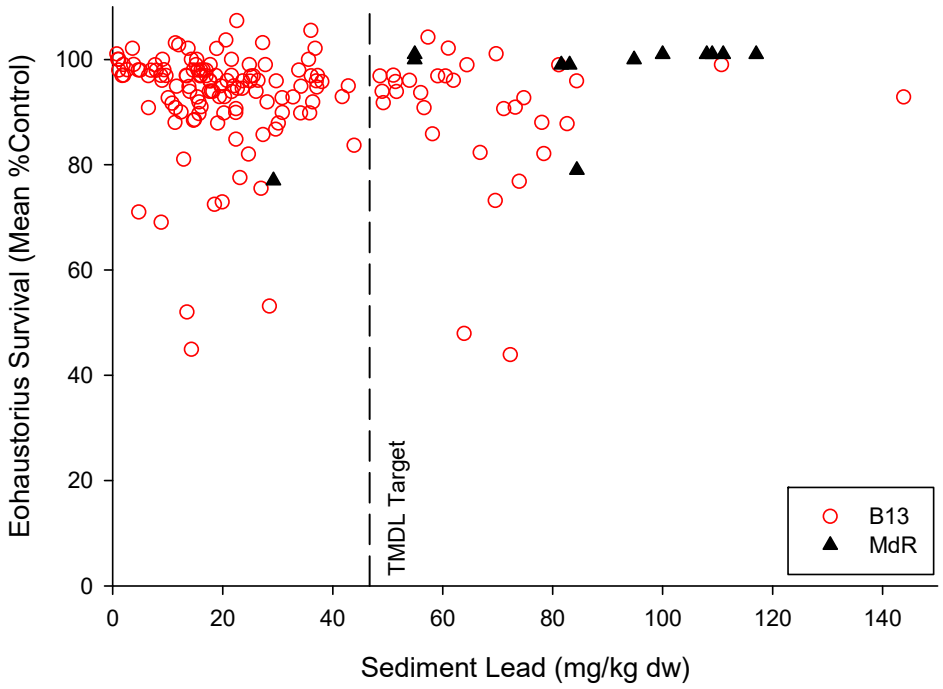

Figure 4-13. *Eohaustorius estuarius* survival as a function of sediment lead concentration from Bight '13 (open red circles) and MdRH (closed black triangles).

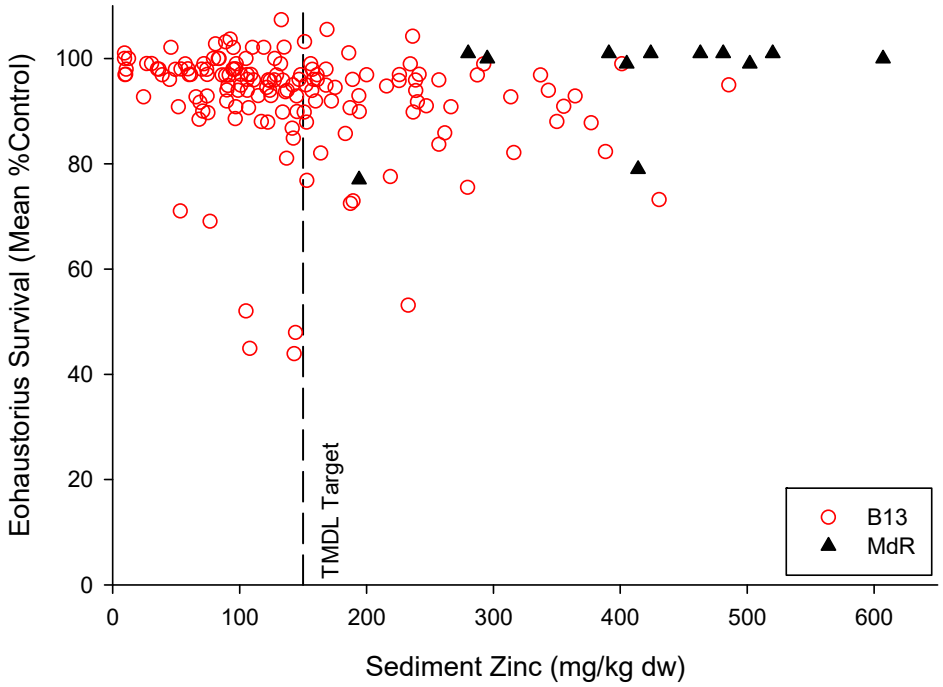

Figure 4-14. *Eohaustorius estuarius* survival as a function of sediment zinc concentration from Bight '13 (open red circles) and MdRH (closed black triangles).

Reference Element Normalization

The reference element normalization plots (Figures 4-15, 4-16, and 4-17) show that sediment metal concentrations are significantly influenced by natural variation the characteristics of sediment particles from the watershed. Concentrations of copper, lead, and zinc are naturally higher in silt and clay particles as a result of geological variation. This relationship is quantified by normalizing the concentration data to the sediment iron (Fe) content, which varies similarly in different sediment types (e.g., higher concentration in silts and clays) but is not substantially influenced by anthropogenic inputs. The regression line shown on each plot describes the influence of sediment particle type on metal concentration, with the upper 99% confidence interval indicating the range of background concentration expected in Southern California sediments in the absence of anthropogenic influence. A similar relationship is evident when sediment particle size is used instead of iron, but the relationship is less robust due to lower precision and specificity in grain size measurements.

The normalization plots verify the assumption in the TMDL that most of the sediment-associated copper, lead, and zinc is from anthropogenic sources, either within the Harbor (e.g., antifouling paint) or from watershed runoff. In all cases, sediment concentrations are above the 99% confidence limit of the reference element regression (Figures 4-15, 4-16, and 4-17).

More significantly, these normalization plots demonstrate that the TMDL targets do not take into account the influence of natural variation in background metal concentrations in Southern California and may be unattainable for copper and zinc under natural conditions. The TMDL targets for copper and zinc are at or below expected background concentrations for the sediment

type (high proportion of silt and clay) present in MdRH. For example, the copper TMDL target is below the expected background concentration for 10 of the 12 MdRH sediment samples sampled in January and July (e.g., Fe concentration of 3-6%). One of the reasons the ERL-based TMDL targets are not accurate for Southern California is because they were derived primarily using data from other parts of the U.S., where sediment geological conditions and levels of metal contamination are different.

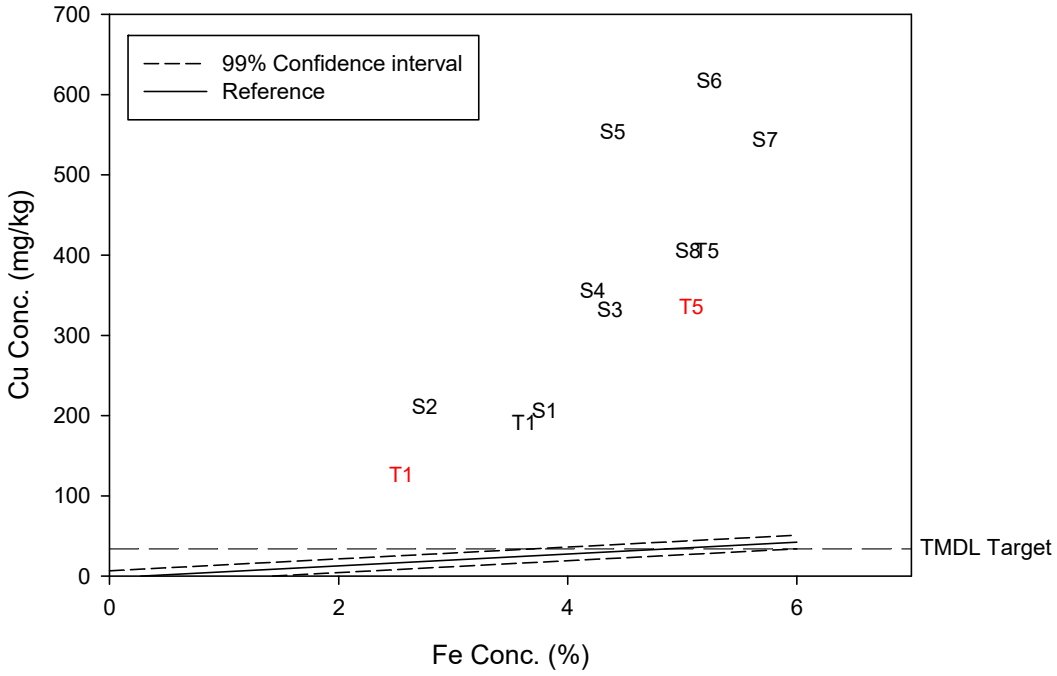

Figure 4-15. Reference element normalization plot for copper. January 2016 stations T1 and T5 are in red, and July 2016 stations are in black.

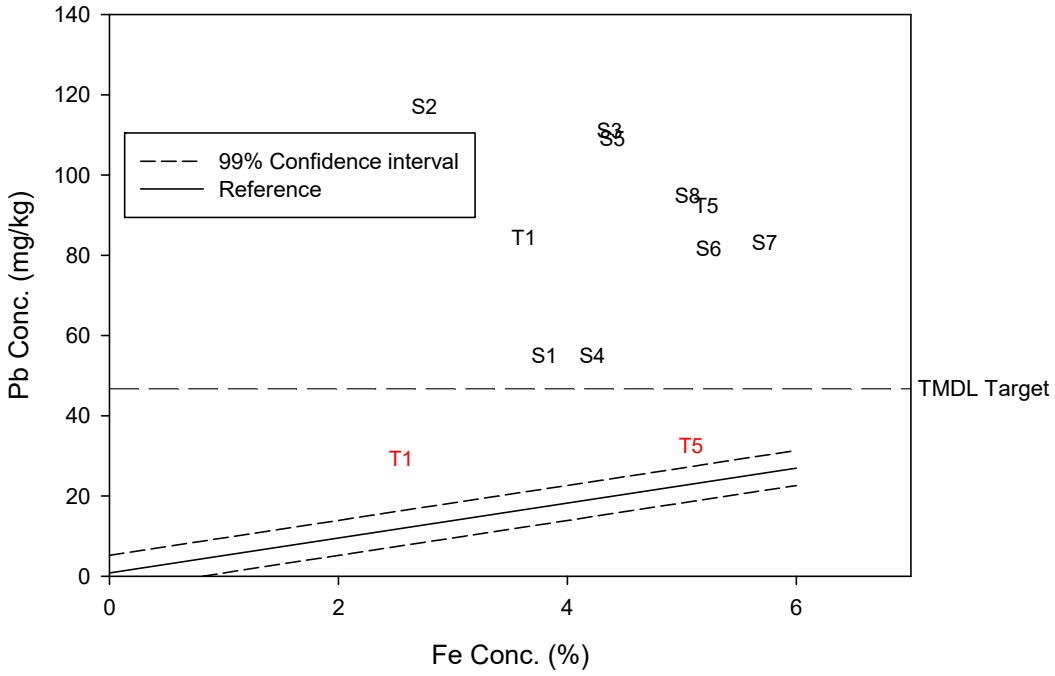

Figure 4-16. Reference element normalization plot for lead. January 2016 stations T1 and T5 are in red, and July 2016 stations are in black.

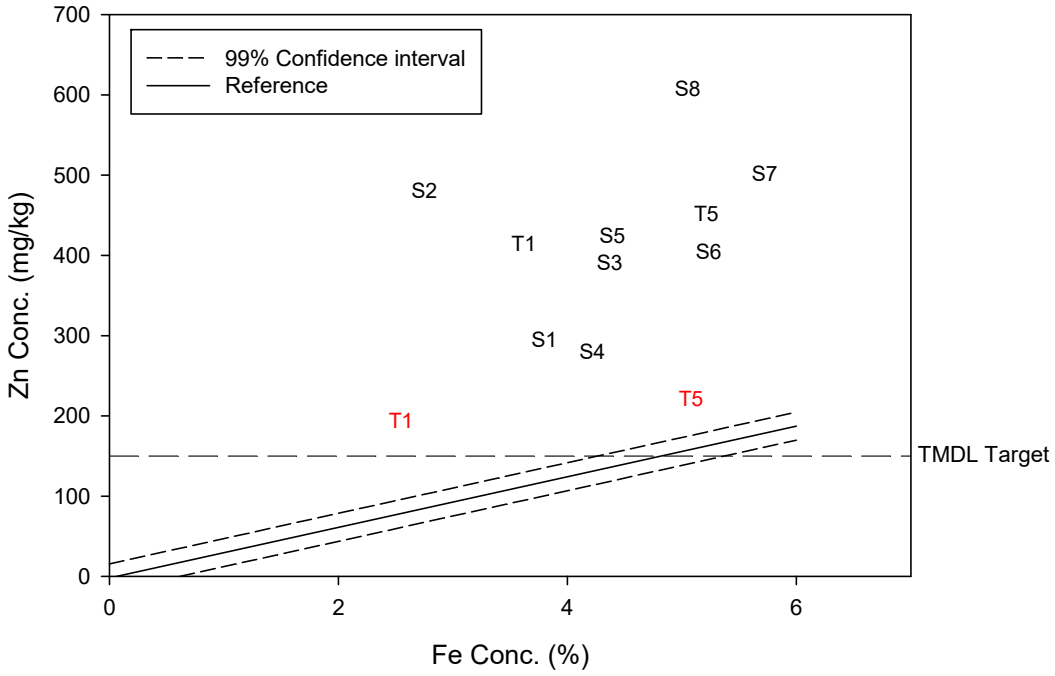

Figure 4-17. Reference element normalization plot for zinc. January 2016 stations T1 and T5 are in red, and July 2016 stations are in black.

4.5 Summary and Conclusions

Multiple lines of evidence, including both direct and comparative analyses, were evaluated in this stressor identification analysis. Although temporal variability was observed for overall toxicity of the sediments, the measured concentrations of contaminants (metals and organics) varied little. Because toxicity was only observed in January, TIE stressor characterization could only be conducted on the first round of samples. This analysis was not able to identify a specific factor likely to be responsible for sediment toxicity.

Variations in toxicity between sampling events did not correspond to changes in the total sediment contaminant concentrations. This suggests that variation in contaminant bioavailability or unmeasured contaminants may have influenced the results. Multiple methods were used in this study to estimate the bioavailability of sediment contaminants and identify contaminants with the greatest potential to cause toxicity. All of the ΣSEM-AVS/*foc* values (concentration of bioavailable metals) determined for the January and July sediment samples indicated no or minimal toxic risk from metals. This conclusion is supported by the lack of toxicity to amphipods in both pore water toxicity tests and the Round 2 sediment tests. Round 1 pore water metal concentrations were also below the water quality objectives for the State of California. The bioavailable metal data, especially the pore water analysis results, demonstrate that sediment metal concentrations are not responsible for sediment toxicity to amphipods in MdRH.

Equilibrium partitioning theory was used in conjunction with acute water quality toxicity thresholds to calculate the total sediment concentration (ESB) of trace organics at which the bioavailable fraction is expected to equal the criterion. For each of the contaminants evaluated, all toxic units were less than 1, and therefore little risk is expected from the TMDL listed organics. The greatest relative potential risk was calculated for two non-TMDL contaminant groups, pyrethroids and PAHs. Calculated toxic units for these two compound classes were still too low to account for sediment toxicity in MdRH.

The ESB results were confirmed by more conservative analyses that compared the chemistry data to thresholds based on spiked sediment toxicity tests. This analysis also indicated that chlordanes and DDTs were not likely to be responsible for MdRH sediment toxicity, and that greater toxicity potential was associated with pyrethroids.

Iron normalization analyses showed anthropogenic enrichment of copper, lead, and zinc at all stations in both sampling periods. These analyses also showed that the high iron content (corresponding to high fines content) of the sediment resulted in naturally elevated levels of copper and zinc that would likely exceed current TMDL targets even without anthropogenic inputs to the Harbor. These TMDL targets are therefore not attainable for MdRH under any management scenario. Development of site-specific TMDL targets are needed to address the bioavailability and geochemical factors influencing metal toxicity. Sufficient data are currently available to support the evaluation of alternative TMDL targets.

A specific stressor responsible for MdRH sediment toxicity could not be determined in this study. Multiple lines of evidence indicate that the trace organic and metal contaminants listed in the TMDL are not the principal cause of sediment toxicity in the Harbor. The seasonal variation in sediment toxicity suggests that other factors, including unidentified toxic chemicals in watershed runoff, have an important influence on MdRH sediment toxicity.

5. BENTHIC COMMUNITY STRESSOR IDENTIFICATION

Benthic macrofauna are useful indicators of the condition of marine and estuarine habitats because the community composition changes in a relatively predictable fashion when disturbed (e.g., Pearson and Rosenberg 1978; Rhoads et al. 1978; Gray et al. 2002). Benthic macrofauna serve as good integrators of their local environmental conditions as they live directly in and often feed upon the sediment where many toxins accumulate. They have limited mobility to escape stress, and many species live for multiple years. In addition, most benthic macrofaunal communities include a taxonomically diverse mixture of organisms spanning multiple phyla, with which comes a wide range of physiological responses to stress. This differential response to multiple stressors make the changes in community composition useful in identifying the different types of stressors to which the organisms have been exposed (e.g., Christman and Dauer 2003; Lenihan et al. 2003; Thrush et al. 2008).

5.1 Analytical Approach

Based upon the SQO Benthic Line of Evidence (BLOE) assessment (Section 3.2.3), the macrobenthic communities of MdRH from the July 2016 sampling event were degraded – all samples were scored as having either Moderate or High Disturbance. As such, the goal of this work was to use the composition of those macrobenthic samples and determine how they compare to other embayments from across Southern California to infer the potential anthropogenic stressors that may have led to their impaired condition. Given the nature of the data from this study, all of the stressor characterization analyses were inferential in nature and were not experimental manipulations, nor mechanistic explanations of stressor exposure to the macrobenthic community. As such, no single analysis should be viewed as definitive. To account for the nature of these types of post-hoc analyses, a multiple line of evidence approach was employed where stressor-macrobenthic community relationships were evaluated with different analytical techniques that ranged from local to regional in scope.

Four broad classes of stressors known to impact macrobenthic fauna in relatively predictable ways were considered for this stressor characterization: 1) Sediment Toxics – chemicals found in the sediment that have deleterious effects on the resident fauna (e.g., heavy metals, PAHs, pesticides); 2) Eutrophication – the production of excessive organic matter that accumulates in the sediment and can be harmful to the resident biota; 3) Physical disturbance – alteration of the sediment environment where the biota reside (e.g., dredging, sediment replacement, scouring from boats); and 4) Low-dissolved oxygen – hypoxic and anoxic conditions in which the benthic macrofauna are suffocated. Each of these classes of stressors was evaluated with up to four different analytical methods (see methods below) to determine if they were a potential cause for the observed impairment in the macrobenthic community observed in MdRH.

The results from each analysis for a given stressor were considered in parallel and used to make a final evaluation on the likelihood of that stressor as a potentially causative agent responsible for the impaired macrobenthic community that was observed. For each analysis type, results were characterized as: 1) Supporting (+) – there was evidence that the stressor could be responsible for the observed benthic community; 2) Weakening (–) – there was evidence that the stressor could not have been responsible for the observed benthic community; 3) Indeterminate (0) – the evidence can neither support nor refute the stressor being responsible for the observed

benthic community; or 4) No evidence (NE) – data were not available to complete the analysis. Final conclusions were based upon the strength of each individual result and concordance in characterization (supporting, weakening, etc.) among the different analyses.

Two types of analysis were conducted: ecological and statistical. The ecological analyses, Community Composition and Indicator Taxa, rely upon the fauna observed in the MdRH samples. The statistical analyses, Stressor Exposure and Stressor Response, are designed to put the observed biological and environmental patterns from the MdRH samples into a regional context and determine if they are similar or dissimilar to what would be expected under normal conditions.

5.2 Analytical Methods

5.2.1 Ecological Analyses

Community Composition

Within this type of analysis, the characteristics of all of the individual taxa found in a sample were considered together (relative abundance, known sensitivities and tolerances to different stressors, natural history, etc.). Patterns in the presence or absence of different species, families, and feeding guilds were noted and used to infer potential exposure to each of the different types of stressors being considered based upon patterns documented in the literature (e.g., Pearson and Rosenberg 1978; Rhoads et al. 1978; O'Brien and Keough 2013; Jumars et al. 2015).

Indicator Taxa

Whereas the community composition analysis was a generalized consideration of the macrobenthic fauna found in the MdRH, indicator taxa are individual species known to indicate specific conditions in a waterbody. The presence or absence of these specific taxa can be used to indicate exposure to a given stressor. For this study, suites of sediment heavy metals or sediment Total Organic Carbon (TOC) / Total Nitrogen (TN) indicator taxa developed for Southern California embayments and estuaries (Gillett, unpublished) were used to evaluate the macrobenthic samples from MdRH. If any taxa indicative of deleteriously high concentrations of metals or sediment organic matter were found in the MdRH samples above the indicator threshold, the samples would be scored as supporting evidence for sediment toxics or eutrophication, respectively, as a causative agent. Conversely, if any taxa indicative of low concentrations of metals or sediment organic matter were present at abundances above the indicator threshold, the samples would be scored as weakening evidence for sediment toxics or eutrophication, respectively, as a causative agent. No indicator taxa were available to evaluate the physical disturbance or low dissolved oxygen stressors.

5.2.2 Statistical Analyses

The goal of the statistical analyses was to infer the structuring factors on macrobenthic community composition by assessing the resident macrofauna of MdRH and the potential stressors they were exposed to in the context of ecologically similar sites across the region. However, changes in community composition are reflective of both natural and anthropogenic stressors. In order to separate the relative influence of anthropogenic stressors, one must first control for as many natural gradients as possible and remove them from consideration. The initial step in this process involves aggregating as much data from Southern California embayments as possible, from which ecologically similar sites can be identified. In stressor

characterization studies, these ecologically similar sites are referred to as comparator sites, as they are used to compare and contrast the biological and environmental conditions at the "test" site(s) (following Norton et al. 2014). Results from this process were used to identify the best available, ecologically similar sites to MdRH, which diminishes much of the natural variation from the subsequent analyses of macrobenthic biota – stressor response relationships.

Compilation of Benthic Data

Data considered for inclusion to the data set had to be collected from embayments within the Southern California Bight (Point Conception in the North to the US/Mexico border in the South) following the biogeographic patterns in macrobenthic community composition observed by Ranasignhe et al. (2012). Additionally, the data sets needed to contain information on macrobenthic infauna (identification and abundance), environmental data (water depth, sediment composition, etc.), and some stressor data (sediment toxics, organic matter, water quality) collected synoptically and with all appropriate QA/QC measures. Furthermore, the data needed to be organized in a manner that facilitates easy integration with the other datasets (e.g., similar formatting, data types, sampling gear, etc.).

The initial data set contained 747 samples from across the region (Table 5-1). These data were primarily from different iterations of the Southern California Bight Regional Monitoring Program (Bergen et al. 1998, Ranasinghe et al. 2003, Ranasinghe et al. 2007, Ranasinghe et al. 2012) and the State of California's Sediment Quality Objectives (SQO) development program (Bay et al. 2014; Ranasinghe et al. 2009). These data were collected between 1994 and 2008, spanned the full range sediment types (100% mud to 100% sand), and were from waters of 0.3 to 30 m deep. The taxonomic information from each of these datasets was synonymized as much as possible to the standards of Edition 8 of the Southern California Association of Marine Invertebrate Taxonomists taxa list (SCAMIT, 2013) to maximize comparability among the data.

Table 5-1. List of compiled datasets and some of their basic environmental and biological characteristics.

Project	Number of Samples	Range in % Fines Observed	Years Covered	Species Richness
Bight Regional Monitoring Program 2003	129	96.9 - 2.8	2003	98.2 (1 - 104)
Bight Regional Monitoring Program 2008	191	99.8 - 0	2008	42.0 (3 - 131)
Bight Regional Monitoring Program Pilot	1	60.2	1994	57
Bight Regional Monitoring Program 1998	127	100.0 - 8.0	1998	41.9 (6 - 117)
SQO Calibration Data	299	100.0 - 1.0	1998-2005	74.3 (6 - 270)

Selecting Comparator Sites

A pool of ecologically similar comparator sites was selected from the aggregated macrobenthic data set. Comparator sites were selected based upon similarity (standardized Euclidean distance, a unit-less measure of position in multivariate space) of natural, abiotic environmental gradients (Table 5-2) known to influence macrobenthic community structure in estuarine and embayment settings (e.g., (Boesch et al. 1976, Holland 1985, Snelgrove and Butman 1994, Gogina et al. 2010). Sites with a standardized Euclidean distance less than 0.5 relative to MdRH conditions were used as comparator sites for the 2016 sampling event.

Table 5-2. Average values of the natural, abiotic environmental gradients from the 2016 MdRH sediment sampling stations.

Environmental Gradient	Average of 2016 Marina del Rey Stations
Water Depth (m)	4.6
Salinity (psu)	31.8
% Sand	12.7
% Fines	87.4
Relative Distance to Mouth[1]	0.53

[1]Relative distance to mouth is the path distance ("as the fish swims") from the station to a point in the center of the mouth of the embayment divided by the maximum possible path distance in that embayment.

A pool of 237 comparator sites met the similarity criteria (Figure 5-1). Macrobenthic samples from these sites were from similar sediment, salinity, depth, and distance regimes to the 2016 Marina del Rey sampling event (Figures 5-2, 5-3, 5-4, and 5-5). The comparator sites represented a variety of different types of embayments across the Southern California Bight, though they typically came from more sheltered portions of the region's coastal zone, as opposed to more open water parts of an embayment (e.g., San Pedro Bay or Central San Diego Bay) (Figure 5-6). Interestingly, they did not include all of the samples collected from Marina del Rey or adjacent to the Ballona Creek Estuary as part of previous studies (Figure 5-7).

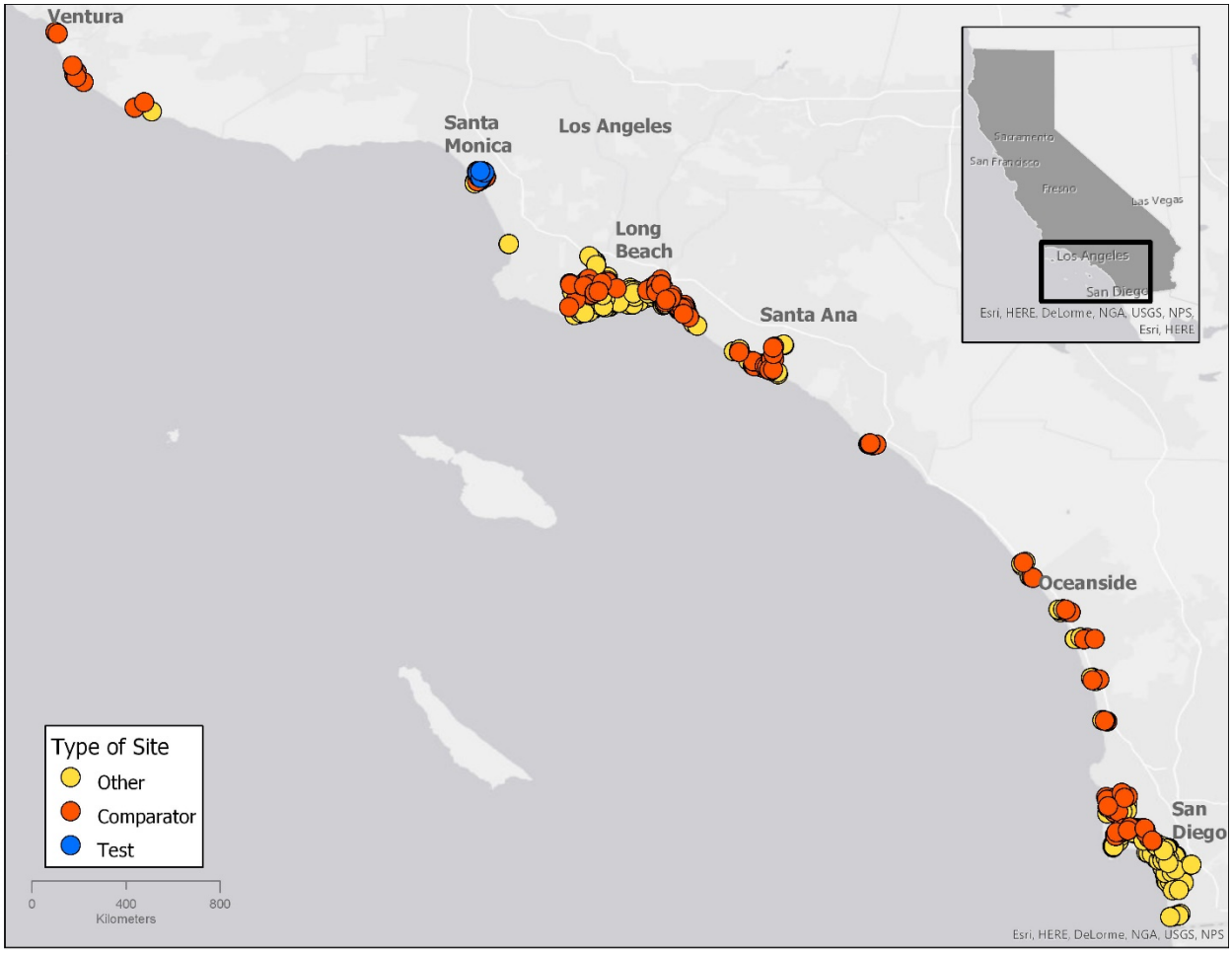

Figure 5-1. A map of the Southern California Bight highlighting the location of the 747 sites considered for inclusion as potential comparator sites for the MdRH 2016 sampling event. Blue dots represent the location of the MdRH samples (test sites). The orange dots represent the location of comparator sites. The yellow dots represent the location of other sites, considered, but eventually not selected via the multivariate evaluation to be comparator sites. The inset depicts the location of the Southern California Bight within the context of the Pacific Coast of the United States.

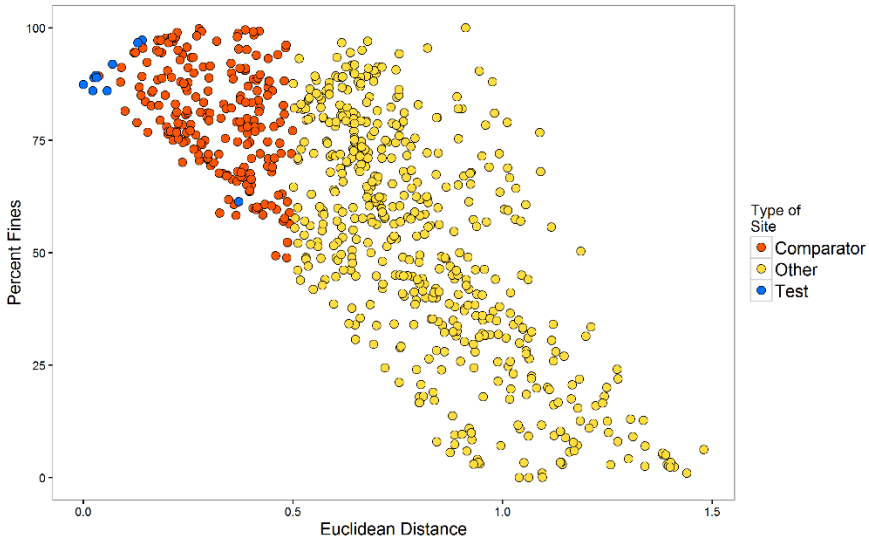

Figure 5-2. Scatter plot of changes in sediment composition (Percent Fines) as related to similarity to the average value among the 2016 samples from Marina del Rey (see Table 5-2). Values are plotted for test sites from Marina del Rey Harbor in 2016 (blue symbols), comparator sites used in subsequent stressor ID analyses (orange), and the remaining other sites (yellow) in the aggregated benthic data set. Note that not all sites in the dataset had sediment data associated with them. Greater Euclidean distance implies greater dissimilarity to the MdRH site average.

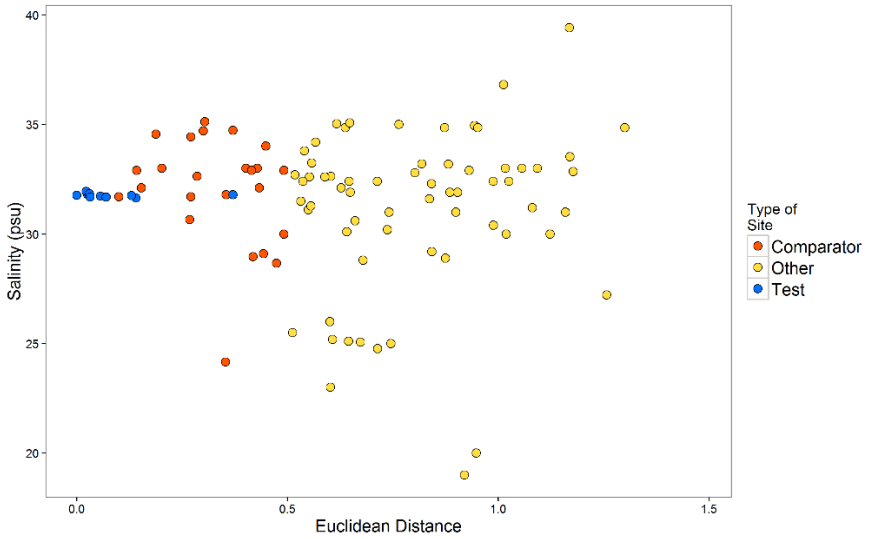

Figure 5-3. Scatter plot of changes in water column salinity (psu) as related to similarity to the average value among the 2016 samples from Marina del Rey (see Table 5-2). See Figure 5-2 for description of symbols. Note that not all sites in the dataset had salinity data associated with them. Greater Euclidean distance implies greater dissimilarity to the MdRH site average.

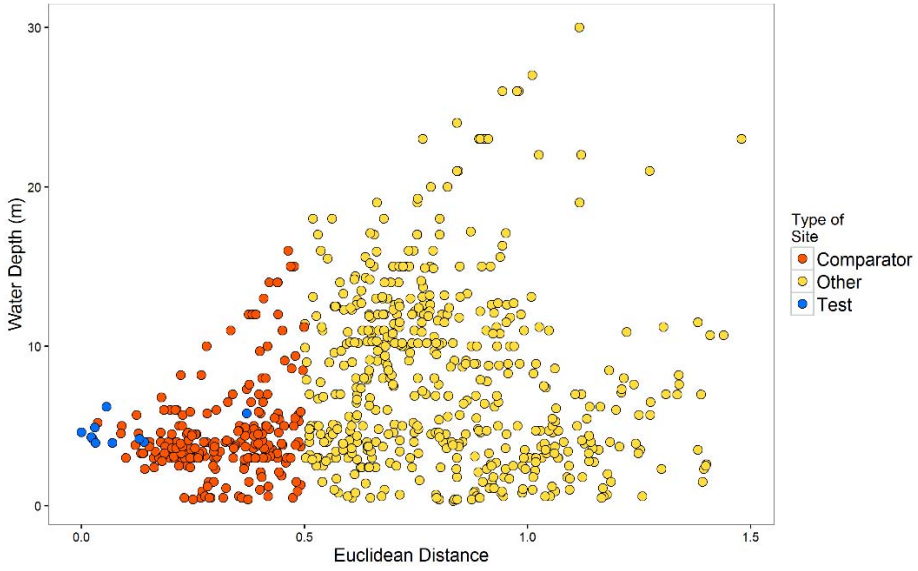

Figure 5-4. Scatter plot of changes in water depth (m) as related to similarity to the average value among the 2016 samples from Marina del Rey (see Table 5-2. See Figure 5-2 for description of symbols. Note that not all sites in the dataset had water depth data associated with them. Greater Euclidean distance implies greater dissimilarity to the MdRH site average.

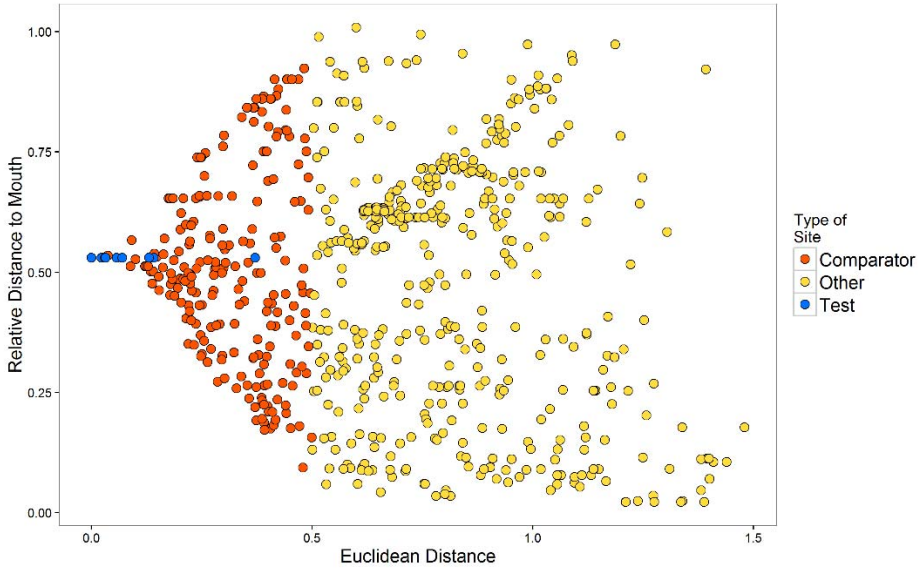

Figure 5-5. Scatter plot of changes in the relative distance to mouth of the embayment as related to similarity to the average value among the 2016 samples from Marina del Rey (see Table 5-2). See Figure 5-2 for description of symbols. Greater Euclidean distance implies greater dissimilarity to the MdRH site average.

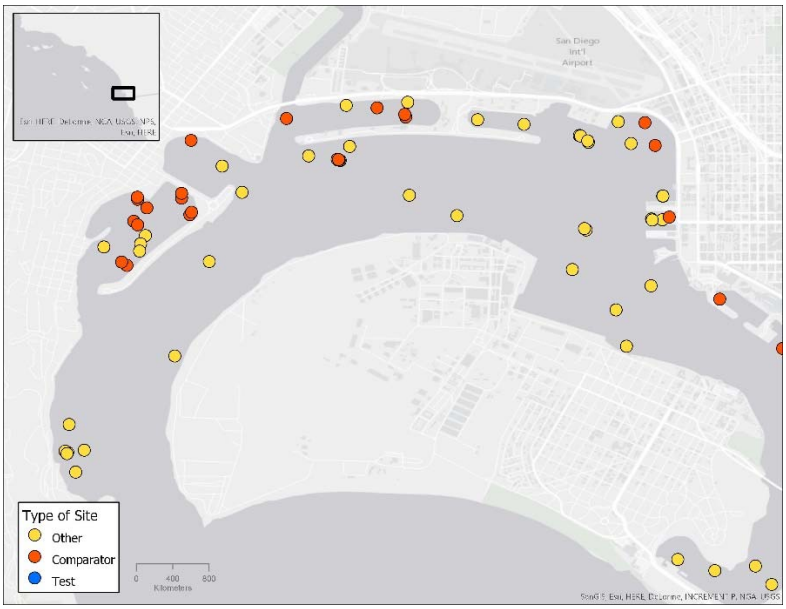

Figure 5-6 A map of the northern San Diego Bay highlighting the location of the comparator sites in the more enclosed portions of the embayment versus the open water parts of a large embayment like San Diego Bay. The orange dots represent the location of comparator sites. The yellow dots represent the location of other sites, considered, but eventually not selected via the multivariate evaluation to be comparator sites. The inset depicts the location of the area along the Southern California Bight.

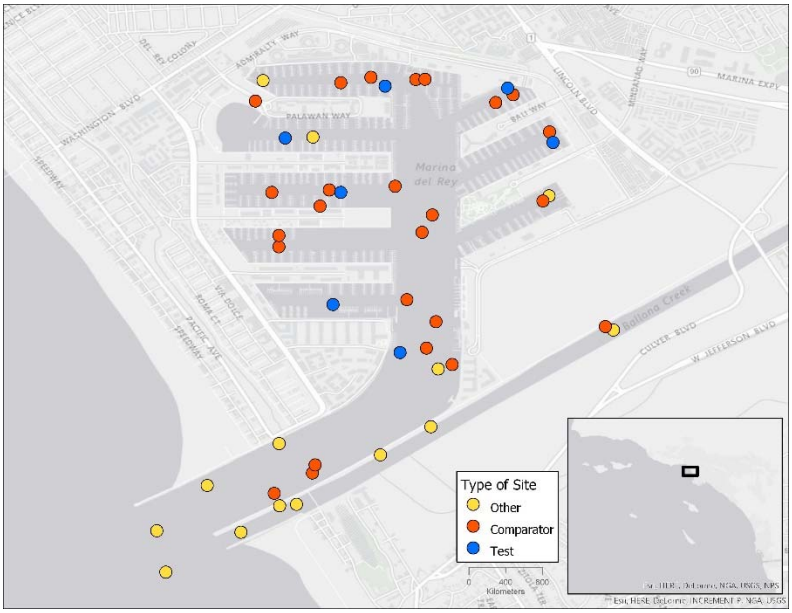

Figure 5-7 A map of the Marina del Rey Harbor highlighting the location of the comparator sites and test sites and that not all sites within MdRH and adjacent Ballona creek were similar to the sites sampled within the harbor in 2016. See Figure 5-6 for description of symbols. The inset depicts the location of the area along the Southern California Bight.

The biological and stressor data from the comparator sites were used to assess the relative influence of the major classes of stressors in coastal embayments (sediment toxics, physical disturbance, low dissolved oxygen, or eutrophication) on the macrobenthic fauna collected from MdRH as part of the July 2016 sediment quality survey. By only using data from ecologically similar comparator sites, a stronger distinction can be made between anthropogenic factors and natural gradients which may affect overall macrobenthic community composition, as well as the SQO benthic community index scores. Subsequent analyses compared potential exposure and observed community responses across gradients of different types of anthropogenic stressors among the comparator and MdRH sediment quality survey sites, which provides some insight into the cause(s) behind the poor SQO benthic line of evidence results observed in 2016.

Stressor Exposure

The degree of stressor exposure at the MdRH sites (e.g., concentration of copper) was compared to the stressor exposure at a subset of comparator sites having less impaired benthic communities (i.e., sites with BLOE scores indicating Reference or Low Disturbance [see section 3.3.4]). Stressor exposure to arsenic, cadmium, chromium, copper, mercury, number of measured compounds that were observed above their ERM value in a given sample (ERM exceedances), nickel, high molecular weight PAHs, low molecular weight PAHs, total PAHs, total PCBs, and total DDTs were evaluated for the sediment toxics stressor. Sediment TOC was evaluated for the eutrophication stressor. Stressor exposure levels were compared in a two-step process. First, median concentrations at the 2016 MdRH sites were compared to those of the comparator sites with a Mann-Whitney U-test. If there were no significant differences at $\alpha=0.1$, then the analysis was scored as weakening the case for that particular stressor as a causative agent. If there were significant differences between MdRH and the comparator sites, schematic box-and-whisker plots (Tukey 1977) comparing the two groups of data were evaluated:

- if the 25th percentile of the MdRH sites was greater than the 75th percentile of the comparator sites, then the analysis was scored as supporting evidence for the stressor class being a causative agent (Figure 5-8A);

- if the 25th percentile of the MdRH sites was less than 75th percentile of the comparators, but the median of the MdRH was still greater than the 75th percentile of the comparators, then the analysis was scored as indeterminate evidence for the stressor class being a causative agent (Figure 5-8B);

- if the median value of the MdRH sites was less than the 75th percentile of the comparators, then the analysis was scored as weakening evidence for the stressor class being a causative agent (Figure 5-8C)

All analyses were done using R v3.2.5.

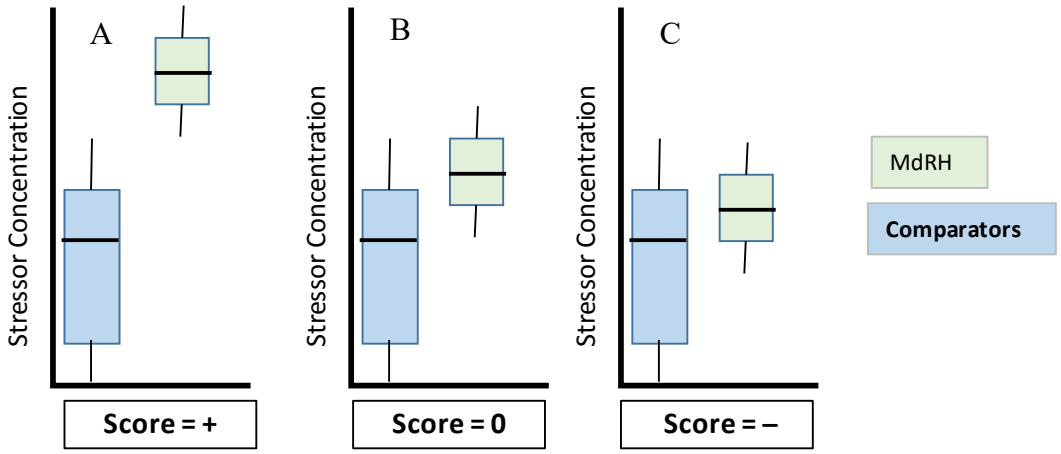

Figure 5-8. Examples of schematic box-and-whisker plots for Stressor Exposure analysis and how they would be scored

Stressor Response

The stressor response analysis evaluated the relationship between stressor exposure level and the likelihood of observing a moderately or highly impacted benthic community (BLOE condition of Moderate or High Disturbance). The apparent stressor-response relationships at all of the comparator sites were modeled using logistic regression with 95% profile likelihood confidence intervals (Allison 1999). Once the stressor response relationship was established, the mean value of the stressor found at in the MdRH samples was compared to the model in order to determine the probability of observing an impaired benthic community at the stressor concentration (Figure 5-9):

- If the probability of observing an impaired benthic community minus the 95% confidence interval was greater than 50%, then the analysis was scored as supporting evidence for the stressor class being a causative agent.

- If the probability of observing an impaired benthic community plus the 95% confidence interval was less than 50%, then the analysis was scored as weakening evidence for the stressor class being a causative agent.

- In all other instances, the analysis was scored as indeterminate evidence for the stressor class being a causative agent.

Stressor-response relationships with arsenic, cadmium, chromium, copper, mercury, ERM exceedances, nickel, high molecular weight PAHs, low molecular weight PAHs, total PAHs, total PCBs, and total DDTs were evaluated. Sediment TOC was evaluated for the eutrophication stressor. All analyses were done with Proc Logistic using SAS v9.4.

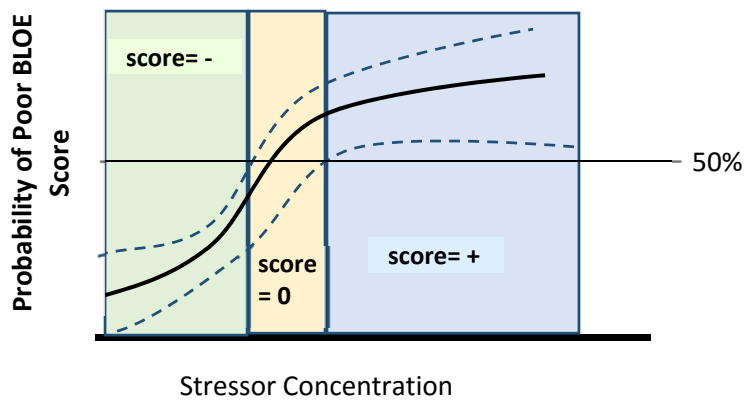

Figure 5-9. Example logistic regression plot illustrating classification of stressor response relationship, as supporting, weakening, or indeterminate. The solid regression line represents the estimated probability of observing poor BLOE scores at the given stressor concentration (x-axis). The dashed lines represent the 95% confidence intervals about those estimates.

5.3 Results and Discussion

Community Composition

The macrobenthic community of MdRH was composed of the taxa typical of marine embayments of Southern California (Appendix H). However, the number of taxa observed at all stations was relatively low (mean species richness = 13.9), less than half the number of taxa typically seen in embayments across the region (Gillett, unpub.). Furthermore, abundances were very low (mean abundance of 95.1 individuals), less than 1/5th of that typically observed in embayments from across the region (Gillett, unpub.). This pattern of low abundance and diversity clearly indicate an impaired community as represented by the SQO BLOE results detailed in section 3.3.4. These patterns are clearly indicative of exposure to some manner of stress – likely a moderate, chronic stressor that continuously effects the resident biota, versus an acute, catastrophic stress (e.g., persistent anoxia, sediment removal) followed by re-colonization.

Despite the low taxonomic richness, the trophic diversity of the samples (i.e., number of different feeding guilds) was relatively high. The presence of a number of predator/scavenger taxa (*Scoletoma* spp., nemerteans, *Glycera Americana*), filter-feeders (*Phoronis* sp, *Tagelus affinis*), interface-feeders (spionid polychaetes, *Theora lubrica*), and deposit feeders (*Leitoscoloplos pugettensis*, *Mediomastus* sp) indicate a mature community that has not recently experienced a strong physical disturbance that would "reset" the benthic community and lead to dominance of early-successional taxa (e.g., Rhoads et al. 1978; Zajac and Whitlach 1982). These patterns provide weakening evidence that physical disturbance is a causative agent for the impaired macrobenthic communities observed in the MdRH.

The absence of nearly any crustacean taxa in the MdRH samples is anomalous compared to typical embayment samples from Southern California. Given their relatively high metabolism, and consequently their susceptibility to low dissolved oxygen conditions, the absence of crustaceans, specifically amphipods and isopods, serve as supporting evidence that low dissolved oxygen may be somewhat responsible for the impaired benthic communities. However, the absence of crustaceans could also be indicative of exposure to sediment toxics, specifically

pesticides designed to kill insect pests, which are taxonomically and physiologically similar to crustacean arthropods.

The absence of high numbers of deposit feeding worms (specifically the polychaete *Capitella capitata*) and any oligochaetes serves as weakening evidence for eutrophication as a causative agent for the impaired benthic communities in MdRH. Under eutrophic conditions, these types of organisms become dominant components of the macrobenthic community (e.g., Pearson and Rosenberg 1978), as they are able to take advantage of the excessive organic matter that accumulates in the sediment (e.g., Sardá et al. 1996). Furthermore, the presence of filter-feeding organisms that obtain their food from the water column near the sediment surface (bivalve mollusks and phoronids) suggests there was relatively good water circulation in the harbor and not excessive amounts of organic matter and sediment in the water column which is evidence against highly eutrophic conditions.

Indicator Taxa

There was at least one high-metals indicative taxon in each of the macrobenthic samples from MdRH and no low-metals indicative taxa (Table 5-3). In two samples, S1 and S3, the abundance of these taxa was above the high-metals threshold, indicating supporting evidence for metals as a causative agent for the impaired benthic communities observed in the harbor. Though the taxa were present in all of the other MdRH samples, their abundances were not above the threshold, so the samples were scored as indeterminate evidence for metals as the causative agent. When all of the samples were considered together, most of which were scored as indeterminate, this analysis was scored as indeterminate for sediment toxics as a causative agent for the impaired macrobenthic communities across the whole harbor.

There was a mix of low-TOC or low TN indicative taxa present in every macrobenthic sample from MdRH (Table 5-4). However, in all but one of these samples (MdRH-S1), the abundance of these taxa was below the threshold that would be scored as weakening evidence for eutrophication. Similarly, in the few samples where high-TOC/TN indicative taxa were observed, those abundances were below the threshold used to indicate supporting evidence for eutrophication. Taken together, the overall indicator taxa analysis was scored as indeterminate evidence for eutrophication as a causative agent for the impaired macrobenthic community observed in the harbor as a whole.

There were no indicator taxa to evaluate the physical disturbance or low dissolved oxygen stressors, so they were scored as no evidence for this analysis.

Table 5-3. Abundances of High Metals indicative taxa from the MdRH macrobenthic samples. Abundances above the indicator threshold (i.e., indicative of potential high metals exposure) are noted. Metals Score for a station indicates supporting evidence (+), indeterminate evidence (0), or no evidence (NE) towards toxics, specifically metals, as a causative agent for impaired benthic communities.

Station ID	Replicate	Species	Abundance	≥ Threshold	Metals Score
MdRH-S1	1	*Leitoscoloplos pugettensis*	21	yes	+
		Scoletoma sp	29	yes	
MdRH-S2	1	*Leitoscoloplos pugettensis*	7		0
		Neotrypaea gigas	1		
		Pseudopolydora paucibranchiata	1		
		Scoletoma sp	4		
MdRH-S3	1	*Leitoscoloplos pugettensis*	13		+
		Pseudopolydora paucibranchiata	147	yes	
		Scoletoma sp	5		
MdRH-S4	1	*Leitoscoloplos pugettensis*	19		0
		Pseudopolydora paucibranchiata	47		
		Scoletoma sp	2		
MdRH-S5	1	*Leitoscoloplos pugettensis*	19		0
		Scoletoma sp	3		
		Theora lubrica	11		
MdRH-S6	1	*Leitoscoloplos pugettensis*	8		
		Scoletoma sp	2		
		Theora lubrica	12		
MdRH-S7	1	*Leitoscoloplos pugettensis*	18		0
		Pseudopolydora paucibranchiata	4		
		Theora lubrica	18		
MdRH-S7	2	*Leitoscoloplos pugettensis*	3		0
		Pseudopolydora paucibranchiata	11		
		Theora lubrica	6		
MdRH-S8	1	*Leitoscoloplos pugettensis*	18		0
		Pseudopolydora paucibranchiata	14		
		Scoletoma sp	1		
		Theora lubrica	3		

Table 5-4. Abundances of High TOC, High TN, Low TOC, Low TN indicative taxa from the MdRH macrobenthic samples. Abundances above the indicator threshold (i.e., indicative of potential high or low organic matter conditions) are noted. Scores indicate supporting evidence (+), indeterminate evidence (0), weakening evidence (-), or no evidence (NE) towards TOC and TN, which are aggregated to determine the overall score for eutrophication as a causative agent for impaired benthic communities.

Station ID	Replicate	Species	Abundance	Indicator Type	≥ Threshold	TOC Score	TN Score	Eutrophication Score
MdRH-S1	1	*Scoletoma* sp A	9	Low TOC	yes	-	NE	-
MdRH-S2	1	*Pseudopolydora paucibranchiata*	1	Low TN		NE	0	0
MdRH-S3	1	*Pseudopolydora paucibranchiata*	147	Low TN		0	NE	0
MdRH-S4	1	*Monticellina* sp	3	Low TN		NE	0	0
		Pseudopolydora paucibranchiata	47	Low TN				
MdRH-S5	1	*Theora lubrica*	11	High TOC		0	NE	0
MdRH-S6	1	*Theora lubrica*	12	High TOC		0	0	0
		Monticellina sp	1	Low TN				
MdRH-S7	1	*Theora lubrica*	18	High TOC		0	0	0
		Pseudopolydora paucibranchiata	4	Low TN				
MdRH-S7	2	*Theora lubrica*	6	High TOC		0	0	0
		Monticellina sp	1	Low TN				
		Pseudopolydora paucibranchiata	11	Low TN				
MdRH-S8	1	*Theora lubrica*	3	High TOC		0	0	0
		Monticellina sp	5	Low TN				
		Pseudopolydora paucibranchiata	14	Low TN				

Stressor Exposure

Nearly all of the individual components associated with sediment toxics as a potential stressor were elevated at MdRH compared to the less impaired subset of comparator sites (Figure 5-10). Only cadmium, low-molecular weight PAH and total PAH concentrations were lower at MdRH than other locations in Southern California embayments. Given the pattern of elevated concentrations and poorer biological condition at MdRH, the stressor exposure analysis was scored as supporting evidence for sediment toxics as a causative agent for the macrobenthic community conditions at MdRH (Table 5-5).

Sediment TOC at MdRH was equivalent to that observed at comparator sites with macrobenthic communities in better condition, so this analysis was scored as weakening evidence for eutrophication to be a causative agent (Table 5-5). Limited data availability prevented the evaluation of physical disturbance or low dissolved oxygen as causative agents in this type of analysis (i.e., scored as NE).

Table 5-5. Scoring results from the stressor exposure analysis comparing the levels of exposure at MdRH to those at comparator sites with reference or low-impact condition macrobenthic communities. Scores can be supporting evidence (+), weakening evidence (-), indeterminate evidence (0), or no evidence (NE) towards the candidate stressor as a causative agent for the impaired macrobenthic communities at MdRH.

Candidate Stressor	Components	Component Score	Candidate Score
Toxics			+
	Arsenic	+	
	Cadmium	-	
	Chromium	+	
	Copper	+	
	Mercury	+	
	ERM Exceedances	+	
	Nickel	+	
	HMW PAH	0	
	LMW PAH	-	
	Total PAH	-	
	Total PCB	+	
	Total DDT	0	
Eutrophication			-
	TN	NE	
	TOC	-	
Physical Disturbance		NE	NE
Low Dissolved Oxygen		NE	NE

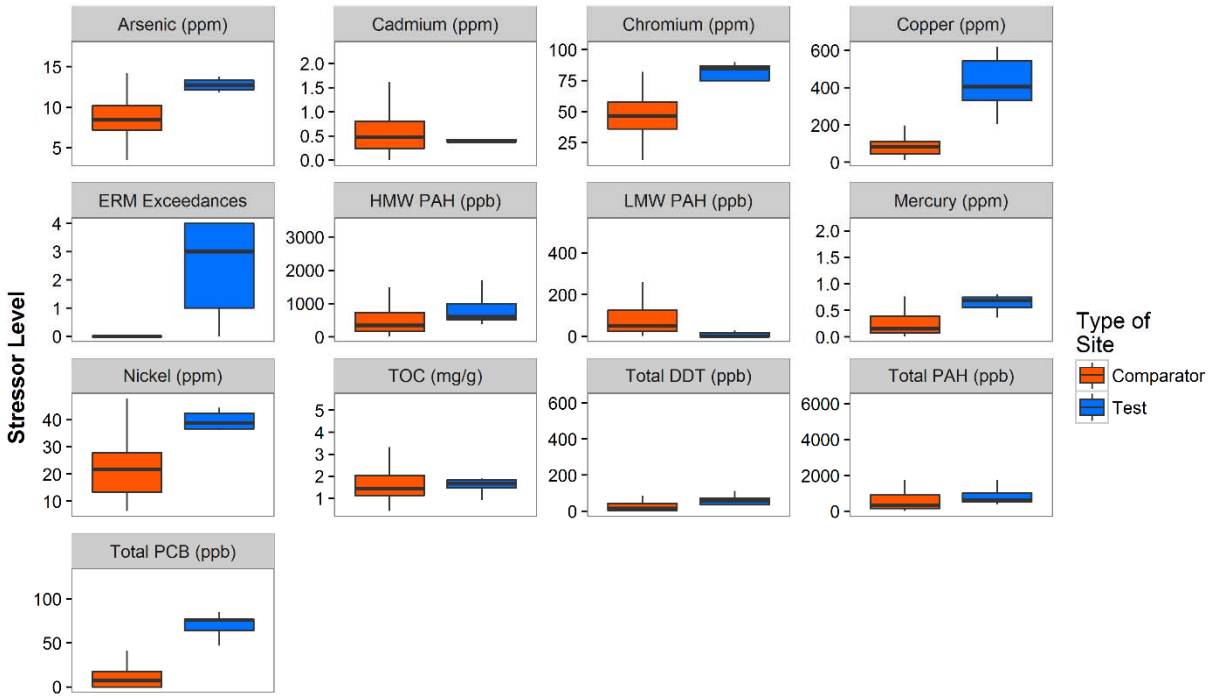

Figure 5-10. Box-and-whisker plots comparing stressor exposure at MdRH (test sites) to comparator sites with reference or low impact benthic community conditions.

Stressor Response

The logistic regression models for the comparator sites indicated that 8 of 13 individual sediment toxics components were present at high enough concentrations in MdRH to potentially lead to the impaired benthic communities observed there (Figure 5-11). Statistically significant models ($\alpha = 0.1$) were not obtained for cadmium, PCBs, PAHs, and DDTs, so the observed concentrations at MdRH were scored as indeterminate for these constituents. The stressor-response analysis was scored as supporting evidence for sediment toxics as a causative agent for the macrobenthic community conditions at MdRH, given the number of supporting scores and the lack of any weakening scores (Table 5-6).

The model for sediment TOC from the comparator sites was not statistically significant ($\alpha = 0.1$), so this analysis was scored as indeterminate evidence for eutrophication to be a causative agent (Table 5-6). Limited data availability prevented the consideration of physical disturbance or low dissolved oxygen as causative agents in this type of analysis (i.e., scored as NE).

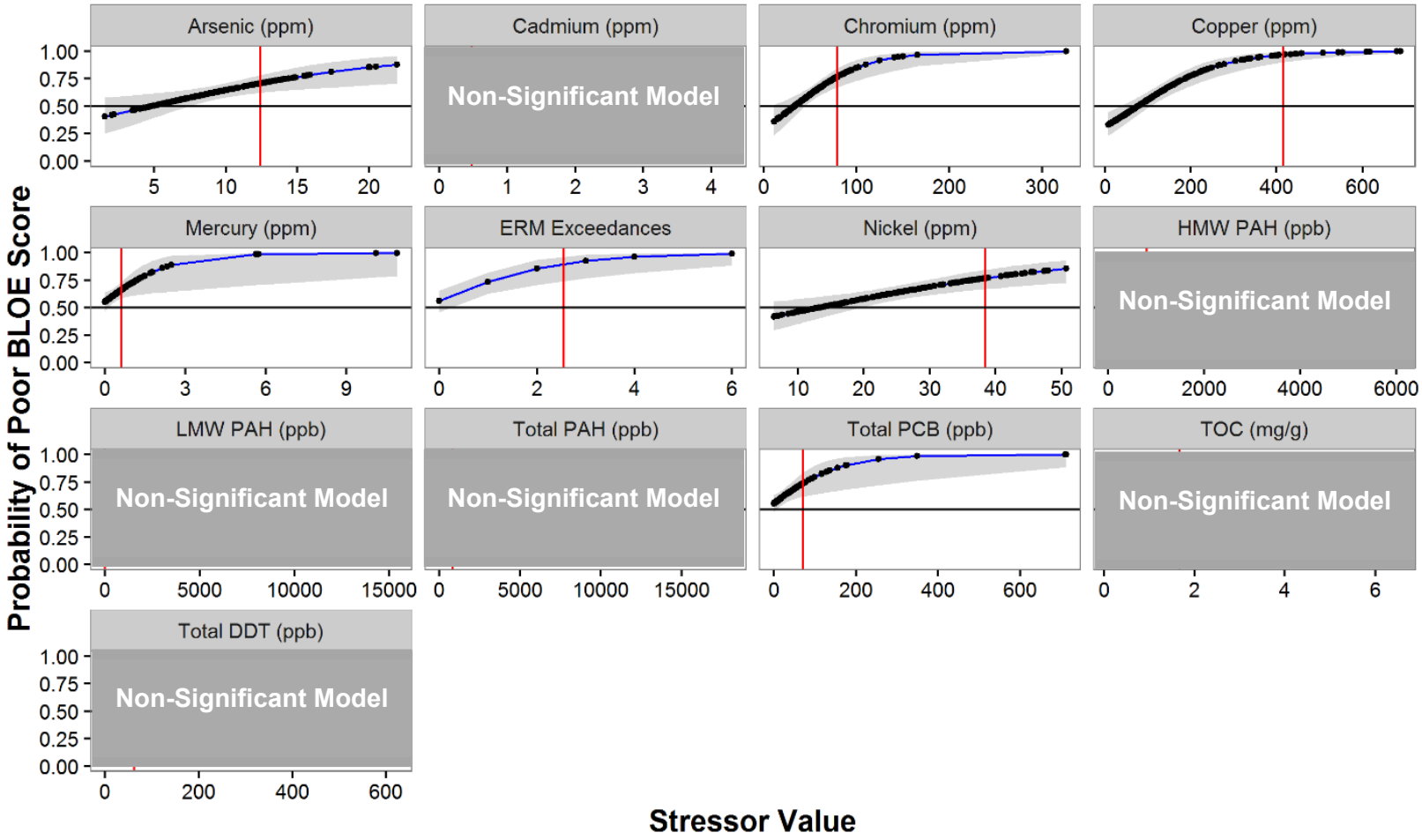

Figure 5-11. Logistic regression plots modelling the probability of observing benthic communities in moderate or high impact condition as a function of stressor concentration among regional comparator sites. The black dots represent the comparator samples, the blue lines represent the modeled probability (regression line), and the grey ribbons represent the 95% confidence intervals around regression. The vertical red lines represent the mean concentration at MdRH in 2016. The horizontal black line indicates the 50% probability threshold used in scoring the data.

Table 5-6. Scoring results from the stressor response analysis. Scores can be supporting evidence (+), weakening evidence (-), indeterminate evidence (0), or no evidence (NE) towards the candidate stressor as a causative agent for the impaired macrobenthic communities at MdRH. An * indicates models that were not statistically significant which were therefore scored 0.

Candidate Stressor	Components	Component Score	Candidate Score
Toxics			+
	Arsenic	+	
	Cadmium*	0	
	Chromium	+	
	Copper	+	
	Mercury	+	
	ERM Exceedances	+	
	Nickel	+	
	HMW PAH*	0	
	LMW PAH*	0	
	Total PAH*	0	
	Total PCB*	+	
	Total DDT*	0	
Eutrophication			0
	TN	NE	
	TOC*	0	
Physical Disturbance		NE	NE
Low Dissolved Oxygen		NE	NE

5.4 Summary and Conclusions

All of the samples collected from MdRH in July 2016 contained moderately or highly degraded macrobenthic communities. There are a variety of different anthropogenically-produced stressors that could lead to degraded macrobenthic communities, but they can be generally classified as either sediment toxics, eutrophication, physical disturbance or low dissolved oxygen. Multiple types of causal analyses were applied in this study to discern the relative likelihood of any one of these stressor classes as a potential causative agent responsible for the degraded macrobenthic community in MdRH.

Both the ecological and statistical analyses suggest that toxic substances in MdRH sediments were at concentrations that may have led to the impaired macrobenthic communities that were observed. The overall low species richness and abundance paired with relatively high trophic diversity was indicative of a chronic, non-catastrophic type of stressor exposure. The near absence of any crustacean taxa could be indicative of exposure to insecticides (e.g., synthetic pyrethroids). Compared to ecologically similar sites from across the region, MdRH had elevated concentrations of a variety of heavy metals and some organic compounds. Furthermore, based upon patterns observed at the ecologically similar sites, MdRH sediment contaminant concentrations could be expected to negatively impact resident macrobenthic fauna. Given the concordance in results from different types of analyses and the confidence in those results (Table

5-7), it is likely that sediment toxics were responsible for the impaired benthic community observed within MdRH.

The near absence of crustacean taxa could also indicate that low dissolved oxygen concentrations in the harbor may possibly be responsible for the impaired benthic communities. However, other lines of evidence suggest that this hypothesis is unlikely. For example, the lack of crustaceans was common among all sample locations within harbor, and those from the main channel – where there would presumably be better water/oxygen circulation – were not appreciably different from the back basin samples. Furthermore, there were limited dissolved oxygen data available from MdRH and none of the records indicated low oxygen conditions. This general lack of data within the harbor and its comparator sites prevented a complete and robust test of low dissolved oxygen as a stressor. Additional data from MdRH on daily variations of dissolved oxygen across spring and neap tidal cycles would allow for a more definitive diagnosis of the influence of low dissolved oxygen on benthic community condition.

There was no evidence from different ecological and statistical analyses that eutrophication or physical disturbance were responsible for the impaired macrobenthic community observed in MdRH. Eutrophication-indicative taxa were absent from the harbor and TOC concentrations in MdRH were similar to those from other embayments where benthic communities were in better condition. The mature, trophically diverse, communities observed in the harbor were indicative of lack of any recent physical disturbances that could have led to observed impaired macrobenthic community.

The specific chemical toxics associated with the macrobenthic community degradation in MdRH cannot be determined from the available information. Stressor identification methods for the macrobenthos are not available, and so it cannot be conclusively determined whether or not the constituents affecting macrobenthic community condition are the same as those causing sediment toxicity in MdRH. However, the near absence of amphipods and other crustaceans in the sediment samples strongly suggests that the seasonally-associated constituents causing amphipod mortality in the laboratory toxicity tests are also important stressors to the macrobenthic community. Whether additional chemical toxics are impacting the macrobenthic community is unknown at this time and cannot be determined without additional research to develop more specific stressor identification methods.

Table 5-7. Results summary from the multiple types of analysis used to assess the four broad classes of stressors as potential causative agents responsible for the impaired macrobenthic community observed in MdRH.

Candidate Stressor	Analysis Types			
	Community Composition	**Indicator Taxa**	**Stressor Exposure**	**Stressor Response**
Toxics	Supporting Evidence	Indeterminate Evidence	Supporting Evidence	Supporting Evidence
Eutrophication	Weakening Evidence	Indeterminate Evidence	Weakening Evidence	Indeterminate Evidence
Physical Disturbance	Weakening Evidence	No Evidence	No Evidence	No Evidence
Low Dissolved Oxygen	Supporting Evidence	No Evidence	No Evidence	No Evidence

6. MANAGEMENT ACTIONS

6.1 Toxicity Testing Findings

The level of toxicity is low and episodic with highest occurrence of a toxic response occurring in winter/spring; this pattern has been present for the last 5 years. The time of year and the sample locations where the strongest toxic effects have been observed historically were included in the study. However, due to the low level and episodic nature of toxicity, the ability to determine a specific contaminant class(es) responsible for the observed toxic effects was not possible for the available evaluation methods. Seasonal toxic effects are likely due to the presence of temporary stressors. Physical variables like temperature and overlying water, which are expected to vary in the harbor by season, are controlled in the laboratory. Therefore, the toxic effects are likely due to short-term stressors that may degrade, dilute, or leave the system by summer. Because those stressors come and go, the observed toxic effect may be linked to external effluent sources and not the static sediment condition within the harbor.

6.2 Benthic Community Findings

The SQO assessment confirmed the benthic community is currently categorized as impaired. MdRH is a manmade basin where both contaminant concentrations and percentage of fine grained materials is higher than other embayments in the Southern California Bight. The size and shape of the harbor, lack of riverine inputs, proximity of the harbor mouth, along with the presence of elevated fine grained materials suggest the harbor may have unique hydrodynamics that may explain the occurrence of specific benthic assemblages.

6.3 TMDL Compliance

The MdRH Toxics TMDL addresses multiple classes of chemicals and types of resource level impacts: water column quality impacts (e.g., aquatic toxicity), sediment quality impacts to benthic health (e.g., sediment toxicity and benthic community impairments), and sediment quality impacts to human health (e.g., ingestion of fish). From the TMDL implementation plan, attainment of water, sediment, and tissue quality will be achieved through management actions such as source reduction, source control, and focused remediation. Different management actions may be required to address each of the classes of chemicals for each type of impact.

TMDL implementation planning may be most efficient if the development of management alternatives considers all the toxics-related impairments in an integrated manner, including benthic community health, fish tissue contamination, and water column copper. Such an approach will allow for the evaluation and prioritization of management efforts that have the greatest impact to the overall water quality. This approach is recommended to ensure that management actions are ecologically beneficial and logistically and economically feasible.

6.3.1 Current Management Actions

It is anticipated that ongoing source reduction will continue as the responsible parties implement additional control measures. To date, the responsible parties have implemented or have plans to implement the following projects adjacent to MdRH:

- The Oxford Basin Multi-Use Enhancement Project - This project included dewatering the basin to excavate the contaminated sediment at the bottom of the basin and also included the construction of a circulation berm to increase oxygenation in Oxford Basin and Basin E. The project serves approximately 600 acres of the upstream watershed and was completed in May 2016.

- Parking Lot 5 BMP Project – This project included the construction of four bio-filtration modular wetlands units to treat the runoff from the 2.3 acre parking lot. The project was completed in September 2014.

- Parking Lot 7 BMP Project – This project included the construction of six bio-retention units to retain the runoff from the 1 acre parking lot. The project was completed in September 2014.

- Parking Lot 9 BMP Project – This project included the construction of four bio-filtration modular wetlands units to treat the runoff from the 1.5 acres parking lot. The project was completed in December 2016.

- Marina del Rey Library BMP Project – This project includes the construction of porous concrete and a catch basin filter BMP to address the runoff from the 0.6 acre parking lot. The project is expected to be completed in spring 2017.

- Marina del Rey Back Basins Water Quality Catch Basin Project – This project includes the construction of a water quality BMPs to address all 13 catch basins in the back basins. The project is currently in the planning phase.

- Marina del Rey Front Basins Water Quality Catch Basin Project – This project includes the construction of water quality BMPs to address all the catch basins in the front basins. The project is currently in the planning phase.

- Culver City has partnered with Costco Wholesale to construct a diversion project on Washington Boulevard that would capture the 85th percentile 24-hour storm event from approximately 40 acres of drainage area. Additional efforts are underway to expand the project to accommodate an additional 25 acres of drainage that are within the City of Los Angeles. The project will be designed by late 2017 with the goal of construction completed by late 2018.

Through the continued implementation of the Los Angeles Municipal Separate Storm Sewer System (MS4) Permit (Order No. R4-2012-0175), additional possible control measures and management actions may educate and refine the approach to addressing the TMDL requirements. Additionally, the MdRH CIMP may provide the opportunity for the analysis of additional contaminants and the use of lower detection limits may provide further information on potential contaminants of concern.

7. KEY FINDINGS

- Results of the sediment quality survey indicated that none of the stations met the SQO for protection of sediment-dwelling aquatic life. These results were largely driven by the presence of moderate and high impacts in the chemistry and benthic community lines of evidence.
- Sediment quality in MdRH appears to be stable, with little evidence of change in the last 10 years.
- Sediment toxicity to amphipods varied between the 2016 sampling periods, with widespread toxicity in winter and no toxicity in summer. This seasonal pattern is typical of MdRH and indicative of wet weather as the likely source of toxicity.
- No specific stressor could be identified as a cause for sediment toxicity. Potential stressors include pyrethroid or PAH toxicity, or unmeasured toxics from runoff inputs.
- TMDL chemical target concentrations were not met throughout MdRH in both sampling periods. However, the TIE characterization, bioavailability, and confirmation analyses demonstrated that none of the TMDL constituents were present at concentrations expected to cause sediment toxicity.
- The TMDL targets show little correspondence to sediment toxicity occurrence in MdRH or other Southern California embayments. The metals TMDL targets are likely unattainable due to being set at concentrations at or below background for the fine-grained sediment typical of embayments in Southern California.
- Benthic community impairment was present at all Harbor locations sampled, and indicative of chronic exposure to stressors. It is likely that sediment toxics are responsible. Methods are not available to identify the specific constituents responsible for benthic community impacts and so no conclusion can be drawn regarding the influence of specific TMDL contaminants of potential concern on benthic community health. It is likely, however, that the constituents influencing sediment toxicity are also important causes of benthic community impairment.
- Some evidence suggests low dissolved oxygen as a cause of impairment but there was not enough data to be conclusive. There was no evidence for physical disturbance or eutrophication as a likely cause of benthic community impairment.

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9. APPENDICES

Appendix A-1. Sediment organic contaminant concentrations for the July 2016 sampling event.

Constituent µg/kg dw	T1	T5	S1	S2	S3	S4	S5	S6	S7	S7-dup	S8	Detection Limit
PBDE 15	< 0.976	< 0.976	< 0.976	< 0.976	< 0.976	< 0.976	< 0.976	< 0.976	< 0.976	< 0.976	< 0.976	0.976
PBDE 28	< 0.24	< 0.24	≤ 0.136	< 0.24	< 0.24	< 0.24	< 0.24	< 0.24	< 0.24	< 0.24	< 0.24	0.24
PBDE 33	< 0.169	< 0.169	< 0.169	< 0.169	< 0.169	< 0.169	< 0.169	< 0.169	< 0.169	< 0.169	< 0.169	0.169
PBDE 47	0.4	0.699	3.86	0.887	1.64	1.04	4.58	1.36	1.35	0.836	1	0.143
PBDE 49	0.161	0.492	2.9	0.605	1.06	0.647	0.548	0.636	1.62	1.1	0.732	0.145
PBDE 66	< 0.234	< 0.234	0.25	< 0.234	< 0.234	< 0.234	≤ 0.23	< 0.234	< 0.234	< 0.234	< 0.234	0.234
PBDE 75	< 0.103	< 0.103	< 0.103	< 0.103	< 0.103	< 0.103	< 0.103	< 0.103	< 0.103	< 0.103	< 0.103	0.103
PBDE 99	0.284	0.571	3.89	0.998	1.52	0.781	8.7	1.75	1.41	0.741	0.957	0.104
PBDE 100	0.096	0.201	1.08	0.244	0.422	0.216	1.66	0.428	0.291	0.23	0.271	0.079
PBDE 153	0.178	0.317	1.52	0.978	1.07	0.547	1.57	0.611	1.33	1.09	0.524	0.039
PBDE 154	0.084	0.141	0.706	0.185	0.315	0.174	0.869	0.289	0.374	0.174	0.193	0.034
PBDE 155	≤ 0.031	0.054	0.118	≤ 0.036	0.067	< 0.047	0.079	0.054	0.207	≤ 0.042	≤ 0.038	0.047
PBDE 183	< 0.084	< 0.084	0.372	0.137	< 0.084	< 0.084	0.119	0.093	0.225	< 0.084	< 0.084	0.084
Acenaphthene	< 10.9	< 10.9	< 10.9	< 10.9	< 10.9	< 10.9	< 10.9	< 10.9	< 10.9	< 10.9	< 10.9	10.9
Acenaphthylene	< 10.5	< 10.5	< 10.5	< 10.5	< 10.5	< 10.5	< 10.5	< 10.5	< 10.5	< 10.5	< 10.5	10.5
Anthracene	< 15.1	< 15.1	< 15.1	< 15.1	< 15.1	< 15.1	< 15.1	< 15.1	< 15.1	< 15.1	< 15.1	15.1
Benz[a]anthracene	< 25.4	26.8	95.5	≤ 19.1	41	29.6	<= 21.5	<= 23.6	68.2	54.6	28.3	25.4
9,10-Diphenylanthracene	< 9.7	< 9.7	< 9.7	< 9.7	< 9.7	< 9.7	< 9.7	< 9.7	< 9.7	< 9.7	< 9.7	9.7
Biphenyl	< 6.81	< 6.81	< 6.81	< 6.81	< 6.81	< 6.81	< 6.81	< 6.81	< 6.81	< 6.81	< 6.81	6.81
Chrysene	≤ 25.2	54.2	177	41.6	79.6	66.4	46.4	53.4	136	122	60.3	26.2
Fluoranthene	22	44	159	31.6	70.2	52.2	37.8	46.7	125	114	53	3.97
Benzo[b]fluoranthene	48	84	259	60.1	122	90	99.6	81.4	187	162	95.6	12.1
Benzo[k]fluoranthene	17.7	32.6	94.1	21.4	38.4	31.6	83.1	26.3	66.8	47.9	30.5	14.2
Fluorene	< 18.2	< 18.2	< 18.2	< 18.2	< 18.2	< 18.2	< 18.2	< 18.2	< 18.2	< 18.2	< 18.2	18.2
11H-Benzo[b]fluorene	< 20.5	< 20.5	23.2	< 20.5	< 20.5	< 20.5	< 20.5	< 20.5	≤ 14.5	≤ 11	< 20.5	20.5

Appendix A-1. Continued.

Constituent µg/kg dw	T1	T5	S1	S2	S3	S4	S5	S6	S7	S7-dup	S8	Detection Limit
Naphthalene	< 4.37	< 4.37	< 4.37	< 4.37	< 4.37	< 4.37	< 4.37	< 4.37	< 4.37	< 4.37	< 4.37	4.37
1-Methylnaphthalene	< 10.7	< 10.7	< 10.7	< 10.7	< 10.7	< 10.7	< 10.7	< 10.7	< 10.7	< 10.7	< 10.7	10.7
2-Methylnaphthalene	< 13.3	< 13.3	< 13.3	< 13.3	< 13.3	< 13.3	< 13.3	< 13.3	< 13.3	< 13.3	< 13.3	13.3
Perylene	57.6	69.1	125	40.4	68.5	55.2	42.4	43.6	70.5	59.8	49.1	15
Benzo[g,h,i]perylene	59.8	103	290	71	140	107	76.2	94	180	140	110	22
2,6-Dimethylnaphthalene	< 22.2	< 22.2	< 22.2	< 22.2	< 22.2	< 22.2	< 22.2	< 22.2	< 22.2	< 22.2	< 22.2	22.2
2,3,5-Trimethylnaphthalene	< 12	< 12	< 12	< 12	< 12	< 12	< 12	< 12	< 12	< 12	< 12	12
Phenanthrene	< 17.2	< 17.2	28.9	< 17.2	≤ 13.2	< 17.2	< 17.2	< 17.2	19.6	17.4	< 17.2	17.2
1-Methylphenanthrene	< 13.7	< 13.7	< 13.7	< 13.7	< 13.7	< 13.7	< 13.7	< 13.7	< 13.7	< 13.7	< 13.7	13.7
2-Methylphenanthrene	< 13	< 13	≤ 9.06	< 13	< 13	< 13	< 13	< 13	< 13	< 13	< 13	13
3,6,Dimethylphenanthrene	< 12.8	< 12.8	< 12.8	< 12.8	< 12.8	< 12.8	< 12.8	< 12.8	< 12.8	< 12.8	< 12.8	12.8
Pyrene	33	54.8	181	39.3	81.8	62.7	47.3	57	132	118	63.5	6.45
Benzo[a]pyrene	26.8	52.8	164	38.8	76.2	59	43.9	51	122	99.4	58.5	20.3
Benzo[e]pyrene	28.3	47.3	135	32.2	65.3	48.4	36.5	44.7	93.7	80.6	49.2	12.1
PCB8	< 24.2	< 24.2	< 24.2	< 24.2	< 24.2	< 24.2	< 24.2	< 24.2	< 24.2	< 24.2	< 24.2	24.2
PCB18	< 30.8	< 30.8	< 30.8	< 30.8	< 30.8	< 30.8	< 30.8	< 30.8	< 30.8	< 30.8	< 30.8	30.8
PCB28	< 30.2	< 30.2	< 30.2	< 30.2	< 30.2	< 30.2	< 30.2	< 30.2	< 30.2	< 30.2	< 30.2	30.2
PCB37	< 33.3	< 33.3	< 33.3	< 33.3	< 33.3	< 33.3	< 33.3	< 33.3	< 33.3	< 33.3	< 33.3	33.3
PCB44	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	19.8
PCB49	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	< 20	20
PCB52	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	< 19.8	19.8
PCB66	≤ 2.89	5.09	≤ 2.99	≤ 2.32	3.52	3.09	≤ 3.04	3.33	3.67	3.94	3.57	3.08
PCB70	2.46	3.31	2.87	≤ 1.93	2.45	≤ 2.2	≤ 1.81	≤ 1.93	2.73	2.67	≤ 2.21	2.35
PCB74	< 4.17	< 4.17	< 4.17	< 4.17	< 4.17	< 4.17	< 4.17	< 4.17	< 4.17	< 4.17	< 4.17	4.17
PCB77	≤ 0.233	0.524	0.526	≤ 0.296	0.487	0.398	≤ 0.295	≤ 0.335	0.436	0.47	0.439	0.375
PCB81	< 0.311	< 0.311	< 0.311	< 0.311	< 0.311	< 0.311	< 0.311	< 0.311	< 0.311	< 0.311	< 0.311	0.311
PCB87	1.67	2.09	3.09	1.35	2.29	1.63	≤ 1.21	1.47	1.77	1.6	1.67	1.28

Appendix A-1. Continued.

Constituent µg/kg dw	T1	T5	S1	S2	S3	S4	S5	S6	S7	S7-dup	S8	Detection Limit
PCB99	2.82	5.6	3.75	2.59	3.76	3.79	4.22	3.72	4.95	4.37	4.6	1.82
PCB101	4.75	7.95	7.85	4.22	6.92	5.72	5.81	5.36	6.34	6.62	6.44	0.749
PCB105	1.8	3.36	4.05	2.02	3.36	2.53	1.95	2.27	2.65	2.77	2.75	0.131
PCB110	5.16	8.09	8.73	4.8	7.52	5.84	5.37	5.21	6	6.55	6.49	0.81
PCB114	0.255	0.547	0.616	0.306	0.496	0.396	0.41	0.397	0.467	0.487	0.46	0.08
PCB118	4.3	8.48	8.07	4.45	7.37	6.04	5.85	5.67	6.48	6.93	6.79	0.089
PCB119	< 0.484	0.549	≤ 0.478	≤ 0.246	≤ 0.406	≤ 0.39	≤ 0.408	≤ 0.331	≤ 0.414	≤ 0.475	0.488	0.484
PCB123	0.678	1.34	1.17	0.673	1.12	0.959	0.968	0.912	1.06	1.12	1.09	0.107
PCB126	< 0.091	≤ 0.078	0.104	≤ 0.048	≤ 0.089	≤ 0.058	≤ 0.047	≤ 0.052	< 0.091	≤ 0.065	≤ 0.076	0.091
PCB128	1.02	2.3	2.7	1.31	2.21	1.7	1.58	1.7	2.35	1.89	1.97	0.09
PCB138	3.46	8.04	9.38	4.17	7.54	5.94	5.51	5.83	6.55	6.18	6.73	0.079
PCB149	2.94	6.87	8.03	3.75	6.38	4.89	4.78	5.35	5.53	5.45	5.64	0.517
PCB151	0.592	1.41	1.76	0.822	1.34	1.06	0.934	1.05	1.09	1.09	1.18	0.099
PCB153/168	4.21	9.96	10.6	5.09	8.81	7.17	7.71	7.23	11.7	8.13	8.42	0.034
PCB156	0.478	1.12	1.47	0.635	1.09	0.823	0.722	0.789	< 0.04	0.927	0.923	0.04
PCB157	0.125	0.287	0.316	0.161	0.268	0.216	0.212	0.208	< 0.047	0.24	0.243	0.047
PCB158	0.404	0.796	1.16	0.456	0.804	0.585	0.54	0.577	0.678	0.658	0.662	0.04
PCB167	0.229	0.532	0.6	0.278	0.479	0.39	0.394	0.379	0.522	0.442	0.443	0.04
PCB169	0.059	0.167	0.222	0.069	0.14	0.114	0.105	0.117	0.12	0.109	0.132	0.051
PCB170	0.758	2.29	2.77	1.2	1.96	1.48	1.52	1.48	1.59	1.58	1.7	0.042
PCB177	0.486	1.45	1.66	0.78	1.28	1.02	1.04	1.03	1.23	1.07	1.16	0.046
PCB180	1.87	5.04	6.51	2.75	4.48	3.38	3.39	3.45	1.14	3.54	3.92	0.035
PCB183	0.464	1.16	1.49	0.619	1.05	0.797	0.847	0.831	0.952	0.872	0.927	0.038
PCB187	1.25	3.3	3.59	1.72	2.82	2.32	2.44	2.43	2.63	2.54	2.77	0.039
PCB189	0.11	0.344	0.37	0.179	0.292	0.224	0.222	0.226	0.242	0.228	0.261	0.036
PCB194	0.513	1.56	1.77	0.77	1.29	0.971	1.04	1.02	1.1	1.04	1.18	0.03
PCB200	0.09	0.223	0.256	0.109	0.195	0.148	0.158	0.157	0.199	0.17	0.183	0.036

Appendix A-1. Continued.

Constituent µg/kg dw	T1	T5	S1	S2	S3	S4	S5	S6	S7	S7-dup	S8	Detection Limit
PCB201	0.782	2.21	2.67	1.14	1.96	1.48	1.49	1.5	1.61	1.53	1.82	0.038
PCB206	0.402	0.96	1.3	0.45	0.924	0.661	0.609	0.654	0.729	0.685	0.895	0.027
p,p'-DDT	< 10.2	< 10.2	< 10.2	< 10.2	< 10.2	< 10.2	< 10.2	< 10.2	< 10.2	< 10.2	< 10.2	10.2
p,p'-DDD	14.3	3.32	17.7	5.38	5.59	4.97	3.09	9.22	32.2	22.8	8.05	2.22
p,p'-DDE	45.8	34.4	47.3	22.9	38	26.9	25.1	41.8	69.1	78.4	46.4	1.88
o,p'-DDT	< 2.75	< 2.75	< 2.75	< 2.75	< 2.75	< 2.75	< 2.75	< 2.75	< 2.75	< 2.75	< 2.75	2.75
o,p'-DDD	3.27	1.35	2.86	1.12	2.14	1.37	≤ 0.861	2.39	4.91	7.67	2.36	1.02
o,p'-DDE	2.95	2.94	5.22	1.64	3.15	1.85	1.6	2.34	3.27	3.44	3.02	0.572
Aldrin	< 0.194	< 0.194	≤ 0.11	< 0.194	< 0.194	< 0.194	< 0.194	< 0.194	< 0.194	< 0.194	< 0.194	0.194
Dieldrin	≤ 0.339	0.528	1.24	≤ 0.377	0.678	≤ 0.314	≤ 0.234	≤0.381	< 0.382	< 0.382	≤ 0.335	0.382
Endrin	< 1.1	< 1.1	< 1.1	< 1.1	< 1.1	< 1.1	< 1.1	< 1.1	< 1.1	< 1.1	< 1.1	1.1
Chlorpyrifos	< 0.26	≤ 0.207	1.59	≤ 0.183	0.333	≤ 0.184	< 0.26	≤ 0.157	< 0.26	< 0.26	≤ 0.207	0.26
Chlordene	0.976	0.713	2.02	0.29	0.839	0.302	≤ 0.226	0.361	0.315	0.353	0.405	0.262
DDMU	≤ 3.18	≤ 2.48	5.59	< 3.54	3.66	≤ 2.3	< 3.54	≤ 2.13	3.8	≤ 3.22	≤ 2.68	3.54
Heptachlor Epoxide B	< 0.206	< 0.206	< 0.206	< 0.206	< 0.206	< 0.206	< 0.206	< 0.206	< 0.206	< 0.206	< 0.206	0.206
Cis-Nonachlor	1.07	1.86	4.56	1.14	2.71	1.52	1.07	1.34	1.44	1.45	1.7	0.05
Trans-Nonachlor	0.827	1.24	5.68	1.02	2.45	1.21	0.685	0.962	1.58	1.11	1.21	0.076
Fipronil	< 0.402	< 0.402	< 0.402	< 0.402	< 0.402	< 0.402	< 0.402	< 0.402	≤ 0.367	< 0.402	< 0.402	0.402
Fipronil desulfinyl	< 0.038	≤ 0.022	< 0.038	< 0.038	≤ 0.03	≤ 0.02	≤ 0.022	≤ 0.02	≤ 0.02	≤ 0.02	≤ 0.02	0.038
Fipronil sulfide	< 0.04	< 0.04	≤ 0.038	< 0.04	< 0.04	< 0.04	< 0.04	< 0.04	< 0.04	< 0.04	< 0.04	0.04
Fipronil sulfone	< 0.527	< 0.527	< 0.527	< 0.527	< 0.527	< 0.527	< 0.527	< 0.527	< 0.527	< 0.527	< 0.527	0.527
Bifenthrin	1.69	2.22	10.9	2.29	4.1	2.49	1.75	3.18	4.09	2.78	2.64	0.613
Cyfluthrin	0.486	0.713	5.65	0.603	1.23	1.58	0.647	2.34	1.21	1	1.31	0.216
Cypermethrin	≤ 0.198	0.251	1.92	0.387	0.731	0.392	≤ 0.133	< 0.201	0.485	0.361	0.363	0.201
Deltamethrin	0.672	1.37	3.92	3.98	3.7	1.34	2.01	1.09	2.93	3.79	2.23	0.535
Esfenvalerate	< 0.09	0.09	0.408	0.093	0.162	0.142	≤ 0.067	0.169	0.84	1.15	0.096	0.09
Fenpropathrin	< 0.427	< 0.427	< 0.427	< 0.427	< 0.427	< 0.427	< 0.427	< 0.427	< 0.427	< 0.427	< 0.427	0.427

Appendix A-1. Continued.

Constituent µg/kg dw	T1	T5	S1	S2	S3	S4	S5	S6	S7	S7-dup	S8	Detection Limit
Lamda-Cyhalothrin	< 0.467	< 0.467	2.1	< 0.467	< 0.467	0.497	< 0.467	< 0.467	< 0.467	< 0.467	< 0.467	0.467
Permethrin	< 4.76	< 4.76	10.9	< 4.76	< 4.76	< 4.76	< 4.76	< 4.76	< 4.76	< 4.76	< 4.76	4.76
Cis-Chlordane (Alpha)	0.85	1.26	5.82	0.953	2.5	1.2	0.641	1.1	1.37	1.05	1.26	0.124
Trans-Chlordane (Gamma)	1.39	1.65	7.96	1.39	3.32	1.61	0.857	1.27	2.96	1.39	1.58	0.066
Oxychlordane	< 0.263	< 0.263	< 0.263	< 0.263	< 0.263	< 0.263	< 0.263	< 0.263	< 0.263	< 0.263	< 0.263	0.263

Constituent µg/kg dw	T1	T5	S1	S2	S3	S4	S5	S6	S7	S7-dup	S8
ΣPBDEs	1.203	2.475	14.696	4.034	6.094	3.405	18.125	5.221	6.807	4.171	3.677
ΣPAHs	293.2	568.6	1731.7	376.4	783	602.1	513.2	498.1	1200.8	1015.7	598
ΣPCBs	44.135	96.949	99.45	46.867	84.605	65.762	59.821	64.347	76.515	75.898	75.946
ΣDDTs	66.32	42.01	73.08	31.04	48.88	35.09	29.79	55.75	109.48	112.31	59.83
ΣOCPs	71.433	49.261	107.54	35.833	65.37	40.932	33.043	60.783	120.945	117.663	65.985
ΣFipronils	0	0	0	0	0	0	0	0	0	0	0
ΣPyrethroids	2.848	4.644	35.798	7.353	9.923	6.441	4.407	6.779	9.555	9.081	6.639
ΣChlordanes	2.24	2.91	13.78	2.343	5.82	2.81	1.498	2.37	4.33	2.44	2.84

Appendix A-2. Sediment metals concentrations for the July 2016 sampling event.

Constituent (mg/kg dw)	T1	T5	S1	S2	S3	S4	S5	S6	S7	S7-dup	S8
Al	34087	45633	31901	24190	40822	37518	36896	47946	50367	46160	45441
As	10.27	16.37	12.73	8.80	13.09	11.85	12.61	13.33	13.55	12.18	13.74
Be	0.77	1.03	0.81	0.60	0.98	0.89	0.94	1.11	1.09	1.05	1.04
Cd	0.82	0.38	1.08	0.38	0.42	0.36	0.40	0.51	0.38	0.42	0.39
Cr	56.52	91.71	77.53	48.67	84.83	74.16	74.91	89.79	86.94	84.28	88.99
Cu	190.74	404.58	205.72	211.40	331.91	355.98	554.10	617.27	543.94	508.19	405.06
Fe	36182	52126	37989	27485	43686	42181	43908	52342	57226	49211	50508
Hg	0.36	1.07	0.43	0.37	0.55	0.58	0.81	0.69	0.76	0.71	0.75
Mn	262.27	368.44	304.71	250.60	338.79	326.65	310.39	369.04	344.41	350.02	347.92
Ni	30.69	44.59	36.20	25.03	38.88	36.70	38.27	44.38	42.42	40.79	43.10
Pb	54.85	116.72	110.78	54.89	108.74	81.58	83.09	94.77	100.43	95.59	108.22
Se	1.16	0.79	0.75	0.35	0.73	0.61	0.77	0.93	0.74	0.72	0.61
Sn	3.08	5.94	5.36	2.75	5.44	7.36	5.20	5.69	5.66	5.56	5.79
Ti	1768	2401	1879	1390	1778	2132	2055	2570	2570	2351	2459
V	102.01	134.59	96.15	69.39	114.68	108.29	111.89	136.64	130.75	129.42	131.50
Zn	294.79	481.27	391.01	280.02	423.57	404.70	502.29	606.70	520.19	481.20	462.89

Appendix B. Station assessment categories resulting from each possible MLOE combination.

Line of Evidence Combination	Chemistry Exposure	Benthic Community Disturbance	Toxicity	Station Assessment
1	Minimal	Reference	Nontoxic	Unimpacted
2	Minimal	Reference	Low	Unimpacted
3	Minimal	Reference	Moderate	Unimpacted
4	Minimal	Reference	High	Inconclusive
5	Minimal	Low	Nontoxic	Unimpacted
6	Minimal	Low	Low	Likely Unimpacted
7	Minimal	Low	Moderate	Likely Unimpacted
8	Minimal	Low	High	Possibly Impacted
9	Minimal	Moderate	Nontoxic	Likely Unimpacted
10	Minimal	Moderate	Low	Likely Unimpacted
11	Minimal	Moderate	Moderate	Possibly Impacted
12	Minimal	Moderate	High	Likely Impacted
13	Minimal	High	Nontoxic	Likely Unimpacted
14	Minimal	High	Low	Inconclusive
15	Minimal	High	Moderate	Possibly Impacted
16	Minimal	High	High	Likely ImpaTcted
17	Low	Reference	Nontoxic	Unimpacted
18	Low	Reference	Low	Unimpacted
19	Low	Reference	Moderate	Likely Unimpacted
20	Low	Reference	High	Possibly Impacted
21	Low	Low	Nontoxic	Unimpacted
22	Low	Low	Low	Likely Unimpacted
23	Low	Low	Moderate	Possibly Impacted
24	Low	Low	High	Possibly Impacted
25	Low	Moderate	Nontoxic	Likely Unimpacted

Appendix B. Continued.

Line of Evidence Combination	Chemistry Exposure	Benthic Community Disturbance	Toxicity	Station Assessment
26	Low	Moderate	Low	Possibly Impacted
27	Low	Moderate	Moderate	Likely Impacted
28	Low	Moderate	High	Likely Impacted
29	Low	High	Nontoxic	Likely Unimpacted
30	Low	High	Low	Possibly Impacted
31	Low	High	Moderate	Likely Impacted
32	Low	High	High	Likely Impacted
33	Moderate	Reference	Nontoxic	Unimpacted
34	Moderate	Reference	Low	Likely Unimpacted
35	Moderate	Reference	Moderate	Likely Unimpacted
36	Moderate	Reference	High	Possibly Impacted
37	Moderate	Low	Nontoxic	Unimpacted
38	Moderate	Low	Low	Possibly Impacted
39	Moderate	Low	Moderate	Possibly Impacted
40	Moderate	Low	High	Possibly Impacted
41	Moderate	Moderate	Nontoxic	Possibly Impacted
42	Moderate	Moderate	Low	Likely Impacted
43	Moderate	Moderate	Moderate	Likely Impacted
44	Moderate	Moderate	High	Likely Impacted
45	Moderate	High	Nontoxic	Possibly Impacted
46	Moderate	High	Low	Likely Impacted
47	Moderate	High	Moderate	Likely Impacted
48	Moderate	High	High	Likely Impacted
49	High	Reference	Nontoxic	Likely Unimpacted
50	High	Reference	Low	Likely Unimpacted
51	High	Reference	Moderate	Inconclusive
52	High	Reference	High	Likely Impacted
53	High	Low	Nontoxic	Likely Unimpacted

Appendix B. Continued.

Line of Evidence Combination	Chemistry Exposure	Benthic Community Disturbance	Toxicity	Station Assessment
54	High	Low	Low	Possibly Impacted
55	High	Low	Moderate	Likely Impacted
56	High	Low	High	Likely Impacted
57	High	Moderate	Nontoxic	Likely Impacted
58	High	Moderate	Low	Likely Impacted
59	High	Moderate	Moderate	Clearly Impacted
60	High	Moderate	High	Clearly Impacted
61	High	High	Nontoxic	Likely Impacted
62	High	High	Low	Likely Impacted
63	High	High	Moderate	Clearly Impacted
64	High	High	High	Clearly Impacted

Appendix C-1. Sediment organic contaminant concentrations for the January 2016 sampling event.

Constituent µg/kg dw	T1	T1-TIE	T5	T5-TIE	Detection Limit
PBDE 15	< 1.32	< 1.32	< 1.32	< 1.32	1.32
PBDE 28	< 0.198	< 0.198	< 0.198	< 0.198	0.198
PBDE 33	< 0.174	< 0.174	< 0.174	< 0.174	0.174
PBDE 47	0.32	0.358	0.564	0.576	0.161
PBDE 49	≤ 0.116	≤ 0.16	0.332	0.386	0.151
PBDE 66	< 0.248	< 0.248	< 0.248	< 0.248	0.248
PBDE 75	< 0.109	< 0.109	< 0.109	< 0.109	0.109
PBDE 99	0.208	0.344	0.502	0.578	0.122
PBDE 100	≤ 0.07	≤ 0.08	0.15	0.148	0.087
PBDE 153	0.06	0.09	0.198	0.234	0.052
PBDE 154	≤ 0.03	≤ 0.04	0.084	0.1	0.043
PBDE 155	< 0.053	< 0.053	< 0.053	< 0.053	0.053
PBDE 183	< 0.106	< 0.106	< 0.106	< 0.106	0.106
Acenaphthene	< 9.3	< 9.3	< 9.3	< 9.3	9.3
Acenaphthylene	< 10.3	< 10.3	< 10.3	< 10.3	10.3
Anthracene	< 17.8	< 17.8	< 17.8	< 17.8	17.8
Benz[a]anthracene	≤ 12.3	≤ 6.8	≤ 15.3	≤ 15.3	18.9
9,10-Diphenylanthracene	< 10.4	< 10.4	< 10.4	< 10.4	10.4
Biphenyl	< 10.4	< 10.4	< 10.4	< 10.4	10.4
Chrysene	25.7	18.3	29.9	34.7	18.1
Fluoranthene	22.3	11.7	25.3	27.2	7.9
Benzo[b]fluoranthene	31.8	21.7	45.4	45.9	8.7
Benzo[k]fluoranthene	13.4	7.7	16	16.6	6.9
Fluorene	< 18.1	< 18.1	< 18.1	< 18.1	18.1
11H-Benzo[b]fluorene	< 22.4	< 22.4	< 22.4	< 22.4	22.4
Naphthalene	< 4.7	< 4.7	< 4.7	< 4.7	4.7
1-Methylnaphthalene	< 8.8	< 8.8	< 8.8	< 8.8	8.8
2-Methylnaphthalene	< 13.9	< 13.9	< 13.9	< 13.9	13.9
Perylene	24.1	18.4	37.9	35.6	16
Benzo[g,h,i]perylene	34.8	25.1	46.3	48.9	18.8
2,6-Dimethylnaphthalene	< 25.1	< 25.1	< 25.1	< 25.1	25.1
2,3,5-Trimethylnaphthalene	< 11.1	< 11.1	< 11.1	< 11.1	11.1
Phenanthrene	≤ 6.66	< 12.4	≤ 7.49	≤ 7.85	12.4
1-Methylphenanthrene	< 11.2	< 11.2	< 11.2	< 11.2	11.2
2-Methylphenanthrene	< 13.7	< 13.7	< 13.7	< 13.7	13.7
3,6,Dimethylphenanthrene	< 10	< 10	< 10	< 10	10
Pyrene	27.1	15.5	31.2	33.3	6.4
Benzo[a]pyrene	23.7	13.9	26.4	27.5	13.5
Benzo[e]pyrene	19.1	12.7	25.6	26.5	8.4

Appendix C-1. Continued.

Constituent µg/kg dw	T1	T1-TIE	T5	T5-TIE	Detection Limit
PCB8	< 24.1	< 24.1	< 24.1	< 24.1	24.1
PCB18	< 19.8	< 19.8	< 19.8	< 19.8	19.8
PCB28	< 21.8	< 21.8	< 21.8	< 21.8	21.8
PCB37	< 12.2	< 12.2	< 12.2	< 12.2	42.188
PCB44	< 18.8	< 18.8	< 18.8	< 18.8	18.8
PCB49	< 15.9	< 15.9	< 15.9	< 15.9	15.9
PCB52	< 15.4	< 15.4	< 15.4	< 15.4	15.4
PCB66	≤ 1.54	≤ 1.82	4.74	5	1.842
PCB70	≤ 1.28	≤ 1.43	2.98	3.03	1.44
PCB74	< 2.64	< 2.64	≤ 1.602	≤ 1.584	2.64
PCB77	≤ 0.156	≤ 0.18	0.522	0.55	0.347
PCB81	< 0.29	< 0.29	< 0.29	< 0.29	0.29
PCB87	0.814	0.936	1.73	1.89	0.762
PCB99	1.47	1.68	4.24	4.49	1.085
PCB101	2.58	3.17	7.03	7.27	0.453
PCB105	1.13	1.36	3.26	3.48	0.09
PCB110	2.79	3.38	6.99	7.44	0.501
PCB114	0.154	0.178	0.476	0.492	0.052
PCB118	2.668	3.15	8.13	8.59	0.062
PCB119	≤ 0.144	≤ 0.184	0.462	0.496	0.271
PCB123	0.398	0.476	1.22	1.31	0.07
PCB126	≤ 0.028	≤ 0.034	0.082	0.09	0.079
PCB128	0.658	0.736	2.042	2.19	0.051
PCB138	2.21	2.56	7.03	7.49	0.046
PCB149	1.87	2.16	5.93	6.51	0.272
PCB151	0.35	0.422	1.17	1.28	0.051
PCB153/168	2.54	2.95	8.61	9.13	0.019
PCB156	0.31	0.366	1.05	1.09	0.028
PCB157	0.098	0.108	0.274	0.284	0.03
PCB158	0.252	0.298	0.732	0.794	0.023
PCB167	0.142	0.172	0.476	0.498	0.026
PCB169	0.04	0.046	0.128	0.128	0.038
PCB170	0.476	0.55	1.9	2.008	0.021
PCB177	0.288	0.346	1.2	1.29	0.025
PCB180	1.09	1.2	4.13	4.44	0.018
PCB183	0.274	0.328	0.966	1.03	0.019
PCB187	0.706	0.84	2.65	2.98	0.019
PCB189	0.066	0.088	0.258	0.28	0.019
PCB194	0.302	0.382	1.17	1.25	0.015

Appendix C-1. Continued.

Constituent µg/kg dw	T1	T1-TIE	T5	T5-TIE	Detection Limit
PCB200	0.052	0.068	0.176	0.192	0.017
PCB201	0.456	0.594	1.73	1.87	0.018
PCB206	0.216	0.302	0.71	0.738	0.012
p,p'-DDT	< 15.7	< 15.7	< 15.7	< 15.7	15.714
p,p'-DDD	8.59	10.1	3.21	3.37	1.932
p,p'-DDE	24.3	27	30.9	32.7	1.494
o,p'-DDT	< 3.63	< 3.63	< 3.63	< 3.63	3.626
o,p'-DDD	1.88	2.63	≤ 0.634	0.866	0.816
o,p'-DDE	1.42	1.55	2.65	2.53	0.566
Aldrin	< 0.15	< 0.15	< 0.15	< 0.15	0.15
Dieldrin	≤ 0.246	0.572	0.408	3.15	0.301
Endrin	< 0.99	< 0.99	< 0.99	< 0.99	0.99
Chlorpyrifos	< 0.372	≤ 0.36	< 0.372	≤ 0.272	0.372
Chlordene	0.496	0.606	0.546	0.628	0.234
DDMU	≤ 1.78	≤ 1.8	≤ 2.49	≤ 3.05	5
Heptachlor Epoxide B	< 0.163	< 0.163	< 0.163	< 0.163	0.163
Cis-Nonachlor	< 0.03	< 0.03	< 0.03	< 0.03	0.03
Trans-Nonchlor	0.634	0.68	0.996	1.11	0.049
Fipronil	< 0.524	< 0.524	< 0.524	< 0.524	0.524
Fipronil desulfinyl	≤ 0.014	≤ 0.018	≤ 0.024	≤ 0.022	0.036
Fipronil sulfide	< 0.033	< 0.033	< 0.033	< 0.033	0.033
Fipronil sulfone	< 0.802	< 0.802	< 0.802	< 0.802	0.802
Bifenthrin	1.07	1.23	1.94	2.27	0.625
Cyfluthrin	< 0.224	< 0.224	2.2	2.47	0.224
Cypermethrin	< 0.209	< 0.209	< 0.209	< 0.209	0.209
Deltamethrin	≤ 0.284	≤ 0.26	1.57	2.73	0.571
Esfenvalerate	≤ 0.046	≤ 0.048	0.11	≤ 0.084	0.108
Fenpropathrin	< 0.35	< 0.35	< 0.35	< 0.35	0.35
Lamda-Cyhalothrin	< 0.75	< 0.75	< 0.75	< 0.75	0.75
Permethrin	< 3.44	< 3.44	< 3.44	< 3.44	3.443
Cis-Chlordane (Alpha)	0.714	0.72	1.09	1.224	0.087
Trans-Chlordane (Gamma)	0.972	1.08	1.39	1.59	0.047
Oxychlordane	< 0.195	< 0.195	< 0.195	< 0.195	0.195

Appendix C-1. Continued.

Chemical class concentrations µg/kg dw	T1	T1-TIE	T5	T5-TIE
ΣPBDEs	0.588	0.792	1.83	2.02
ΣPAHs	222	145	284	296
ΣPCBs	24.4	28.8	84.2	89.6
ΣDDTs	36.19	41.28	36.76	39.466
ΣOCPs	39	44.9	41.2	47.2
ΣFipronils	0	0	0	0
ΣPyrethroids	1.07	1.23	5.82	7.47
ΣChlordanes	1.686	1.8	2.48	2.814

Appendix C-2. Sediment metals concentrations for the January 2016 sampling event.

Constituent (mg/kg dw)	T1	T1-TIE	T5	T5-TIE
Ag	0.53	0.61	1.36	1.32
Al	23202	24492	40230	46096
As	7.47	8.43	17.14	17.31
Ba	93.32	96.68	129.65	168.43
Be	0.65	0.65	1.16	1.28
Cd	0.46	0.54	0.43	0.44
Co	5.82	5.90	10.28	10.50
Cr	37.27	40.23	84.30	91.25
Cu	126.19	151.63	335.57	371.86
Fe	25447	27523	50790	52089
Hg	0.23	0.40	0.89	0.93
Mn	187.52	202.83	351.60	368.91
Mo	1.59	1.94	2.40	2.22
Ni	19.79	21.70	41.95	42.44
Pb	29.19	32.49	84.37	92.23
Sb	0.67	0.79	0.87	0.90
Se	0.71	0.83	0.69	0.53
Sn	2.07	3.28	4.80	6.56
Sr	78.73	86.29	99.70	97.21
Ti	1541	1632	2469	2763
Tl	0.27	0.25	0.46	0.51
V	72.46	78.69	128.12	141.39
Zn	194.25	221.43	414.46	451.41

Appendix D. TOC-normalized sediment concentrations used in ESB TU calculations.

		January 2016		July 2016									
Class	**Chemical (µg/g$_{OC}$)**	**T1**	**T5**	**T1**	**T5**	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**
Chlordanes	Chlordane	0.235	0.149	0.193	0.176	0.562	0.290	0.320	0.205	0.126	0.155	0.270	0.172
DDTs	4,4'-DDT	1.96	0.872	0.785	0.567	0.408	1.133	0.537	0.680	0.729	0.600	0.600	0.567
PAHs	Acenaphthene	1.16	0.517	0.838	0.606	0.436	1.21	0.574	0.727	0.779	0.641	0.641	0.606
	Acenaphthylene	1.29	0.572	0.808	0.583	0.420	1.17	0.553	0.700	0.750	0.618	0.618	0.583
	Anthracene	2.23	0.989	1.16	0.839	0.604	1.68	0.795	1.01	1.08	0.888	0.888	0.839
	Benzo(a) anthracene	2.36	0.850	1.95	1.49	3.82	2.12	2.16	1.97	1.54	1.39	4.01	1.57
	Benzo(a) pyrene	2.96	1.47	2.06	2.93	6.56	4.31	4.01	3.93	3.14	3.00	7.18	3.25
	Benzo(b) fluoranthene	3.98	2.52	3.69	4.67	10.36	6.68	6.42	6.00	7.11	4.79	11.00	5.31
	Benzo(k) fluoranthene	1.68	0.889	1.36	1.81	3.76	2.38	2.02	2.11	5.94	1.55	3.93	1.69
	Chrysene	3.21	1.66	1.94	3.01	7.08	4.62	4.19	4.43	3.31	3.14	8.00	3.35
	Fluoranthene	2.79	1.41	1.69	2.44	6.36	3.51	3.69	3.48	2.70	2.75	7.35	2.94
	Fluorene	2.26	1.01	1.40	1.01	0.728	2.02	0.958	1.21	1.30	1.07	1.07	1.01
	Naphthalene	0.59	0.261	0.336	0.243	0.175	0.486	0.230	0.291	0.312	0.257	0.257	0.243
	Phenanthrene	0.83	0.416	1.32	0.956	1.16	1.91	0.695	1.15	1.23	1.01	1.15	0.956
	Pyrene	3.39	1.73	2.54	3.04	7.24	4.37	4.31	4.18	3.38	3.35	7.76	3.53
	Benzo(e) pyrene	2.39	1.42	2.18	2.63	5.40	3.58	3.44	3.23	2.61	2.63	5.51	2.73
	Benzo(ghi) perylene	4.35	2.57	4.60	5.72	11.60	7.89	7.37	7.13	5.44	5.53	10.59	6.11
	2,6-dimethylnaphthalene	3.14	1.39	1.71	1.23	0.888	2.47	1.17	1.48	1.59	1.31	1.31	1.23
	1-methylnaphthalene	1.10	0.489	0.823	0.594	0.428	1.19	0.563	0.713	0.764	0.629	0.629	0.594
	2-methylnaphthalene	1.74	0.772	1.02	0.739	0.532	1.48	0.700	0.887	0.950	0.782	0.782	0.739
	Perylene	3.01	2.11	4.43	3.84	5.00	4.49	3.61	3.68	3.03	2.56	4.15	2.73
	1-methylphenanthrene	1.40	0.622	1.05	0.761	0.548	1.52	0.721	0.913	0.979	0.806	0.806	0.761
	2,3,5-trimethylnaphthalene	1.39	0.617	0.923	0.667	0.480	1.33	0.632	0.800	0.857	0.706	0.706	0.667

Appendix D. Continued.

		January 2016		July 2016									
Class	**Chemical (μg/g$_{OC}$)**	**T1**	**T5**	**T1**	**T5**	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**
PCBs	Total PCBs	3.05	4.68	3.40	5.39	3.98	5.21	4.45	4.38	4.27	3.79	4.50	4.22
Pyrethroids	Bifenthrin	0.134	0.108	0.130	0.123	0.436	0.254	0.216	0.166	0.125	0.187	0.241	0.147
	Cyfluthrin	0.028	0.122	0.037	0.040	0.226	0.067	0.065	0.105	0.046	0.138	0.071	0.073
	Cypermethrin	0.026	0.012	0.015	0.014	0.077	0.043	0.038	0.026	0.010	0.012	0.029	0.020
	Deltamethrin	0.036	0.087	0.052	0.076	0.157	0.442	0.195	0.089	0.144	0.064	0.172	0.124
	Esfenvalerate	0.006	0.006	0.007	0.005	0.016	0.010	0.009	0.009	0.005	0.010	0.049	0.005
	Fenpropathrin	0.044	0.019	0.033	0.024	0.017	0.047	0.022	0.028	0.031	0.025	0.025	0.024
	Lamda-Cyhalothrin	0.094	0.042	0.036	0.026	0.084	0.052	0.025	0.033	0.033	0.027	0.027	0.026
	Permethrin	0.430	0.191	0.366	0.264	0.436	0.529	0.251	0.317	0.340	0.280	0.280	0.264

Appendix E. Whole sediment TIE results.

Sample	Survival (%)		
	Mean	Standard Deviation	N
Home Sediment	100	0.0	6
T1 Baseline	62	9.8	6
T5 Baseline	70	8.9	6
Dilution Control T1 (20% Home Sediment)	80	14.1	6
Dilution Control T5 (20% Home Sediment)	92	11.7	6
Carboxylesterase Blank (1 unit/ml OW every 48 hr)	95	8.4	6
T1 Carboxylesterase (1 unit/ml OW every 48 hr)	53	27.3	6
T5 Carboxylesterase (1 unit/ml OW every 48 hr)	47	32.7	6
Bovine Serum Albumin Blank (0.042 mg/ml OW every 48 hr)	68	45.8	6
T1 BSA (0.042 mg/ml OW every 48 hr)	13	32.7	6
T5 BSA (0.042 mg/ml OW every 48 hr)	27	30.8	6
Piperonyl Butoxide Blank (400 ug/L)	98	4.1	6
T1 Piperonyl Butoxide (400 ug/L)	22	17.2	6
T5 Piperonyl Butoxide (400 ug/L)	68	22.3	6
SIR 300 Blank (20%)	93	8.2	6
T1 SIR 300 (20%)	85	10.5	6
T5 SIR 300 (20%)	78	19.2	5
Zeolite Blank (20%)	98	4.1	6
T1 Zeolite (20%)	50	17.9	6
T5 Zeolite (20%)	77	5.2	6
Charcoal Blank (15%)	50	16.7	6
T1 Charcoal (15%)	58	17.2	6
T5 Charcoal (15%)	63	16.3	6

Appendix F. Pore water TIE results.

	Survival (%)								
	4 Day			7 Day			10 Day		
Sample	Mean	Stdev	N	Mean	Stdev	N	Mean	Stdev	N
Control (32 ppt lab water)	100	0.0	5	92	17.9	5	72	17.9	5
Dana Pt.	92	11.0	5	64	32.9	5	48	33.5	5
T1 Baseline	92	11.0	5	88	17.9	5	72	22.8	5
T5 Baseline	96	8.9	5	88	11.0	5	84	8.9	5
Carboxylesterase Blank	95	10.0	4	95	10.0	4	90	11.5	4
T1 Carboxylesterase	95	10.0	4	95	10.0	4	90	11.5	4
T5 Carboxylesterase	95	10.0	4	90	11.5	4	75	19.1	4
Bovine Serum Albumin Blank	100	0.0	4	100	0.0	4	95	10.0	4
T1 BSA	100	0.0	4	95	10.0	4	90	11.5	4
T5 BSA	100	0.0	4	95	10.0	4	80	16.3	4
Piperonyl Butoxide Blank	70	47.6	4	65	47.3	4	40	49.0	4
T1 Piperonyl Butoxide	85	19.1	4	85	19.1	4	85	19.1	4
T5 Piperonyl Butoxide	75	37.9	4	60	49.0	4	45	52.6	4
EDTA Blank	90	11.5	4	85	10.0	4	75	10.0	4
T1 EDTA	100	0.0	4	90	11.5	4	70	11.5	4
T5 EDTA	90	11.5	4	90	11.5	4	80	23.1	4
Zeolite Blank	95	10.0	4	90	11.5	4	90	11.5	4
T1 Zeolite	95	10.0	4	80	0.0	4	60	16.3	4
T5 Zeolite	90	11.5	4	80	16.3	4	70	25.8	4
C8 Column Blank	90	20.0	4	85	19.1	4	80	28.3	4
T1 C8 Eluate	100	0.0	4	80	16.3	4	70	25.8	4
T5 C8 Eluate	100	0.0	4	100	0.0	4	85	19.1	4
Sodium Thiosulfate Blank	95	10.0	4	80	16.3	4	65	30.0	4
T1 STS	100	0.0	4	90	20.0	4	80	28.3	4
T5 STS	100	0.0	4	90	11.5	4	75	10.0	4

Appendix G-1. Sediment acid volatile sulfides and simultaneously extracted metals for the January 2016 sampling event.

Constituent	T1	T1-TIE	T5	T5-TIE
AVS (µmoles/g)	3.23	4.09	3.11	3.17
Ni (µmoles/g)	0.024	0.024	0.039	0.040
Zn (µmoles/g)	2.287	2.263	4.105	4.531
Cd (µmoles/g)	0.002	0.003	ND	ND
Pb (µmoles/g)	0.067	0.077	0.223	0.265
Cu (µmoles/g)	ND	0.006	0.410	0.697
Ag (µmoles/g)	ND	ND	ND	ND
ΣSEM	2.38	2.37	4.78	5.53
ΣSEM-AVS	-0.85	-1.72	1.67	2.37
TOC (%)	0.76	0.90	1.84	1.87
ΣSEM-AVS/foc (µmoles/g OC)	-111.58	-191.18	90.68	126.58

Appendix G-2. Sediment acid volatile sulfides and simultaneously extracted metals for the July 2016 sampling event.

Constituent	T1	T5	S1	S2	S3	S4	S5	S6	S7	S7-dup	S8
AVS (µmoles/g)	3.75	1.35	0.725	0.428	1.70	0.852	2.37	2.55	1.88	2.46	1.13
Ni (µmoles/g)	0.030	0.041	0.040	0.024	0.037	0.033	0.031	0.032	0.024	0.036	0.034
Zn (µmoles/g)	2.762	4.599	3.584	2.780	4.226	1.063	5.177	5.480	4.627	4.491	4.321
Cd (µmoles/g)	0.003	ND	0.005	ND	ND	ND	ND	ND	ND	ND	ND
Pb (µmoles/g)	0.125	0.345	0.367	0.172	0.351	0.246	0.217	0.207	0.224	0.226	0.333
Cu (µmoles/g)	0.023	1.242	0.672	0.645	0.879	0.757	0.395	0.384	0.213	0.561	1.0572
Ag (µmoles/g)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
ΣSEM	2.94	6.23	4.67	3.62	5.49	5.10	5.82	6.10	5.09	5.31	5.75
ΣSEM-AVS	-0.81	4.87	3.94	3.19	3.80	4.25	3.45	3.56	3.21	2.85	4.61
TOC (%)	1.3	1.8	2.5	0.9	1.9	1.5	1.4	1.7	1.7	1.7	1.8
ΣSEM-AVS/foc (µmoles/g OC)	-62.01	270.69	157.75	354.77	199.75	283.19	246.52	209.13	189.03	167.60	256.17

Appendix H. Benthic organisms identified and enumerated from the July 2016 MdRH sediment quality survey.

	Station/Abundance								
Species	**S1**	**S2-**	**S3**	**S4**	**S5**	**S6**	**S7**	**S7 Dup**	**S8**
Scoletoma sp	29	4	5	2	3	2			1
Leitoscoloplos pugettensis	21	7	13	19	19	8	18	3	18
Scoletoma sp A	9								
Scoletoma sp C	7	5	4	15	7	6	7	5	6
Malacoplax californiensis	7	3							
Scoletoma sp B	7								
Hartmanodes hartmanae	4								
Mediomastus sp	3	4		2	3				1
Tagelus affinis	2	30	3	10	1	5	6	7	17
Chaetozone corona	2								
Euchone limnicola	1	10	1	14		3	8	2	20
Heteronemertea	1								
Prionospio heterobranchia	1								
Cossura sp A	1								
Monticellina sp 1		3		1	3	1		2	8
Mayerella acanthopoda		2	1	1			1		
Lineidae sp LAH1		2							
Paraprionospio alata		2							
Pseudopolydora paucibranchiata		1	147	47			4	11	14
Acteocina carinata		1	10	36	6	4	7		
Laevicardium substriatum		1	2					1	
Alpheus californiensis		1	1						
Streblospio benedicti		1		1	1		2		
Neotrypaea gigas		1							
Polynoidae		1							
Lineidae		1						1	
Nephtys caecoides			2						1
Macoma nasuta			1						1
Deltamysis holmquistae			1						
Monticellina sp				3		1		1	5
Scoletoma erecta				1			1	1	
Corymorphidae sp SD1				1				2	
Haminoea vesicula				1					
Theora lubrica					11	12	18	6	3
Phoronida					1		1		1
Euchone incolor					1				
Musculista senhousia						2		2	

Appendix H. Continued.

	Station/Abundance								
Species	**S1**	**S2-**	**S3**	**S4**	**S5**	**S6**	**S7**	**S7 Dup**	**S8**
Phoronis sp							10	9	1
Exogone sp A									1
Glycera americana								1	
Spiophanes duplex								1	