



Analysis of Causality Between  
Aquatic Life Beneficial Use  
Impairment and Site  
Primary COCs at the San  
Diego Shipyard Sediment  
Site

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# 1 Introduction

This report presents an analysis of the relative contributions of five chemicals of concern (COCs; polychlorinated biphenyls (PCBs), high molecular weight polycyclic aromatic hydrocarbons (HPAHs), tributyltin (TBT), copper, and mercury) to Beneficial Use Impairment (BUI) of Aquatic Life identified in *Draft Technical Report For Tentative Cleanup And Abatement Order No. R9-2011-0001 for the Shipyard Sediment Site*, published by the California Regional Water Quality Control Board (CRWQCB, 2010) at the San Diego Shipyard Sediment Site (Site). The five COCs are identified as the primary COCs at the Site by the CRWQCB (2010). The Site is located along the eastern shore of central San Diego Bay at the National Steel and Shipbuilding Company Shipyard facility (NASSCO) and the BAE Systems San Diego Ship Repair Facility (BAE) also known as the Southwest Marine (SWM) Shipyard.

The analysis presented in this report uses well-established scientific techniques and available Site data (Exponent, 2003; CRWQCB, 2010) to evaluate causality between individual chemicals and Aquatic Life BUI. The analysis herein does not provide direct evidence for Aquatic BUI; however, it can be used as a chemical line of evidence (e.g., a chemical line of evidence within a Sediment Quality Triad Evaluation) to explain causality between Aquatic BUI and chemicals. It should be noted that the author does not necessarily agree with the CRWQCB (2010) identification of Aquatic Life BUI at the Site, analyses supporting the CRWQCB (2010) Aquatic BUI conclusions, or the confinement of the analysis presented in this report to consideration of only the five primary Site COCs. The CRWQCB (2010) analysis of Aquatic Life BUI and identification of the five COCs is not a purely technical exercise and is subject to non-technical policy considerations such as those outlined in State Water Board Resolution No. 92-49 (*Policies and Procedures for Investigation and Cleanup and Abatement of Discharges Under Water Code Section 13304*; SWQCB, 1996). My analysis is confined to the assumptions of Site Aquatic Life BUI and evaluation of potential links between Aquatic Life BUI and to the five primary COCs, as put forth by CRWQCB (2010).

My expertise is in the area of environmental toxicology and chemistry, with a particular focus on ecotoxicology, environmental chemistry, and bioaccumulation of chemicals by invertebrates, fish, and wildlife. I have a Doctoral (PhD) degree in Environmental Science. The majority of my 7 years of graduate-level thesis and doctoral research, as supported in part by USEPA-sponsored research fellowships, focused on the bioavailability and effects of chemicals to soil and sediment-dwelling invertebrates. I have been employed with ENVIRON International Corporation (ENVIRON), an environmental consultancy, since July 2004. The main focus of my experience with ENVIRON has been on ecological and human health risk assessment of contaminated sites. I am listed as a primary or co-author on over 20 peer-reviewed publications in environmental toxicology, environmental chemistry, and risk assessment. My career vita is included as Appendix A of this report.

The remainder of this report is organized in the following sections:

- Section 2 provides an overview of the Aquatic Life BUI identification analysis performed by CRWQCB (2010), focusing on the lack of scientific evidence regarding the establishment of causality between primary COCs and Aquatic Life BUI.

- Section 3 provides an analysis to establish causality between the five primary Site COCs and Aquatic Life BUI using available Site data.
- Section 4 presents the conclusions of the analysis.

## 2 Overview of the CRWQCB (2010) Aquatic Life Beneficial Use Impairment Analysis

CRWQCB evaluated impairment of Aquatic Life Beneficial Uses for Estuarine Habitat, Marine Habitat, and Migration of Aquatic Organisms by evaluating exposure and adverse effects to the benthic macroinvertebrate community and fish (Findings 14-15 in CRWQCB, 2010) using data from the 2001-2002 Site investigation by Exponent (2003). Adverse effects to fish from Site chemicals were not identified (Appendix for Finding 15 of CRWQCB, 2010). Adverse effects to the benthic macroinvertebrate community were evaluated by CRWQCB (2010) at each of the 66 sediment stations using one of two approaches, depending on the data collected at each of 66 sampling stations at the Site:

- 1. Triad Approach:** The Triad Approach was based on a CRWQCB-derived Sediment Quality Triad approach (Findings 16 and 18 in CRWQCB, 2010) that integrated three lines of evidence: 1) concentrations of chemicals in Site surface sediment; 2) effects observed in laboratory toxicity tests conducted with Site surface sediment; and 3) enumeration of benthic macroinvertebrates collected from Site surface sediment. This approach was used to evaluate the likelihood of sediment chemical-derived effects on the benthic macroinvertebrate community at the 30 stations where data was collected for each of the three Triad lines of evidence. Six of the 30 Triad stations were classified as “Likely” for chemically-associated impairment (Table 1, Figure 1). A Triad Approach conclusion of “Likely” was equated with impairment of the benthic macroinvertebrate community at a level CRWQCB (2010) assumed to represent Aquatic Life BUI.
- 2. Non-Triad Data Approach:** The Non-Triad Data Approach was based on a CRWQCB-derived empirical approach (Finding 32 in CRWQCB, 2010) that used average quotients calculated from dividing concentrations of PCBs (sum of 40 congeners), HPAHs, copper, mercury, and TBT by empirically-derived median values, as well as comparison of single values to 60% of the Lowest Adverse Effect Thresholds (LAETs) in Site surface sediment to predict the likelihood of sediment chemical-derived effects on the benthic macroinvertebrate community at the 36 stations for which only surface sediment chemistry was available. It should be noted that this analysis was used as a substitute for a full Sediment Quality Triad evaluation because sediment toxicity and benthic macroinvertebrate community census data were not collected at the Non-Triad stations. Seven of the 36 Non-Triad stations were classified as “Likely” for chemically-associated impairment (Table 1, Figure 1). A Non-Triad Data Approach conclusion of “Likely” for a station was equated with impairment of the benthic macroinvertebrate community at a level CRWQCB (2010) assumed to represent Aquatic Life BUI.

The Triad and Non-Triad Data Approaches are inadequate and incomplete for understanding the relative contribution of the five COCs to adverse toxic effects on benthic macroinvertebrate communities that result in an Aquatic Life BUI. As described in Sections 2.1 and 2.2 below, the primary shortcoming for both approaches is that they fail to establish causality between chemicals in Site surface sediment and Aquatic Life BUI.

## 2.1 Triad Approach

The sediment chemistry line of evidence used by the CRWQCB (2010) in the Triad Approach relies on chemical concentrations in Site surface sediment to indicate risk potential associated with chemicals in sediment. The CRWQCB (2010) approach uses Site surface sediment chemistry data and empirical Sediment Quality Guidelines (SQGs). SQGs represent concentrations of a chemical in sediment associated with a threshold for an adverse effect such that when an SQG is exceeded, unacceptable or questionable sediment quality is assumed (Wenning et al., 2005). The SQG approach used in the Triad Approach is represented by the SQGQ1 metric, as shown in Figure 18-1 of CRWQCB (2010). The SQGQ1 value for a sediment is estimated by dividing concentrations of cadmium, copper, lead, silver, zinc, total chlordane, dieldrin, total polycyclic aromatic hydrocarbons (PAHs; normalized by sediment organic carbon content), and total PCBs (sum of 18 congeners) in sediment by each chemical's empirical SQG (Fairey et al., 2001).

A primary flaw in this approach is that TBT is not considered by the SQGQ1 metric, despite the fact that TBT was selected by CRWQCB (2010) as a primary Site COC. CRWQCB (2010) does not explain the rationale for excluding TBT from Aquatic Life BUI evaluation at the Triad Stations. TBT has been found as high as 3,250 µg/kg in Site surface sediments (average Spatially Weighted Average Concentration of 162 µg/kg). It is possible for sediments with potentially toxic levels of TBT to be ranked as “Low” (no indication of toxic chemical presence) using the CRWQCB (2010) Triad Approach. TBT has been referred to as “the most toxic compound ever released into the environment” (Meador, 2010). TBT is toxic to aquatic invertebrate life, with effects noted in water at concentrations of 0.07 to 0.007 micrograms per liter (µg/L) and in sediment at concentrations less than 100 micrograms per kilogram (µg/kg) (Meador et al., 2002; Meador, 2011). Because of its high toxicity to aquatic life, TBT was routinely added as a pesticide to marine paints to prohibit the accumulation of invertebrates on marine vessels (Meador et al., 2002; USEPA, 2003a; Meador, 2011). Shipyard uses of TBT at the Site are well-documented in CRWQCB (2010) and the San Diego Shipyard Site administrative record.

The second critical flaw in the CRWQCB (2010) approach concerns the nature of the SQGs used in the SQGQ1 metric. The SQGs used in the SQGQ1 approach are referred to as “empirical” SQGs because they are derived from studies that have measured concentrations of chemicals and laboratory toxicity in field-collected sediments containing a variety of chemicals and exhibiting a variety of physical properties. As these sediments contain a wide variety of unmeasured and measured physical and chemical properties that may adversely affect the laboratory toxicity test organisms, it is impossible from that approach alone to know which chemical, group of chemicals, or physical condition may be responsible for the presence of adverse effects (Batley et al., 2005). Despite the lack of causality between concentrations of individual chemicals and adverse effects, SQGs for each measured chemical are derived nonetheless. Empirical SQGs are often criticized as overly-conservative and only useful in an initial screening evaluation to identify sediments that are not toxic (Batley et al., 2005). CRWQCB (2010) noted that empirical SQGs have a limited ability relative to other SQG approaches (such as causal SQGs based on methods that evaluate links between effects and

chemicals on an individual chemical basis) to determine the specific contaminants responsible for toxicity.

Furthermore, chemical concentrations in sediment that exceed their corresponding empirical SQGs do not necessarily indicate that toxicity to aquatic life is certain, probable, or likely. Empirical SQGs should not be used to attribute causality between any one particular chemical and an observed adverse effect (Batley et al., 2005; Becker and Ginn, 2007). For example, Fuchsman et al. (2006) and Becker and Ginn (2007) both noted that adverse effects in benthic invertebrates occur at concentrations of PCBs in sediment that are at least an order of magnitude higher than the “consensus-based” empirical PCB SQG (400 µg/kg) used in the Fairey et al. (2001) SQGQ1 value, as developed by MacDonald et al. (2000). Thus, it is clear that the sediment chemistry evaluation in the Triad Approach used by CRWQCB (2010) is not useful in identifying COCs for Aquatic Life BUI because the empirical SQGs used in the SQGQ1 approach do not include TBT, a primary COC at the Site, and are not useful in establishing risk potential or relating causality of individual chemicals in sediment to adverse effects.

## 2.2 Non-Triad Data Approach

The Non-Triad Data approach used by the CRWQCB (2010) relies solely on an empirical evaluation of the concentration of chemicals in Site surface sediment, because laboratory toxicity tests and benthic macroinvertebrate censuses were not performed at 36 of the 66 Site sampling stations. The Non-Triad Data Approach consists of an evaluation of concentrations of the five primary Site COCs in surface sediments using two benchmark approaches (Finding 32 in CRWQCB, 2010): 1) comparison of station chemistry data to concentrations corresponding to 60% of the LAETs for PCBs (total of 41 congeners), HPAHs, copper, mercury, and TBT; and 2) use of station chemistry data to calculate a Site-specific Median Effects Quotient (SS-MEQ) value that is compared to a Site-derived SS-MEQ threshold. Both approaches are critically flawed and cannot be used to quantify or understand the relative causal contribution of the five COCs to adverse toxic effects on macroinvertebrate communities.

The first approach of the Non-Triad Data Approach is a simple comparison of concentration of the 5 COCs in Site surface sediment to 60% of COC-specific LAETs. LAETs are based on an SQG-derivation approach referred to as Apparent Effects Thresholds (AETs). As described on page 32-31 of CRWQCB (2010):

*“The [AET] is a tool for identifying concentrations of a pollutant in sediment above which adverse biological effects are always expected. When multiple site-specific effects endpoints are measured, several AET values can be combined to derive a single set of AET values by conservatively applying the lowest of any of the individual AET values for each chemical. This is known as the lowest AET or LAET.”*

For each of the five COCs, LAET values were identified as the highest concentration found in the 24 Triad stations for which “Likely” Triad results (i.e., no Aquatic Life BUI) were not observed (Table A32-9 in CRWQCB, 2010). CRWQCB (2010) lowered the LAET values by multiplying the values by 60% to “provide an additional margin of protection”; the resultant values are referred to as 60% LAETs. If any of the five 60% LAET values were exceeded at a Non-Triad Data station, it was considered to indicate a result similar to a Triad “Likely” classification, and



thus, was assumed to indicate Aquatic Life BUI. It should be noted that CRWQCB (2010) did not provide scientific basis or similar justification for the selection of the 60% adjustment factor. The use of the 60% value is arbitrary and is not supported by any technical or regulatory guidance.

The SS-MEQ metric used by CRWQCB (2010) is based on an approach similar to that used to derive AET values. As described in CRWQCB (2010), SS-MEQ values for each of the 38 Non-Triad stations were derived by dividing concentrations of each of the five primary Site COCs in sediment by Site-specific “SS-Median” values. SS-Median values were derived by calculating the median concentrations of the five COCs at the six stations identified with a “Likely” Triad classification. If the SS-MEQ value at a Non-Triad Data station exceeded a value of 0.9, it was considered to indicate a result similar to a Triad “Likely” classification, and thus, was assumed to indicate Aquatic Life BUI. The SS-MEQ threshold of 0.9 was obtained through an optimization process based on Site data. Neither the SS-MEQ approach, nor its threshold of 0.9 is supported by any technical or regulatory guidance.

AETs (the basis of the 60% LAET metric) and SS-Medians (the basis of the SS-MEQ metric) do not provide information about the causality of chemicals that result in Aquatic Life BUI. AETs and SS-Medians are empirically-derived SQGs that lack of an establishment of causality between chemicals and adverse effects. As with other empirical SQGs, LAETs and SS-Medians have been developed in multi-chemical mixtures. Thus, it is impossible to use LAETs and SS-Medians to identify which chemical, or chemicals, could be eliciting the observed effects. Furthermore, the LAET approach is sensitive to outliers, a single one of which can drastically increase the LAET value (Batley et al., 2005). For example, a LAET can be based on a single sediment replicate exhibiting an extremely high concentration and lack of adverse effect. There is no consideration for the possibility that the single result used as the basis for the LAET is a potential outlier that exhibits extremely high concentrations relative to the other samples within the dataset.

Additionally, the CRWQCB (2010) LAET and SS-Median approaches, as with many empirical SQGs based on a concentration of chemical in bulk sediment, do not explicitly consider bioavailability. The concentration of chemicals in bulk sediment alone is usually insufficient to determine whether adverse toxicological effects may or may not occur because partitioning and binding phases can affect the bioavailability of chemicals in the sediment. Physical and chemical conditions such as total organic carbon (TOC), pH, acid volatile sulfides (AVS), and simultaneously extracted metals (SEM) are hypothesized to provide better insight on benthic effects than measured bulk sediment chemical concentrations (Di Toro et al., 2005a; Di Toro et al., 2005b; USEPA, 2003b; USEPA, 2005a; Simpson and Batley, 2007). These conditions affect the binding/presence of chemicals in organic matter or other solid phases that prevent the interaction between chemicals and biota. It is possible that the particular sediment replicate from which an LAET was developed exhibits extreme conditions (relative to the rest of the Site) or contains a form of the chemical that results in much lower bioavailability, thus, resulting in a bias that inflates the LAET.

LAETs and SS-Median values can also represent underestimates of toxic thresholds (resulting in an overestimation of toxicity). If a COC is present in the dataset only at non-toxic

concentrations, the approaches will nevertheless generate an LAET and SS-Median value for any of the selected COCs, despite the lack of established causality between the observed concentrations of the COC and adverse effects. The LAET and SS-Median values will represent values of questionable significance that are bounded within the range of concentrations in the dataset. Conceptually, one might expect that in such a case, the LAET would be equal to the highest concentration of the non-toxic COC. However, if the higher concentrations of the non-toxic COC co-vary with the higher concentrations of COC that is present at toxic concentrations, then the non-toxic COC will be attributed causality for the adverse effect to the same degree as the COC actually eliciting the adverse effects. In this scenario, the LAET for the non-toxic COC may actually be substantially lower than its highest measured concentration at the Site. It is impossible to determine whether any given LAET or SS-Median value is similar to, higher than, or lower than the actual toxicity threshold for any given chemical. Thus, it is clear that the Non Triad Data Approach is critically flawed such that it provides no useful information in addressing Aquatic Life BUI at the Site, and may be falsely attributing Aquatic BUIs to chemicals that are present at non-toxic concentrations at the Site.

### 3 Analysis of the Aquatic Life BUI Potential for the Five Primary Site COCs

The analysis presented in this section evaluates the risk potential to benthic invertebrates at each of the 13 Site stations identified by CRWQCB (2010) as exhibiting Aquatic Life BUI (Table 1, Figure 1). The analysis does not provide direct evidence for Aquatic BUI; however, it can be used as a chemical line of evidence (e.g., a chemical line of evidence within a Sediment Quality Triad Evaluation) to explain causality between Aquatic BUI and chemicals. This analysis is based on well-established scientific approaches for understanding chemical-specific risk to aquatic life that have been derived from USEPA guidances and peer-reviewed scientific literature to demonstrate causality between the presence of COCs and Aquatic Life BUIs at the Site. These approaches have been used at a number of contaminated sediment sites for identifying chemical-specific causes of observed sediment toxicity and evaluating sediment quality (Meador et al., 2002; Douglas et al., 2005; Fuchsman et al., 2006; Sorensen et al., 2007; Ingersoll et al., 2009; Ololade, 2010). The approach relies on equilibrium partitioning (EqP) and tissue-based analyses referred to as “causal SGQ” approaches by CRWQCB (2010). As stated in CRWQCB (2010): “*Causal SQGs have a greater ability relative to empirical SQGs to determine the specific contaminants responsible for toxicity.*”

The following subsections provide an overview of each step of the analysis:

- Section 3.1 describes the methods and sources of information used to derive station-specific exposure estimates;
- Section 3.2 describes the sources of information used to derive chemical-specific effects benchmarks;
- Section 3.3 describes the method by which station-specific exposure estimates are compared to chemical-specific effects benchmarks to evaluate Aquatic Life BUI risk potential; and
- Section 3.4 presents the results of the analysis.

#### 3.1 Exposure Estimation Methods

Chemical exposure estimates are based on measured and/or estimated concentrations of chemicals in surface sediment, porewater and benthic invertebrate tissue (provided in Exponent, 2003) at each of the 13 stations identified by CRWQCB (2010) as exhibiting Aquatic Life BUI (Table 1, Figure 1). Exposure estimates are derived for each COC based on methods that explicitly consider bioavailability, are based on peer-reviewed and/or USEPA-approved guidance, and could be used with the available Site data reported in Exponent (2003):

- **Copper** exposure is estimated using (1) an EqP model developed by USEPA (2005a) that predicts copper in sediment porewater by accounting for Site measurements of AVS in surface sediment, (2) an EqP model that predicts copper concentration in sediment porewater; (3) available Site-specific porewater measurements; and (4) an empirical model derived by Exponent (2003) that predicts the concentration of copper in porewater from measured concentrations of copper in surface sediment (Section 3.1.2).

- **Mercury** exposure is estimated using (1) a Site-specific approach to predict the concentration of mercury in invertebrate tissue using a bioaccumulation factor (BAF) developed in CRWQCB (2010) or (2) available Site-specific tissue residue measurements (Section 3.1.3);
- **HPAH** exposure is estimated using: (1) an EqP model developed by USEPA (2003b) that predicts HPAHs in sediment porewater; and (2) available Site-specific porewater measurements (Section 3.1.4);
- **PCB** exposure is estimated using: (1) an EqP model developed from USEPA (2003b) by Fuchsman et al. (2006) that predicts PCBs in sediment porewater; (2) available Site-specific porewater measurements; and (3) an empirical model derived by Exponent (2003) that predicts the concentration of PCBs in porewater from measured concentrations of PCBs in surface sediment (Section 3.1.5); and
- **TBT** exposure is estimated using: (1) an EqP model developed from USEPA (2003b) by Meador et al. (2002) that predicts TBT in sediment porewater; (2) available Site-specific porewater measurements; and (3) an empirical model derived by Exponent (2003) that predicts the concentration of TBT in porewater from measured concentrations of TBT in surface sediment (Section 3.1.6).

Four of the five exposure estimation methods used in this analysis are based on methods that quantify exposure in sediment porewater. As noted above, up to four sources of information regarding the concentrations of COCs in porewater were considered for comparison to effects benchmarks. The concentrations of COCs derived by these methods are conservative, and likely higher than the true concentrations of COCs that are bioavailable to invertebrates. For example, measurements of COCs in Site porewater were noted by Exponent (2003) to likely represent overestimates because a portion of the COC mass found in sediment porewater may have been bound to fine suspended or colloidal material that could not be removed by centrifugation techniques used in the analysis. COCs bound to this material would not be bioavailable (Hawthorne et al., 2007), and a value that included this COC mass would overestimate bioavailable chemicals in porewater. The empirical models (Exponent, 2003) used to predict concentrations of COCs in porewater at stations where porewater measurements were not available are also conservative since they are based on measured porewater data from other stations. Additionally, EqP-predicted concentrations in porewater may also be biased high, because the EqP model has been shown to overestimate actual concentrations of chemicals in porewater under circumstances in which binding to sediment solid phases is overestimated by model parameters (Hawthorne et al., 2007; Arp et al., 2009; Gschwend et al., in press).

Ideally, all five COCs should be evaluated using identical or similar approaches such that possible biases inherent to each method affect all five evaluations equally. However, a porewater approach was not possible for the exposure estimation approach for mercury. In the aquatic environment, mercury can be found in two geochemical forms: inorganic mercury and methylmercury. The presence of each form in sediment porewater is controlled by a variety of physical and chemical properties such that concentrations cannot be reliably predicted using

concentrations of inorganic mercury and methylmercury in bulk sediment with EqP or similar models. There are also large differences between inorganic mercury and methylmercury uptake rates and toxicity in aquatic biota (Beckvar et al., 2000; Beckvar et al., 2005). Additionally, only total mercury (which comprises both inorganic mercury and methylmercury) was measured in Site sediment, sediment porewater, and biota (Exponent, 2003). Because a mercury porewater exposure estimate was not possible, mercury exposure was quantified according to concentrations in invertebrate tissue predicted from concentrations of mercury in surface sediment.

Estimating exposure on the basis of concentrations of a chemical in invertebrate tissue is a valid approach for explicitly considering bioavailability and risk potential of many bioaccumulative chemicals (McCarty et al., 2010; Meador et al., 2011). However, it is not possible to compare the concentrations of all metals (other than mercury, for example) in invertebrates to effects benchmarks in tissue. Estimation of effects benchmarks based on concentrations of total metals in invertebrate tissue may not be possible for some metals, or may only be possible when exposures are near lethal thresholds because many species are able to actively regulate and/or detoxify metals during sublethal exposures (Conder and Lanno, 2002; Conder et al., 2002; Rainbow and Luoma, 2011). Copper is an example for which an effects benchmark is difficult or impossible to derive because the internal concentration of copper in invertebrates is physiologically controlled (Rainbow and Luoma, 2011). Adverse effects due to copper may be observed without an increase in concentration within tissue, and vice versa. Thus, observed concentrations in invertebrates often correlate poorly with adverse effects.

Another disadvantage of the tissue residue approach is a lack of effects benchmarks to which Site-specific concentrations of chemicals in tissue can be compared. Although the approach is gaining acceptance and is considered an extremely useful line of evidence in understanding risk (Meador et al., 2011), widely-accepted and agency-promulgated tissue residue effects benchmarks are not available to the same extent as effects benchmarks based in solution (surface water, sediment porewater), such as Ambient Water Quality Criteria and their associated Final Chronic Value (FCV) effects benchmarks. However, because concentrations of PCBs, HPAHs, and TBT in mussels were available at some of the Site stations exhibiting Aquatic Life BUI (laboratory bioaccumulation experiments with *Macoma nasuta* and measurement of COCs in mussels collected from the Site (Exponent, 2003)), a tissue-based approach was used as an additional line of evidence to evaluate the risk potential of PCBs, HPAHs, and TBT.

### 3.1.1 Sources of Site Data for Exposure Estimation

Site-specific data used to estimate exposure was obtained from a Site investigation conducted in 2001-2002 by Exponent (2003). This investigation provided surface sediment, porewater, and mussel soft tissue chemistry data for this report. Only chemistry data in surface sediment samples collected from a depth interval of 0-2 centimeters was used for exposure estimation, as in CRWQCB (2010). For up to four of the 13 stations exhibiting Aquatic Life BUIs, multiple values for concentrations of COCs and other parameters in surface sediment and sediment porewater are available due to the collection of multiple samples over time, duplicates, and/or split samples. Multiple values were simplified to a single value at each station using the same data algorithm used by CRWQCB (2010) to generate single values for the concentrations of

COCs in surface sediment in the estimation of Site Surface Area Weighted Concentrations (and subsequent analyses to estimate pre- and post-remedy risk). Concentrations of chemicals and other relevant parameters in surface sediment from the 13 stations exhibiting Aquatic Life BUIs are provided in Tables 2 through 6. Concentrations of chemicals in porewater are provided in Table 7 through 9.

### 3.1.2 Copper Exposure Estimation Method

Exposure to copper was estimated using two approaches that focused on bioavailable copper in sediment porewater. The first approach assumes that the main partitioning phases influencing the presence of bioavailable copper in sediment are AVS and organic carbon (USEPA, 2005a). AVS and organic carbon bind copper and other cationic metals, limiting metal exposure to benthic invertebrates (USEPA, 2005a). In the model, the measured concentration of AVS is subtracted from the measured concentration of copper simultaneously extracted in the same surface sediment (Simultaneously Extracted Metal, SEM) and normalized by the measured TOC in surface sediment to quantify the amount of bioavailable copper present in surface sediment. If this value (the TOC-normalized excess SEM concentration of copper in surface sediment) is negative, the amount of bioavailable copper is assumed to be zero, because all copper mass is assumed to be rendered unavailable via binding to AVS.

The second approach for estimating copper exposure was based on predicted and measured concentrations of copper that were not corrected for the presence or absence of AVS or copper. To account for binding to solid phases that result in a reduction in copper bioavailability, an EqP model (USEPA, 2005a) parameterized with a literature-derived sediment-porewater partition coefficient ( $K_d$ ) (USEPA, 2005b) was used to estimate the concentration of copper in porewater using the concentration of copper in Site surface sediment. Concentrations of copper in porewater were also based on an empirical Site-specific model to predict the concentrations of copper in porewater using the concentrations of copper in surface sediment (Table 5-2 of Exponent, 2003), or measurements by Exponent (2003) (Table 7).

As noted above, predicted and measured concentrations of chemicals in porewater may overestimate the true concentration of bioavailable chemicals in porewater. Thus, if the measured concentration of copper in porewater is reported, the lesser value of the measured concentration in porewater or the estimated concentration in porewater [EqP model approach] is used for comparison to the effects benchmark. If there is no available measured concentration at a Site station, the lesser value of that predicted by the EqP or Exponent (2003) models is used for comparison to the effects benchmark. It should be noted that the algorithm for selection of the lowest estimates of exposure was also followed for the other COCs.

### 3.1.3 Mercury Exposure Estimation Method

Mercury exposure was evaluated on a tissue residue basis due to the complexities of bioavailability and geochemistry associated with this contaminant. At each of the 13 Site stations exhibiting Aquatic Life BUI, a mercury exposure estimate was derived from the concentration of mercury in Site benthic mussel (*Musculista senhousi*) soft tissue (station NA19) or estimated by multiplying the concentration of mercury in surface sediment by a Site-specific BAF. The BAF-approach for estimating the concentration of mercury in invertebrate

tissue is based on the same approach and data used by CRWQCB (2010). Specifically, the BAF was calculated as the measured total mercury concentration in Site mussel (Table E-1. In Exponent, 2003) divided by the CRWQCB (2010) pre-remedy Spatially Weighted Average Concentration of mercury in Site sediment (Table 10). As in CRWQCB (2010), the BAF value for each station with a measured concentration of mercury in Site mussels was averaged to obtain the final BAF value of 0.026 kg sediment/kg tissue, wet weight (ww).

#### 3.1.4 HPAH Exposure Estimation Method

HPAH exposure was estimated using concentrations of the ten individual HPAHs (as designated in CRWQCB, 2010) in sediment porewater. Exposure estimates were obtained from EqP-model predictions (USEPA, 2003b) or measurements by Exponent (2003). The EqP model approach (Di Toro et al., 1992) calculates the concentration of chemical in porewater as the concentration in surface sediment (organic carbon normalized) divided by organic carbon-water partition coefficients ( $K_{OCs}$ ). EqP predictions are calculated for each individual HPAH (HPAH<sub>i</sub>) (USEPA, 2003b). Measured concentrations of HPAHs in sediment porewater were available for four stations, SW01, SW04, SW24, and SW28 (Table 8).

As noted above, predicted and measured concentrations of chemicals porewater in Site sediment may both overestimate the true concentration of bioavailable chemicals in porewater. Thus, for stations where measured values were available, the lesser of the measured or predicted values was selected for each HPAH and used for comparison to the effects benchmark (selection of the lowest estimates was also followed for the other COCs).

An empirical Site-specific approach to predict the concentrations of HPAHs in porewater using the concentrations of HPAHs in surface sediment was not used. Exponent (2003) did not present regression models relating measured concentrations in porewater to concentrations in sediment for stations at which both measurements were available. Development of statistically significant and meaningful regression models was not possible for all individual HPAHs due to the small dataset, presence of outliers, and high proportion of porewater results that were below detection limits.

#### 3.1.5 PCB Exposure Estimation Method

PCB exposure was estimated using concentrations of the ten PCB homologues in sediment porewater. Exposure estimates were obtained from EqP-model predictions (Fuchsman et al., 2006), an empirical Site-specific model to predict the concentrations of total PCB homologues in porewater using the concentrations of PCBs in surface sediment (Table 5-2 of Exponent, 2003), or measurements by Exponent (2003) (Table 9).

The EqP model approach (Di Toro et al., 1992) calculates the concentration of chemical in porewater as the concentration in surface sediment (organic carbon normalized) divided by  $K_{OC}$ . For PCBs, the  $K_{OC}$  for nonionic organic chemicals is assumed to be equal to the  $K_{OW}$  (Bucheli and Gustafsson, 2001) as assumed by Fuchsman et al. (2006). Because nine of the ten PCB homologue groups are each comprised of more than one PCB congener, each PCB congener has a different  $K_{OW}$ , and the congener-specific composition of the PCB homologues likely varies among the Stations, an average  $K_{OW}$  value for the sample-specific composition of total PCBs at each of the 13 stations exhibiting Aquatic Life BUI was calculated (Table 11). This average total

PCB  $K_{OW}$  value served as the  $K_{OC}$  value used to estimate the concentration of total PCB homologues in sediment porewater via the EqP model (DiToro et al., 1992; Fuchsman et al., 2006).

As noted above, predicted and measured concentrations of chemicals in Site porewater may overestimate the true concentration of bioavailable chemicals in porewater. Thus, if the measured concentration of PCBs in porewater is reported, the lesser value of the measured concentration in porewater or the estimated concentration in porewater [EqP model approach] is used for comparison to the effects benchmark. If there is no available measured concentration at a Site station, the lesser value of that predicted by the EqP or Exponent (2003) models is used for comparison to the effects benchmark. It should be noted that the algorithm for selection of the lowest estimates of exposure was also followed for the other COCs.

### 3.1.6 TBT Exposure Estimation Method

TBT exposure was estimated using concentrations of TBT in sediment porewater. Exposure estimates were obtained from EqP-model predictions (Meador et al., 2002; USEPA, 2003b), an empirical Site-specific model to predict the concentrations of TBT in porewater using the concentrations of TBT in surface sediment (Table 5-2 of Exponent, 2003), or measurements by Exponent (2003) (Table 7).

The EqP model approach (Di Toro et al., 1992) calculates the concentration of chemical in porewater as the concentration in surface sediment (organic carbon normalized) divided by  $K_{OC}$ . The  $K_{OC}$  value for TBT was calculated using the following equation from Di Toro (1985)  $\log_{10}K_{OC} = 0.00028 + 0.983 \log_{10}K_{OW}$  (USEPA, 2003b). The  $\log_{10}K_{OW}$  value for TBT (4.4) was obtained from Meador (2011).

As noted above, predicted and measured concentrations of chemicals porewater in Site sediment may overestimate the true concentration of bioavailable chemicals in porewater. Thus, if the measured concentration of TBT in porewater is reported, the lesser value of the measured concentration in porewater or the estimated concentration in porewater [EqP model approach] is used for comparison to the effects benchmark. If there is no available measured concentration at a Site station, the lesser value of that predicted by the EqP or Exponent (2003) models is used for comparison to the effects benchmark. It should be noted that the algorithm for selection of the lowest estimates of exposure was also followed for the other COCs.

## 3.2 Derivation of Effects Benchmarks

Effects benchmarks for the five primary Site COCs were based on published field and laboratory studies using multiple species and multiple sensitive endpoints relevant to population-level effects, such as growth, development, and reproduction. Effects benchmarks for PCBs, HPAHs, copper, and TBT were obtained directly from peer-reviewed literature and/or USEPA-promulgated values.

The effect benchmark for mercury was derived from freshwater and saltwater studies selected from the United States Army Corps of Engineers (USACOE, 2010) Environmental Residue-Effects Database (ERED). Values were selected with a preference for controlled, single-COC exposure studies exhibiting a dose-response relationship characterized by well-defined ranges



between No Observed Effect Doses (NOEDs) and Lowest Observed Effect Doses (LOEDs). Toxic effects considered included development, reproduction, growth, and lethal effects that, if exhibited, would likely result in population-level effects resulting in degradation of benthic invertebrate communities. Due to the paucity of concentrations of mercury associated with adverse effects in invertebrates, tissue residues associated with an absence of adverse effects (NOEDs) were included in the analysis. The use of NOED values represents a considerable source of uncertainty and conservatism for establishing effect benchmarks in tissue, as NOEDs are not usually used in deriving tissue effects benchmarks (Di Toro et al., 2000; Conder and Lanno, 2002; Meador et al., 2002; McCarty et al., 2010).

The mercury effects tissue benchmark review yielded values ranging from 0.19 mg/kg, ww to 40 mg/kg, ww (Table 12). The tissue residue concentration of 0.19 mg/kg, ww was selected as the effects benchmark for this analysis, corresponding to a value that is protective of at least 90% of aquatic species (Figure 2). This value was derived from a study by Beckvar et al. (2000), which observed impairment of growth in the mussel *Elliptio complanata* exposed along several points in the Sudbury River. This value has considerable uncertainty since the organisms used in the study were not from a controlled environment and may have been exposed to other contaminants or stressors prior to or during the Beckvar et al. (2000) field study. Additionally, the adverse effects were observed in organisms exhibiting lower concentrations of mercury in tissue, suggesting that mercury may not have been responsible for the adverse effects. Mortality was not observed in individuals of this species exhibiting tissue concentrations of 3.0 mg/kg, ww (Tessier et al., 1996). Vertebrate data were not included in this review (Table 12) due to uncertainties regarding the mercury and methylmercury potency between invertebrate and vertebrates (Beckvar, et al., 2000; Beckvar et al., 2005). However, the selected mercury tissue effects benchmark for invertebrates (0.19 mg/kg, ww) is similar to a whole-body tissue residue effects benchmark for growth, reproduction, development, and behavior effects in fish (Beckvar et al., 2005). As such, the selected tissue residue effects benchmarks would be protective of early life stage fish that may be exposed to sediment porewater.

Effects benchmarks for the five primary Site COCs are shown in Table 13. As discussed above, the effects benchmark for mercury is derived from a review of effect benchmarks shown in Table 12. Effects benchmarks for the ten Site HPAHs are based on FCVs or similar benchmarks and are derived directly from USEPA (2003b). The effects benchmark for copper for both sediment and porewater approaches are derived directly from USEPA (2005a).

The effects benchmark for PCBs is based on a FCV derived from studies with paired median lethal concentration ( $LC_{50}$ ) values and chronic effects for Aroclor 1242, 1248, and 1254 with aquatic invertebrates (Fuchsman et al., 2006). Although the EqP approach typically uses effects benchmark data derived from both invertebrates and fish, the mode of toxic action exerted by PCBs may differ significantly between invertebrates and fish. Specifically, the toxicity of planar PCB congeners to vertebrates is mediated by the aryl hydrocarbon (Ah) receptor, which may result in lower effects benchmarks for many vertebrates than for invertebrates, as invertebrates generally lack the Ah receptor (Fuchsman et al., 2006).

The effects benchmark for TBT is based on a FCV derived from lethal and sublethal endpoints for various aquatic organisms (USEPA, 2003a). The FCV selected as the effects benchmark in

this analysis, 0.0658 µg/L, is the initial FCV protective of aquatic life developed by USEPA (2003a). A more conservative FCV (0.0074 µg/L) was also provided in USEPA (2003b) to explicitly protect the growth of certain species such as commercially-important mollusks, the copepod *Acartia tonsa*, and the gastropod *Nucella lapillus*. The less conservative, but more generalized FCV value of 0.0658 µg/L was selected for this analysis because the State Water Board Resolution No. 92-49 approach used by CRWQCB (2010) to address Aquatic Life BUI does not appear to identify particular species of importance, but rather, focused on addressing Aquatic Life BUI of the entire benthic community. The level of protection provided by the species-specific USEPA (2003a) FCV of 0.0074 µg/L may or may not be appropriate as a benchmark for the protection of the entire benthic community. However, comparison between station-specific exposure estimates and this FCV is discussed quantitatively with regards to the likelihood of TBT effects on mollusks, copepods, and gastropods.

### 3.3 Comparison of Exposure Estimates to Effects Benchmarks

Chemical-specific exposure estimates for each of the 13 stations are compared to effects benchmarks via calculation of Toxic Units (TUs). TU values are calculated by dividing the exposure estimate by the effects benchmark (USEPA, 2003b). Calculation of TUs is directly analogous to the method in which measures of exposure are compared to effects benchmarks for the evaluation of potential non-cancer hazards in a human health risk assessment (e.g., the Hazard Index, HI) and risk potential to wildlife in ecological risk assessments (e.g., the Hazard Quotient, HQ) (Suter et al., 2000). Analogous to the evaluation of HQ and HI values, TU values are evaluated individually on a COC-by-COC basis and are not usually summed to estimate the cumulative contribution of all chemicals to risk potential (Suter et al., 2000). The exception of this general rule applies to PAHs. PAHs are assumed to act via the same mode of toxic action, and thus, the TUs for each of the ten Site HPAHs are summed, per USEPA (2003b) guidance.

Two TU values were calculated for copper because two different, complimentary approaches were used to estimate copper exposure on a sediment porewater basis. Exposure estimates for the two methods are not directly comparable due to a difference in measurement units (mmol/kg, TOC for the AVS-normalized porewater approach and µg/L for the porewater method). Because both may overestimate exposure (and thus, risk potential), the lesser TU value of the two approaches at each station was used as the indicator of station-specific copper risk potential. It should be noted that the algorithm for selection of the lowest estimates of exposure/risk potential was also followed for the other COCs.

As for HI and HQ values, risk potential for a particular exposure scenario is indicated when the threshold value of 1 is exceeded (Suter et al., 2000). When an HQ or HI values for a particular chemical are less than 1 for all relevant exposure scenarios, that chemical is usually excluded from further consideration for risk potential, possible effects, and remedial action. For example, among all aquatic-dependent wildlife representative species evaluated in the Site "Tier 2" ecological risk assessment, the maximum HQ value for nickel was 0.51. Because Site HQ values were below 1, CRWQCB (2010) did not conclude nickel posed a risk to aquatic dependent wildlife (Table 24-2 in CRWQCB, 2010). The same conclusion was found for HI values for the human health risk assessment at the Site (CRWQCB, 2010). Because HQ and HI values were both below 1, nickel was excluded as an aquatic-dependent wildlife and human

health chemical of concern and was not used to identify areas for remedial action to address Site Aquatic-Dependent and Human Health BUIs.

### 3.4 Results

#### 3.4.1 Overview of TU Values

Exposure estimation for PCBs, HPAHs, copper, and TBT in sediment porewater and subsequent comparison to respective porewater-based effects benchmarks are provided in Tables 14, 15, 16, and 17, respectively. Exposure estimation for mercury in invertebrate tissue and subsequent comparison to the mercury tissue-based effects benchmarks are provided in Table 18. Risk potential was identified for HPAHs, copper, and TBT ( $TU > 1$ ). Risk potential was not identified for mercury or PCBs. Exposure estimates for mercury and PCBs were 4 to 20 and 5 to 90 times and lower than effects benchmarks, respectively.

Risk potential ( $TU > 1$ ) was found for at least one COC at 7 of the 13 stations exhibiting Aquatic Life BUI, indicating causal links between Aquatic Life BUI and TBT, copper, and HPAHs (Table 19; Figure 3). The analysis did not identify a  $TU > 1$  for any of the five Site primary COCs at six stations (NA22, SW05, SW20, SW22, SW23, SW28), suggesting an uncertain link between this analysis and Aquatic Life BUI. It is possible that physical sediment properties, habitat conditions, or shipyard-associated disturbances are responsible for Aquatic Life BUI at these stations (CRWQCB, 2010).

#### 3.4.2 Consideration of Additional Lines of Evidence

TBT was the COC most-often identified COC with Aquatic Life risk potential by this analysis, because approximately 70% (5 of 7) of the stations that indicated risk potential for at least one chemical exhibited TBT TU values greater than 1. If the TBT effects benchmark ( $0.0658 \mu\text{g/L}$ ) is replaced by the USEPA (2003a) species-specific FCV ( $0.0074 \mu\text{g/L}$ ), TU values for TBT at each of the 13 stations exhibiting Aquatic Life BUI would be greater than 1, suggesting the 13 stations were impaired due to TBT. As discussed above, it is uncertain whether this benchmark would apply to the CRWQCB (2010) Aquatic Life BUI approach.

Evaluation of available measurements of COCs in invertebrate tissues (available at 4 of the 13 stations: NA19, SW04, SW13, and SW28) confirms causal links between Aquatic Life BUIs and TBT and HPAHs (a copper tissue approach is not relevant, as noted above):

- Concentrations of TBT of laboratory mussels from the bioaccumulation test by Exponent (2003) with Site sediment collected at SW04 and SW13 were as high as 720 and 150  $\mu\text{g/kg, ww}$ , respectively. This is within or exceeds the range of 140-640  $\mu\text{g/kg, ww}$  associated with growth and reproductive effects on aquatic invertebrates and fish (Meador, 2011). Porewater exposure-based TU values for TBT at these stations were 2.1 to 8.4, suggesting agreement between risk potential indicated by the tissue evaluation and risk potential indicated by the porewater evaluation (Table 17). The TBT TU at NA19 was 1.6, whereas the concentration in Site mussel was only 94  $\mu\text{g/kg, ww}$  (Exponent, 2003), below the thresholds identified by Meador (2011).
- Invertebrate tissue concentrations of HPAHs were not measured at the stations (SW10 and SW24) implicated by the porewater-based approach in this analysis (Table 15). The

highest concentrations of HPAHs in Site invertebrate tissue were found in laboratory invertebrates exposed to SW04 (1,500-2,000 µg/kg, ww, corresponding to 1-1.3 µmol/g lipid weight). These values are below the tissue effects benchmarks of 2.24 µmol/g, lipid weight identified by USEPA (2003b). If addressed according to TU-based approach, values of 1-1.3 corresponds to a tissue-based TU of approximately 0.5, which is in agreement with the porewater-based HPAH TU result of 0.37 observed for SW04 (Table 15). This suggests that a tissue-based approach using measured concentrations in invertebrates would confirm the identification of risk potential due to HPAHs at stations SW10 and SW24.

Evaluation of the available measurements of PCBs in invertebrate tissues confirms the absence of benthic invertebrate risk potential for PCBs. SW04 exhibited the highest PCB porewater-based TU (0.21, Table 14) and highest concentration of total PCB homologues in invertebrate tissues (300 µg/kg, ww). This tissue concentration is 2 to 3 orders of magnitude lower than concentrations associated with lethal effects in invertebrates (29,000-294,000 µg/kg, ww; Wenning et al., 2011). The range of whole-body tissue concentrations of PCBs in invertebrates associated with ecologically-relevant adverse sublethal effects (development, reproduction, growth) was also consulted via a review of individual studies highlighted in USACOE (2010) ERED. Only studies of at least seven days' duration were considered, because short-term (acute) sublethal effects (as in Hansen et al., 1974) were associated with exposure to very high levels of waterborne PCBs in which PCB toxicity may operate by a different mechanism and is less relevant to the comparatively low-level, long-term exposures occurring at the Site. Among the results from numerous studies referenced in the ERED (Duke et al., 1970; Lowe et al., 1972; Sanders and Chandler, 1972; Nebeker et al., 1974; Dillon et al., 1990; Fisher et al., 1999; Hwang et al., 2001), all ecologically-relevant effects-associated tissue concentrations were above 8,100 µg/kg, ww (Duke et al., 1970), except for slight adverse effects on growth observed in *Daphnia magna* (Dillon et al., 1990). These values ranged from 700 to 11,000 µg/kg, ww, as found in individuals exposed to two exposure levels of PCB congener PCB-101. Effects on growth were characterized as "minimal" by Dillon et al. (1990). Both exposure levels resulted in approximately the same reduction in growth (20% reduction compared to control exposures). Concentrations in PCBs in invertebrates at the Site are 2 to 40 times lower than this range of effect concentrations in tissue and confirm the lack of risk potential for PCBs indicated by the porewater approach (Table 14).

The range of concentrations of PCBs among the 13 stations at which Aquatic Life BUI was implicated, 250-5,200 µg/kg, was also found to be non-toxic to invertebrates exposed to sediments at Fox River, Green Bay, WI, USA and Waukegan Harbor, IL, USA by Becker and Ginn (2007). Although Becker and Ginn (2007) did not attempt a causal SGQ approach such as that used in this analysis, such an analysis as applied to the range of concentrations of PCBs in the non-toxic sediments in their study would also likely indicate a lack of risk potential for PCBs.

### **3.4.3 Implications of Results for Aquatic Life BUI COC Status for PCBs and Mercury**

PCBs and mercury are present in Site sediments at non-toxic concentrations that are below thresholds for PCB- and mercury-associated Aquatic Life BUI (Table 14 and Table 18). As is shown in Section 2.2, the calculation of LAET and SS-Median values in datasets in which a

COC is present at non-toxic exposure levels results in an inaccurate and misleading value because adverse effects are due to other chemicals. As applied, the resulting LAET and SS-Median values for PCBs and mercury are not technically defensible or consistent with established scientific principle for characterizing Aquatic Life BUI at Non Triad Stations. Additionally, the 400 µg/kg PCBs SQG used in the SQGQ1 metric greatly overestimates the predicted Site specific risk potential by a factor of 20 to 170 (analysis via substituting a value of 400 µg/kg for the concentration of PCBs in sediment in Table 14). As such, the SQGQ1 metric and Triad Approach are critically flawed in the evaluation of PCBs in Site sediment.

There is an absence of risk potential for PCBs and mercury at the Site, as indicated by direct causal analysis. These chemicals should be excluded as COCs associated with Aquatic Life BUI. TU values for these chemicals were both substantially less than 1 at all stations exhibiting Aquatic Life BUI (Table 14 and Table 18). The CRWQCB (2010) decision-making approach clearly supports exclusion of chemicals for which toxic thresholds are not exceeded from further risk-based decision making (e.g., nickel, which was excluded due to lack of threshold exceedance in human health and wildlife risk assessments). Thus, PCBs and mercury should be excluded as COCs associated with Aquatic Life BUI. As such, PCBs and mercury should also be excluded from evaluations that aim to identify areas of the Site exhibiting Aquatic Life BUI and any ranking of such areas for potential sediment remediation. These evaluations include the currently flawed sediment chemistry line of evidence within the Triad Approach and the SS-MEQ and 60% LAET lines of evidence in the Non-Triad Data Approach.

## 4 Conclusion

The CRWQCB (2010) evaluation of chemicals in sediment with respect to characterizing Aquatic Life BUI is inappropriate for determining the specific contaminants responsible for Aquatic Life BUI at the Site. The analysis presented herein explicitly evaluated causality between individual chemicals and Aquatic Life BUI in a manner that includes the pesticide TBT and explicitly considers bioavailability of the five Site primary COCs. Results indicate that only TBT, HPAHs, and copper should be included as COCs associated with Aquatic Life impairment. PCBs and mercury should be excluded as COCs in the Aquatic Life BUI assessment excluded from decision making regarding efforts to restore Aquatic Life Beneficial Uses at the Site.

Because aquatic dependent wildlife and human health BUI considerations should be withdrawn by CRWQCB due to lack of evidence for wildlife risk and human health risks at the Site (Exponent, 2003; Conder, 2011), the Site remedial footprint should only focus on addressing Aquatic Life BUI. Among the BUIs investigated at the Site by CRWQCB (2010), scientific evidence most-strongly supports the presence of an Aquatic Life BUI. The analysis presented in this report has identified areas of the Site with potential risk to benthic invertebrates for Site primary COCs HPAHs, TBT, and copper (Figure 3). Because the analysis presented in this report is more scientifically rigorous than the CRWQCB (2010) evaluation of sediment chemistry causality for Aquatic Life BUI, the Site remedy footprint should be restricted to the areas with TU values greater than one, as shown in Figure 3.

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## Tables

**Table 1. Stations Exhibiting Aquatic Life Beneficial Use Impairment**

<b>Station</b>	<b>Triad Data</b>	<b>Triad Data Approach Result</b>	<b>Non-Triad Approach Thresholds Exceeded</b>
NA19	Yes	Likely	Not Applicable
NA22	Yes	Likely	Not Applicable
SW01	No	Not Applicable	SS-MEQ
SW04	Yes	Likely	Not Applicable
SW05	No	Not Applicable	SS-MEQ
SW10	No	Not Applicable	60% LAET exceedance for HPAH
SW13	Yes	Likely	Not Applicable
SW16	No	Not Applicable	SS-MEQ
SW20	No	Not Applicable	SS-MEQ
SW22	Yes	Likely	Not Applicable
SW23	Yes	Likely	Not Applicable
SW24	No	Not Applicable	60% LAET for HPAH and SS-MEQ
SW28	No	Not Applicable	60% LAET for HPAH and SS-MEQ

**Notes:**

- <sup>1</sup> Stations with benthic Beneficial Use Impairment (BUI) were identified as the stations listed with a 'Likely' Triad Line of Evidence (LOE) Category as referenced from 'Table 18-1 Results of the Sediment Quality Triad Lines-of-Evidence' (CRWQRB, 2010).
- <sup>2</sup> Stations with benthic BUI were also identified as the stations with Non-Triad Threshold Exceedances of either SS-MEQ or 60% LAET as referenced from 'Table 32-23 Site-Specific 60%LAET and SS-MEQ Threshold Exceedances SPI Successional Stage, and Remedial Designations at the Shipyard Sediment Site Non-Triad Stations' (CRWQRB, 2010).
- <sup>3</sup> SS-MEQ = Site-Specific Median Effects Quotient.
- <sup>4</sup> LAET = Lowest Apparent Effects Threshold.
- <sup>5</sup> HPAH = High Molecular Weight Polycyclic Aromatic Hydrocarbons.

**Table 2. Measured Concentration of PCB Homologs in Surface Sediment**

Station	Sample Number	Date	Monochlorobiphenyl (µg/kg, dw)	Dichlorobiphenyl (µg/kg, dw)	Trichlorobiphenyl (µg/kg, dw)	Tetrachlorobiphenyl (µg/kg, dw)	Pentachlorobiphenyl (µg/kg, dw)	Hexachlorobiphenyl (µg/kg, dw)	Heptachlorobiphenyl (µg/kg, dw)	Octachlorobiphenyl (µg/kg, dw)	Nonachlorobiphenyl (µg/kg, dw)	Decachlorobiphenyl (µg/kg, dw)	Total PCB Homologs (µg/kg, dw)
NA19	SD0042	8/12/2001	0.24	1.6	11	150	610	450	120	18	5.2	3.7	1,400
NA22	SD0052	8/14/2001	0.14	0.88	6.7	31	78	77	42	9.9	2.1	1.9	250
SW01	SD0001	8/6/2001	3	12	210	640	740	490	200	65	11	1.7	2,400
SW01b	SD0137	9/14/2002	0.56	4.8	110	450	910	660	210	41	8.6	2.3	2,400
SW01	Average	--	1.78	8.4	160	545	825	575	205	53	9.8	2.0	2,400
SW04	SD0012	8/7/2001	12	36	230	1,000	2,500	1,500	410	110	20	3.2	5,800
SW04b	SD0112	9/10/2002	1.7	11	150	800	1,800	1,300	410	95	35	7	4,600
SW04	Average	--	6.85	23.5	190	900	2,150	1,400	410	103	27.5	5.1	5,200
SW05	SD0003	8/6/2001	0.79	5.8	88	400	640	450	140	54	14	2.6	1,800
SW10	SD0008	8/6/2001	0.47	10	120	240	290	180	64	19	4.5	1.6	930
SW13	SD0022	8/9/2001	0.32	1.4	13	88	210	240	120	31	7.3	3.9	710
SW16	SD0025	8/10/2001	0.24	1.2	9.8	70	180	210	110	26	5	2.9	610
SW20	SD0059	8/15/2001	0.31	1.2	21	250	430	930	750	170	13	2.6	2,600
SW22	SD0060	8/15/2001	0.23	1.4	22	160	320	470	320	72	8.4	3.6	1,400
SW23	SD0058	8/15/2001	0.36	1.3	15	150	360	550	370	82	9.6	3.3	1,500
SW24	SD0015	8/8/2001	0.17	1	13	140	300	510	360	87	8.3	2.9	1,400
SW24b	SD0113	9/10/2002	0.18	1.4	13	130	300	600	430	77	11	3	1,600
SW24	Average	--	0.18	1.2	13	135	300	555	395	82	9.65	2.95	1,500
SW28	SD0029	8/11/2001	0.38	1.5	10	100	420	1,200	980	200	13	2.7	3,000
SW28b	SD0121	9/11/2002	0.24	1.5	9.9	88	370	890	670	120	14	3.9	2,200
SW28	Average	--	0.31	1.5	9.95	94	395	1,045	825	160	13.5	3.3	2,600

Station	Monochlorobiphenyl (µg/kg, dw)	Dichlorobiphenyl (µg/kg, dw)	Trichlorobiphenyl (µg/kg, dw)	Tetrachlorobiphenyl (µg/kg, dw)	Pentachlorobiphenyl (µg/kg, dw)	Hexachlorobiphenyl (µg/kg, dw)	Heptachlorobiphenyl (µg/kg, dw)	Octachlorobiphenyl (µg/kg, dw)	Nonachlorobiphenyl (µg/kg, dw)	Decachlorobiphenyl (µg/kg, dw)	Total PCB Homologs (µg/kg, dw)
NA19	0.24	1.6	11	150	610	450	120	18	5.2	3.7	1,400
NA22	0.14	0.88	6.7	31	78	77	42	9.9	2.1	1.9	250
SW01	1.78	8.4	160	545	825	575	205	53	9.8	2.0	2,400
SW04	6.85	23.5	190	900	2,150	1,400	410	103	27.5	5.1	5,200
SW05	0.79	5.8	88	400	640	450	140	54	14	2.6	1,800
SW10	0.47	10	120	240	290	180	64	19	4.5	1.6	930
SW13	0.32	1.4	13	88	210	240	120	31	7.3	3.9	710
SW16	0.24	1.2	9.8	70	180	210	110	26	5	2.9	610
SW20	0.31	1.2	21	250	430	930	750	170	13	2.6	2,600
SW22	0.23	1.4	22	160	320	470	320	72	8.4	3.6	1,400
SW23	0.36	1.3	15	150	360	550	370	82	9.6	3.3	1,500
SW24	0.18	1.2	13	135	300	555	395	82	9.65	2.95	1,500
SW28	0.31	1.5	9.95	94	395	1,045	825	160	13.5	3.3	2,600

**Notes:**

- <sup>1</sup> Raw data in Table 2.A. is referenced from 'Table B1-8 PCB congener and homolog results for surface sediment samples' in Exponent, 2003.
- <sup>2</sup> All surface sediment samples were collected from a depth interval of 0–2 cm.
- <sup>3</sup> Field splits and duplicates are averaged.
- <sup>4</sup> Stations ending with the letter b indicate the sample was collected for the porewater study and was analyzed for PCB homologs only.
- <sup>5</sup> In Table 2.B. the data is summarized by station.
- <sup>6</sup> PCB = polychlorinated biphenyl.
- <sup>7</sup> µg/kg, dw = micrograms per kilogram, dry weight.

**Table 3. Measured Concentration of HPAHs in Surface Sediment**

Station	Sample Number	Date	Fluoranthene (µg/kg, dw)	Pyrene (µg/kg, dw)	Benz[a]-anthracene (µg/kg, dw)	Chrysene (µg/kg, dw)	Benzo[b]-fluoranthene (µg/kg, dw)	Benzo[k]-fluoranthene (µg/kg, dw)	Benzo[a]-pyrene (µg/kg, dw)	Indeno-[1,2,3-cd]-pyrene (µg/kg, dw)	Dibenz[a,h]-anthracene (µg/kg, dw)	Benzo[ghi]perylene (µg/kg, dw)	Total HPAH (to two significant figures) (µg/kg, dw)
NA19	SD0042	8/12/2001	190	200	170	330	500	350	460	410	76	330	3,000
NA22	SD0052	8/14/2001	210	590	170	350	610	470	540	330	70	290	3,600
SW01	SD0001	8/6/2001	1,100	1,400	760	1,300	1,400	1,100	1,500	980	190	760	10,000
SW01	SD0169	11/6/2002	470	830	340	600	920	470	680	410	84	300	5,100
SW01	SD0171	11/6/2002	480	800	360	490	780	500	750	440	81	330	5,000
SW01	Average - Field Split	--	475	815	350	545	850	485	715	425	83	315	5,100
SW01	Average - Duplicate	--	788	1,108	555	923	1,125	793	1,108	703	136	538	7,500
SW04	SD0012	8/7/2001	2,100	2,000	1,100	1,800	1,600	1,300	1,500	880	230	640	13,000
SW04	SD0170	11/6/2002	1,400	1,400	1,400	2,600	2,400	1,500	2,100	1,200	230	770	15,000
SW04	Average	--	1,750	1,700	1,250	2,200	2,000	1,400	1,800	1,040	230	705	14,000
SW05	SD0003	8/6/2001	2,300	2,200	910	1,500	1,500	1,200	1,500	790	200	580	13,000
SW10	SD0008	8/6/2001	4,000	3,500	1,200	1,500	1,600	1,200	1,600	790	200	590	16,000
SW13	SD0022	8/9/2001	2,200	1,400	860	1,900	1,500	1,300	1,400	860	170	640	12,000
SW16	SD0025	8/10/2001	260	720	210	400	1,100	790	1,000	600	130	500	5,700
SW20	SD0059	8/15/2001	930	1,200	760	1,800	1,500	1,200	1,400	970	200	770	11,000
SW22	SD0060	8/15/2001	910	1,100	890	1,900	1,800	1,300	1,700	1,100	230	830	12,000
SW23	SD0058	8/15/2001	960	1,000	850	1,800	1,500	1,200	1,500	1,000	220	820	11,000
SW24	SD0015	8/8/2001	7,100	3,100	6,300	11,000	7,000	7,300	8,800	3,700	1,100	2,800	58,000
SW24	SD0173	11/6/2002	3,600	9,800	2,700	4,700	7,300	5,500	7,200	2,800	750	1,700	46,000
SW24	Average	--	5,350	6,450	4,500	7,850	7,150	6,400	8,000	3,250	925	2,250	52,000
SW28	SD0029	8/11/2001	1,300	1,400	1,900	4,300	2,900	2,100	3,100	1,700	330	1,100	20,000
SW28	SD0177	11/7/2002	1,400	1,600	970	2,000	2,300	1,500	2,000	1,200	230	740	14,000
SW28	Average	--	1,350	1,500	1,435	3,150	2,600	1,800	2,550	1,450	280	920	17,000

Station	Fluoranthene (µg/kg, dw)	Pyrene (µg/kg, dw)	Benz[a]-anthracene (µg/kg, dw)	Chrysene (µg/kg, dw)	Benzo[b]-fluoranthene (µg/kg, dw)	Benzo[k]-fluoranthene (µg/kg, dw)	Benzo[a]-pyrene (µg/kg, dw)	Indeno-[1,2,3-cd]-pyrene (µg/kg, dw)	Dibenz[a,h]-anthracene (µg/kg, dw)	Benzo[ghi]perylene (µg/kg, dw)	Total HPAH (µg/kg, dw)
NA19	190	200	170	330	500	350	460	410	76	330	3,000
NA22	210	590	170	350	610	470	540	330	70	290	3,600
SW01	790	1,100	560	920	1,100	790	1,100	700	140	540	7,500
SW04	1,800	1,700	1,300	2,200	2,000	1,400	1,800	1,000	230	710	14,000
SW05	2,300	2,200	910	1,500	1,500	1,200	1,500	790	200	580	13,000
SW10	4,000	3,500	1,200	1,500	1,600	1,200	1,600	790	200	590	16,000
SW13	2,200	1,400	860	1,900	1,500	1,300	1,400	860	170	640	12,000
SW16	260	720	210	400	1,100	790	1,000	600	130	500	5,700
SW20	930	1,200	760	1,800	1,500	1,200	1,400	970	200	770	11,000
SW22	910	1,100	890	1,900	1,800	1,300	1,700	1,100	230	830	12,000
SW23	960	1,000	850	1,800	1,500	1,200	1,500	1,000	220	820	11,000
SW24	5,400	6,500	4,500	7,900	7,200	6,400	8,000	3,300	930	2,300	52,000
SW28	1,400	1,500	1,400	3,200	2,600	1,800	2,600	1,500	280	920	17,000

**Notes:**

- <sup>1</sup> Raw data in Table 3.A. is referenced from 'Table B1-5 Polycyclic aromatic hydrocarbon results for surface sediment samples' and 'Table B1-6 Alkylated polycyclic aromatic hydrocarbon results for surface sediment samples' in Exponent, 2003.
- <sup>2</sup> All surface sediment samples were collected from a depth interval of 0–2 cm.
- <sup>3</sup> Field splits and duplicates are averaged. If field splits are taken from a duplicate sample, the field split is averaged, then the duplicates are averaged.
- <sup>4</sup> In Table 3.B. the data is summarized by station and represented to two significant figures.
- <sup>5</sup> HPAH = High Molecular Weight Polycyclic Aromatic Hydrocarbon.
- <sup>6</sup> Total HPAH is computed as the sum of the concentrations of fluoranthene, pyrene, benz[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene, dibenz[a,h]anthracene, and benzo[ghi]perylene. Total HPAH is calculated to two significant figures. The field splits and duplicate averages are calculated based on values with two significant figures.
- <sup>7</sup> µg/kg, dw = micrograms per kilogram, dry weight.

**Table 4. Measured Concentration of AVS and SEM for Copper in Surface Sediment**

Station	Sample Number	Date	AVS (mg/kg, dw)	SEM for Copper (mg/kg, dw)
NA19	SD0042	8/12/2001	130	280
NA22	SD0052	8/14/2001	180	120
SW01	SD0001	8/6/2001	57	350
SW04	SD0012	8/7/2001	35	1100
SW05	SD0003	8/6/2001	190	200
SW10	SD0008	8/6/2001	610	140
SW13	SD0022	8/9/2001	220	1200
SW16	SD0025	8/10/2001	280	500
SW20	SD0059	8/15/2001	370	260
SW22	SD0060	8/15/2001	60	300
SW23	SD0058	8/15/2001	160	270
SW24	SD0015	8/8/2001	60	240
SW28	SD0029	8/11/2001	450	270

**Notes:**

- <sup>1</sup> Raw data is referenced from 'Table B1-4 Acid-volatile sulfide and simultaneously extracted metal results for surface sediment samples' in Exponent, 2003.
- <sup>2</sup> All surface sediment samples were collected from a depth interval of 0–2 cm.
- <sup>3</sup> AVS = Acid-Volatile Sulfide.
- <sup>4</sup> SEM = Simultaneously Extracted Metals.
- <sup>5</sup> mg/kg, dw = milligrams per kilogram, dry weight.



**Table 5. Measured TOC Content in Surface Sediment**

<b>Table 5.A. Raw Data (with field split and duplicate averaging)</b>			
<b>Station</b>	<b>Sample Number</b>	<b>Date</b>	<b>TOC (%, dw)</b>
NA19	SD0042	8/12/2001	1.84
NA22	SD0052	8/14/2001	1.65
NA22	SD0129	9/12/2002	NR
NA22	SD0132	9/12/2002	NR
NA22	Average - Field Split	--	NR
NA22	Average - Duplicate	--	1.65
SW01	SD0001	8/6/2001	2.25
SW01	SD0137	9/14/2002	2.31
SW01	SD0169	11/6/2002	2.18
SW01	SD0171	11/6/2002	2.14
SW01	Average - Field Split	--	2.16
SW01	Average - Duplicate	--	2.24
SW04	SD0012	8/7/2001	1.59
SW04	SD0112	9/10/2002	3.01
SW04	SD0170	11/6/2002	2.24
SW04	Average	--	2.28
SW05	SD0003	8/6/2001	1.55
SW10	SD0008	8/6/2001	1.21
SW13	SD0022	8/9/2001	2.33
SW16	SD0025	8/10/2001	2.24
SW20	SD0059	8/15/2001	2.14
SW22	SD0060	8/15/2001	2.46
SW23	SD0058	8/15/2001	2.52
SW24	SD0015	8/8/2001	1.61
SW24	SD0113	9/10/2002	2.06
SW24	SD0173	11/6/2002	1.59
SW24	Average	--	1.75
SW28	SD0029	8/11/2001	2.53
SW28	SD0121	9/11/2002	2.6
SW28	SD0177	11/7/2002	2.42
SW28	Average	--	2.52

<b>Table 5.B. Data Summarized By Station</b>	
<b>Station</b>	<b>TOC (%, dw)</b>
NA19	1.84
NA22	1.65
SW01	2.24
SW04	2.28
SW05	1.55
SW10	1.21
SW13	2.33
SW16	2.24
SW20	2.14
SW22	2.46
SW23	2.52
SW24	1.75
SW28	2.52

**Notes:**

- <sup>1</sup> Raw data in Table 5.A. is referenced from 'Table B1-1. Conventional results for surface sediment samples' in Exponent, 2003.
- <sup>2</sup> All surface sediment samples were collected from a depth interval of 0–2 cm.
- <sup>3</sup> Field splits and duplicates are averaged. If field splits are taken from a duplicate sample, the field split is averaged, then the duplicates are averaged.
- <sup>4</sup> In Table 5.B. the data is summarized by station.
- <sup>5</sup> TOC = Total Organic Carbon.
- <sup>6</sup> %, dw = percent, dry weight.

**Table 6. Measured Concentration of Copper, Mercury, and Tributyltin in Surface Sediment**

Station	Sample Number	Date	Copper (mg/kg, dw)	Mercury (mg/kg, dw)	Tributyltin (µg/kg, dw)
NA19	SD0042	8/12/2001	270	0.78	570
NA22	SD0052	8/14/2001	150	0.38	120
SW01	SD0001	8/6/2001	620	1.4	520
SW01	SD0137	9/14/2002	500	1.5	380
SW01	Average	--	560	1.45	450
SW04	SD0012	8/7/2001	1,900	1.2	2800
SW04	SD0112	9/10/2002	1,100	2.3	3,700
SW04	Average	--	1,500	1.75	3250
SW05	SD0003	8/6/2001	230	0.96	170
SW10	SD0008	8/6/2001	160	0.58	250
SW13	SD0022	8/9/2001	800	0.86	790
SW16	SD0025	8/10/2001	430	1	1100
SW20	SD0059	8/15/2001	290	0.99	130
SW22	SD0060	8/15/2001	260	1.1	190
SW23	SD0058	8/15/2001	280	1	210
SW24	SD0015	8/8/2001	260	1.6	170
SW24	SD0113	9/10/2002	340	2.2	160
SW24	Average	--	300	1.9	165
SW28	SD0029	8/11/2001	270	0.98	180
SW28	SD0121	9/11/2002	260	0.77	120
SW28	Average	--	265	0.875	150

Station	Copper (mg/kg, dw)	Mercury (mg/kg, dw)	Tributyltin (µg/kg, dw)
NA19	270	0.78	570
NA22	150	0.38	120
SW01	560	1.45	450
SW04	1500	1.75	3250
SW05	230	0.96	170
SW10	160	0.58	250
SW13	800	0.86	790
SW16	430	1	1100
SW20	290	0.99	130
SW22	260	1.1	190
SW23	280	1	210
SW24	300	1.9	165
SW28	265	0.88	150

**Notes:**

- <sup>1</sup> Raw data in Table 6.A. is referenced from 'Table B1-3. Metal and butyltin results for surface sediment samples' in Exponent, 2003.
- <sup>2</sup> All surface sediment samples were collected from a depth interval of 0–2 cm.
- <sup>3</sup> Field splits and duplicates are averaged.
- <sup>4</sup> In Table 6.B. the data is summarized by station.
- <sup>5</sup> mg/kg, dw = milligrams per kilogram, dry weight.
- <sup>6</sup> µg/kg, dw = micrograms per kilogram, dry weight.

Table 7. Measured Concentration of Copper and Tributyltin in Porewater

Station	Sample Number	Date	Copper (µg/L)	Tributyltin (µg/L)
SW01	PW0018W	9/14/2002	17	0.037
SW04	PW0006W	9/10/2002	55	0.55
SW24	PW0007W	9/10/2002	25	0.074
SW28	PW0010W	9/11/2002	19	<b>0.016</b>

Station	Copper (µg/L)	Tributyltin (µg/L)
NA19	NR	NR
NA22	NR	NR
SW01	17	0.04
SW04	55	0.55
SW05	NR	NR
SW10	NR	NR
SW13	NR	NR
SW16	NR	NR
SW20	NR	NR
SW22	NR	NR
SW23	NR	NR
SW24	25	0.07
SW28	19	0.02

**Notes:**

- <sup>1</sup> Raw data in Table 7.A. is referenced from 'Table D-1. Metal and butyltin results for porewater samples' in Exponent, 2003.
- <sup>2</sup> Sediment for porewater extraction was collected from a depth interval of 0–2 cm.
- <sup>3</sup> In Table 7.B. the data is summarized by station.
- <sup>4</sup> Bold and italicized values indicate concentrations are below detection. Non-detect concentrations are represented as half of the detection limit.
- <sup>5</sup> µg/L = micrograms per liter.
- <sup>6</sup> NR = Not Reported.

Table 8. Measured Concentration of HPAHs in Porewater

Station	Sample Number	Date	Fluoranthene (µg/L)	Pyrene (µg/L)	Benz[a]-anthracene (µg/L)	Chrysene (µg/L)	Benzo[b]-fluoranthene (µg/L)	Benzo[k]-fluoranthene (µg/L)	Benzo[a]-pyrene (µg/L)	Indeno-[1,2,3-cd]-pyrene (µg/L)	Dibenz[a,h]-anthracene (µg/L)	Benzo[ghi]-perylene (µg/L)	Total HPAH (µg/L)
SW01	PW0023W	11/6/2002	0.037	0.130	0.023	0.026	0.089	0.070	0.070	0.042	<b>0.011</b>	0.032	0.53
SW04	PW0026W	11/6/2002	0.073	0.110	0.041	0.064	0.099	0.062	0.067	0.045	<b>0.012</b>	0.030	0.6
SW04	Average	--	0.073	0.110	0.041	0.064	0.099	0.062	0.067	0.045	0.012	0.030	0.6
SW24	PW0027W	11/6/2002	0.069	0.180	0.040	0.066	0.270	0.200	0.250	0.090	<b>0.011</b>	0.059	1.2
SW28	PW0031W	11/7/2002	0.044	0.090	<b>0.010</b>	<b>0.010</b>	0.052	0.029	0.033	<b>0.010</b>	<b>0.010</b>	<b>0.010</b>	0.30

Station	Fluoranthene (µg/L)	Pyrene (µg/L)	Benz[a]-anthracene (µg/L)	Chrysene (µg/L)	Benzo[b]-fluoranthene (µg/L)	Benzo[k]-fluoranthene (µg/L)	Benzo[a]-pyrene (µg/L)	Indeno-[1,2,3-cd]-pyrene (µg/L)	Dibenz[a,h]-anthracene (µg/L)	Benzo[ghi]-perylene (µg/L)	Total HPAH (µg/L)
NA19	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
NA22	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW01	0.037	0.130	0.023	0.026	0.089	0.070	0.070	0.042	0.011	0.032	0.530
SW04	0.073	0.110	0.041	0.064	0.099	0.062	0.067	0.045	0.012	0.030	0.600
SW05	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW10	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW13	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW16	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW20	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW22	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW23	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW24	0.069	0.180	0.040	0.066	0.270	0.200	0.250	0.090	0.011	0.059	1.200
SW28	0.044	0.090	0.010	0.010	0.052	0.029	0.033	0.010	0.010	0.010	0.300

**Notes:**

- <sup>1</sup> Raw data in Table 8.A. is referenced from 'Table D-2. Polycyclic aromatic hydrocarbon and total organic carbon results for porewater samples' in Exponent, 2003.
- <sup>2</sup> Sediment for porewater extraction was collected from a depth interval of 0–2 cm.
- <sup>3</sup> Total HPAH is computed as the sum of the concentrations of fluoranthene, pyrene, benz[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene, dibenz[a,h]anthracene, and benzo[ghi]perylene.
- <sup>4</sup> Bold and italicized values indicate concentrations are below detection. Non-detect concentrations are represented as half of the detection limit. These values are used to calculate Total HPAH.
- <sup>5</sup> Field splits and duplicates are averaged.
- <sup>6</sup> In Table 8.B. the data is summarized by station.
- <sup>7</sup> NR = Not Reported.
- <sup>8</sup> HPAH = High Molecular Weight Polycyclic Aromatic Hydrocarbon.
- <sup>9</sup> µg/L = micrograms per liter.

Table 9. Measured Concentration of PCB Homologs in Porewater

Station	Sample Number	Date	Monochlorobiphenyl (ng/L)	Dichlorobiphenyl (ng/L)	Trichlorobiphenyl (ng/L)	Tetrachlorobiphenyl (ng/L)	Pentachlorobiphenyl (ng/L)	Hexachlorobiphenyl (ng/L)	Heptachlorobiphenyl (ng/L)	Octachlorobiphenyl (ng/L)	Nonachlorobiphenyl (ng/L)	Decachlorobiphenyl (ng/L)	Total PCB Homologs (ng/L)
SW01	PW0018W	9/14/2002	<b>0.405</b>	<b>0.405</b>	26	160	160	100	40	7.2	1.4	<b>0.405</b>	500
SW04	PW0006W	9/10/2002	<b>0.415</b>	<b>0.415</b>	16	130	220	160	59	12	3.5	1.2	600
SW24	PW0007W	9/10/2002	<b>0.495</b>	<b>0.495</b>	1.7	80	140	260	150	29	2.4	0.99	670
SW28	PW0010W	37510	<b>0.43</b>	<b>0.43</b>	5.5	18	57	130	71	10	1.2	<b>0.43</b>	290

Station	Monochlorobiphenyl (µg/L)	Dichlorobiphenyl (µg/L)	Trichlorobiphenyl (µg/L)	Tetrachlorobiphenyl (µg/L)	Pentachlorobiphenyl (µg/L)	Hexachlorobiphenyl (µg/L)	Heptachlorobiphenyl (µg/L)	Octachlorobiphenyl (µg/L)	Nonachlorobiphenyl (µg/L)	Decachlorobiphenyl (µg/L)	Total PCB Homologs (µg/L)
NA19	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
NA22	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW01	0.0004	0.0004	0.0260	0.1600	0.1600	0.1000	0.0400	0.0072	0.0014	0.0004	0.5000
SW04	0.0004	0.0004	0.0160	0.1300	0.2200	0.1600	0.0590	0.0120	0.0035	0.0012	0.6000
SW05	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW10	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW13	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW16	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW20	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW22	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW23	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW24	0.0005	0.0005	0.0017	0.0800	0.1400	0.2600	0.1500	0.0290	0.0024	0.0010	0.6700
SW28	0.0004	0.0004	0.0055	0.0180	0.0570	0.1300	0.0710	0.0100	0.0012	0.0004	0.2900

**Notes:**

- <sup>1</sup> Raw data in Table 9.A. is referenced from 'Table D-3. PCB homolog results for porewater samples' in Exponent, 2003.
- <sup>2</sup> Sediment for porewater extraction was collected from a depth interval of 0–2 cm.
- <sup>3</sup> Bold and italicized values indicate concentrations that are below the detection limit. Non-detect concentrations are represented as half of the detection limit. These values are used to calculate Total PCB Homologs.
- <sup>4</sup> In Table 9.B. the data is summarized by station and units are converted from ng/L to µg/L.
- <sup>5</sup> PCB = polychlorinated biphenyl.
- <sup>6</sup> ng/L = nanograms per liter.
- <sup>7</sup> µg/L = micrograms per liter.
- <sup>8</sup> NR = Not Reported.

**Table 10. Derivation of the Bioaccumulation Factor for Mercury**

Notes	1	2	3
<b>Station</b>	<b>Measured Concentration in Tissue (mg/kg, ww)</b>	<b>Pre-remedy SWAC (mg/kg, dw)</b>	<b>BAF (<math>\text{kg}_{\text{sediment, dw}}/\text{kg}_{\text{tissue, ww}}</math>)</b>
NA19	0.024	0.75	0.032
NA24	0.020		0.027
SW18	0.017		0.023
SW27	0.018		0.024
	Average		0.026

**Notes:**

<sup>1</sup> Measured concentration in mussel (soft tissue) is referenced from 'Table E-1. Lipid, solids, metal, and butyltin results for tissue samples' in Exponent (2003). Composite samples of benthic mussel tissues with shell removed (*Musculista senhousi*) were collected at two stations inside each shipyard, NASSCO and Southwest Marine.

<sup>2</sup> The pre-remedy surface-area weighted average concentration (SWAC) is referenced from 'Table 32-5 Current and Post-Remedial SWACs' of CRWQCB (2010).

<sup>3</sup> Bioaccumulation Factor (BAF) is calculated as the measured total mercury concentration in tissue divided by the pre-remedy SWAC, as in CRWQCB (2010).

<sup>4</sup> mg/kg, ww = milligrams per kilogram, wet weight.

<sup>5</sup> mg/kg, dw = milligrams per kilogram, dry weight.

<sup>6</sup>  $\text{kg}_{\text{sediment, dw}}/\text{kg}_{\text{tissue, ww}}$  = kilogram of sediment, dry weight per kilogram of tissue, wet weight.

Table 11. Octanol-water Partition Coefficient for the PCB Homologue Mixture in Surface Sediment

Station	Monochlorobiphenyl (µg/kg, dw)	Dichlorobiphenyl (µg/kg, dw)	Trichlorobiphenyl (µg/kg, dw)	Tetrachlorobiphenyl (µg/kg, dw)	Pentachlorobiphenyl (µg/kg, dw)	Hexachlorobiphenyl (µg/kg, dw)	Heptachlorobiphenyl (µg/kg, dw)	Octachlorobiphenyl (µg/kg, dw)	Nonachlorobiphenyl (µg/kg, dw)	Decachlorobiphenyl (µg/kg, dw)	Total PCB Homologs (µg/kg, dw)
NA19	0.24	1.6	11	150	610	450	120	18	5.2	3.7	1,400
NA22	0.14	0.88	6.7	31	78	77	42	9.9	2.1	1.9	250
SW01	1.78	8.4	160	545	825	575	205	53	9.8	2	2,400
SW04	6.85	23.5	190	900	2,150	1,400	410	102.5	27.5	5.1	5,200
SW05	0.79	5.8	88	400	640	450	140	54	14	2.6	1,800
SW10	0.47	10	120	240	290	180	64	19	4.5	1.6	930
SW13	0.32	1.4	13	88	210	240	120	31	7.3	3.9	710
SW16	0.24	1.2	9.8	70	180	210	110	26	5	2.9	610
SW20	0.31	1.2	21	250	430	930	750	170	13	2.6	2,600
SW22	0.23	1.4	22	160	320	470	320	72	8.4	3.6	1,400
SW23	0.36	1.3	15	150	360	550	370	82	9.6	3.3	1,500
SW24	0.175	1.2	13	135	300	555	395	82	9.65	2.95	1,500
SW28	0.31	1.5	9.95	94	395	1,045	825	160	13.5	3.3	2,600

Station	Monochlorobiphenyl (µg/µg <sub>Total PCB Homologs</sub> )	Dichlorobiphenyl (µg/µg <sub>Total PCB Homologs</sub> )	Trichlorobiphenyl (µg/µg <sub>Total PCB Homologs</sub> )	Tetrachlorobiphenyl (µg/µg <sub>Total PCB Homologs</sub> )	Pentachlorobiphenyl (µg/µg <sub>Total PCB Homologs</sub> )	Hexachlorobiphenyl (µg/µg <sub>Total PCB Homologs</sub> )	Heptachlorobiphenyl (µg/µg <sub>Total PCB Homologs</sub> )	Octachlorobiphenyl (µg/µg <sub>Total PCB Homologs</sub> )	Nonachlorobiphenyl (µg/µg <sub>Total PCB Homologs</sub> )	Decachlorobiphenyl (µg/µg <sub>Total PCB Homologs</sub> )
NA19	0.0002	0.0012	0.0080	0.1095	0.4453	0.3285	0.0876	0.0131	0.0038	0.0027
NA22	0.0006	0.0035	0.0268	0.1242	0.3125	0.3085	0.1683	0.0397	0.0084	0.0076
SW01	0.0007	0.0035	0.0671	0.2285	0.3459	0.2411	0.0860	0.0222	0.0041	0.0008
SW04	0.0013	0.0045	0.0364	0.1726	0.4122	0.2684	0.0786	0.0197	0.0053	0.0010
SW05	0.0004	0.0032	0.0490	0.2228	0.3565	0.2507	0.0780	0.0301	0.0078	0.0014
SW10	0.0005	0.0108	0.1291	0.2582	0.3120	0.1936	0.0688	0.0204	0.0048	0.0017
SW13	0.0004	0.0020	0.0182	0.1231	0.2937	0.3357	0.1679	0.0434	0.0102	0.0055
SW16	0.0004	0.0020	0.0159	0.1138	0.2926	0.3414	0.1788	0.0423	0.0081	0.0047
SW20	0.0001	0.0005	0.0082	0.0973	0.1674	0.3621	0.2920	0.0662	0.0051	0.0010
SW22	0.0002	0.0010	0.0160	0.1161	0.2323	0.3412	0.2323	0.0523	0.0061	0.0026
SW23	0.0002	0.0008	0.0097	0.0973	0.2335	0.3568	0.2400	0.0532	0.0062	0.0021
SW24	0.0001	0.0008	0.0087	0.0904	0.2008	0.3715	0.2644	0.0549	0.0065	0.0020
SW28	0.0001	0.0006	0.0039	0.0369	0.1551	0.4102	0.3238	0.0628	0.0053	0.0013

**Notes:**

- Measured Concentration of PCB Homologs in Surface Sediment are referenced from Table 2.B of this document.
- The fraction of PCB mixture consisting of each homolog is calculated in Table 11.B. by dividing the sum of the PCB homologs concentrations by the individual PCB homolog concentration.
- The octanol-water partition coefficient ( $K_{OW}$ ) for homolog  $i$  is shown in Table 11.C. This data is calculated as 10 to the power of the Log  $K_{OW}$  as referenced from 'Table 3. Percentage homologue composition and chronic sediment-quality benchmarks for selected U.S. commercial and environmental polychlorinated biphenyl (PCB) mixtures' in Fuchsman et al. 2006.
- The overall log  $K_{OW}$  for the PCB Mixture (Log  $K_{OW-Total PCB}$ ) is calculated in Table 11.D. as the logarithm base 10 of the inverse of the sum of the ratio of the fraction of PCB mixture consisting of homolog  $i$  ( $f_{homolog i}$ ) to the Log  $K_{OW}$  for homolog  $i$  (Log  $K_{OW-homolog i}$ ).
- PCB = polychlorinated biphenyl.
- µg/kg, dw = micrograms per kilogram, dry weight.
- µg/µg<sub>Total PCB Homologs</sub> = microgram individual PCB homolog per microgram total PCB homologs.
- $L_{water}/L_{octanol}$  = liter water per liter octanol.
- µg•L<sub>octanol</sub>/µg<sub>Total PCB Homologs</sub>•L<sub>water</sub> = microgram individual PCB homolog times liters of octanol per microgram total PCB homolog times liters of water.

Table 11. Octanol-water Partition Coefficient for the PCB Homologue Mixture in Surface Sediment, continued

Monochlorobiphenyl ( $L_{water}/L_{octanol}$ )	Dichlorobiphenyl ( $L_{water}/L_{octanol}$ )	Trichlorobiphenyl ( $L_{water}/L_{octanol}$ )	Tetrachlorobiphenyl ( $L_{water}/L_{octanol}$ )	Pentachlorobiphenyl ( $L_{water}/L_{octanol}$ )	Hexachlorobiphenyl ( $L_{water}/L_{octanol}$ )	Heptachlorobiphenyl ( $L_{water}/L_{octanol}$ )	Octachlorobiphenyl ( $L_{water}/L_{octanol}$ )	Nonachlorobiphenyl ( $L_{water}/L_{octanol}$ )	Decachlorobiphenyl ( $L_{water}/L_{octanol}$ )
4.37E+04	1.32E+05	4.17E+05	1.10E+06	3.09E+06	6.92E+06	9.55E+06	5.25E+07	1.74E+08	1.82E+08

Station	$f_{\text{homolog } i} \div K_{OW-\text{homolog } i}$											$\Sigma(f_{\text{homolog } i} \div K_{OW-\text{homolog } i})$ ( $\mu\text{g} \cdot L_{octanol} / \mu\text{g}_{Total PCB Homologs} \cdot L_{water}$ )	Log $K_{OW-Totals PCB}$ ( $L_{water}/L_{octanol}$ )
	Monochlorobiphenyl ( $\mu\text{g} \cdot L_{octanol} / \mu\text{g}_{Total PCB Homologs} \cdot L_{water}$ )	Dichlorobiphenyl ( $\mu\text{g} \cdot L_{octanol} / \mu\text{g}_{Total PCB Homologs} \cdot L_{water}$ )	Trichlorobiphenyl ( $\mu\text{g} \cdot L_{octanol} / \mu\text{g}_{Total PCB Homologs} \cdot L_{water}$ )	Tetrachlorobiphenyl ( $\mu\text{g} \cdot L_{octanol} / \mu\text{g}_{Total PCB Homologs} \cdot L_{water}$ )	Pentachlorobiphenyl ( $\mu\text{g} \cdot L_{octanol} / \mu\text{g}_{Total PCB Homologs} \cdot L_{water}$ )	Hexachlorobiphenyl ( $\mu\text{g} \cdot L_{octanol} / \mu\text{g}_{Total PCB Homologs} \cdot L_{water}$ )	Heptachlorobiphenyl ( $\mu\text{g} \cdot L_{octanol} / \mu\text{g}_{Total PCB Homologs} \cdot L_{water}$ )	Octachlorobiphenyl ( $\mu\text{g} \cdot L_{octanol} / \mu\text{g}_{Total PCB Homologs} \cdot L_{water}$ )	Nonachlorobiphenyl ( $\mu\text{g} \cdot L_{octanol} / \mu\text{g}_{Total PCB Homologs} \cdot L_{water}$ )	Decachlorobiphenyl ( $\mu\text{g} \cdot L_{octanol} / \mu\text{g}_{Total PCB Homologs} \cdot L_{water}$ )			
NA19	4.01E-09	8.86E-09	1.93E-08	9.99E-08	1.44E-07	4.75E-08	9.17E-09	2.50E-10	2.18E-11	1.48E-11	3.33E-07	6.48	
NA22	1.28E-08	2.67E-08	6.44E-08	1.13E-07	1.01E-07	4.46E-08	1.76E-08	7.56E-10	4.84E-11	4.18E-11	3.81E-07	6.42	
SW01	1.71E-08	2.67E-08	1.61E-07	2.08E-07	1.12E-07	3.48E-08	9.00E-09	4.23E-10	2.36E-11	4.61E-12	5.69E-07	6.24	
SW04	3.01E-08	3.42E-08	8.74E-08	1.57E-07	1.33E-07	3.88E-08	8.23E-09	3.74E-10	3.03E-11	5.37E-12	4.90E-07	6.31	
SW05	1.01E-08	2.45E-08	1.18E-07	2.03E-07	1.15E-07	3.62E-08	8.17E-09	5.73E-10	4.49E-11	7.96E-12	5.16E-07	6.29	
SW10	1.16E-08	8.16E-08	3.10E-07	2.35E-07	1.01E-07	2.80E-08	7.21E-09	3.89E-10	2.79E-11	9.46E-12	7.75E-07	6.11	
SW13	1.03E-08	1.49E-08	4.36E-08	1.12E-07	9.51E-08	4.85E-08	1.76E-08	8.26E-10	5.88E-11	3.00E-11	3.43E-07	6.46	
SW16	8.94E-09	1.48E-08	3.82E-08	1.04E-07	9.47E-08	4.93E-08	1.87E-08	8.05E-10	4.68E-11	2.59E-11	3.29E-07	6.48	
SW20	2.77E-09	3.54E-09	1.96E-08	8.88E-08	5.42E-08	5.23E-08	3.06E-08	1.26E-09	2.91E-11	5.56E-12	2.53E-07	6.60	
SW22	3.82E-09	7.71E-09	3.83E-08	1.06E-07	7.52E-08	4.93E-08	2.43E-08	9.96E-10	3.51E-11	1.44E-11	3.06E-07	6.51	
SW23	5.35E-09	6.40E-09	2.33E-08	8.87E-08	7.56E-08	5.16E-08	2.51E-08	1.01E-09	3.58E-11	1.18E-11	2.77E-07	6.56	
SW24	2.68E-09	6.09E-09	2.09E-08	8.24E-08	6.50E-08	5.37E-08	2.77E-08	1.05E-09	3.72E-11	1.09E-11	2.60E-07	6.59	
SW28	2.79E-09	4.47E-09	9.37E-09	3.37E-08	5.02E-08	5.93E-08	3.39E-08	1.20E-09	3.05E-11	7.12E-12	1.95E-07	6.71	



**Table 12. Summary of Studies Used to Derive a Mercury Effects Benchmark in Invertebrate Tissue**

Media	Taxa	Class	Tissue Effect Concentration (mg/kg, ww)	Tissue(s) Analyzed	Effect	End Point	Reference
Freshwater	<i>Chironomus riparius</i>	Arthropoda	40	Whole Body	Mortality	NOED	Rossaro et al. (1986)
Freshwater	<i>Carcinus maenas</i>	Crustacea	39	Digestive Tract, Exoskeleton, Gill, Other	Mortality	LD50 (Average of Tissues Analyzed)	Mount and Stephan (1967)
Freshwater	<i>Perna perna</i>	Mollusca	17	Soft Tissue	Physiological (filtration rate)	NOED	Gregory et al. (2002)
Freshwater	<i>Crepidula fornicata</i>	Mollusca	8	Whole Body	Development	LOED	Thain (1984)
Freshwater	<i>Viviparus georgianus</i>	Mollusca	6	Soft Tissue	Mortality	NOED	Tessier et al. (1996)
Saltwater	<i>Hexagenia rigida</i>	Arthropoda	2	Whole Body	Growth, Mortality	NOED	Odin et al. (1994)
Saltwater	<i>Palaemonetes pugio</i>	Crustacea	1.6	Whole Body	Behavior, Mortality	LOED, NOED	Barthalmus (1977)
Saltwater	<i>Hexagenia</i>	Arthropoda	1.1	Whole Body	Growth	NOED	Naimo et al. (2000)
Saltwater	<i>Elliptio complanata</i>	Mollusca	0.19	Whole Body	Growth	NOED	Beckvar et al. (2000)

**Notes:**

- <sup>1</sup> Data referenced from the Environmental Residue-Effects Database (ERED).
- <sup>2</sup> Concentrations of mercury in tissue residue are assumed to be total mercury.
- <sup>3</sup> Endpoints were selected from the ERED based on the following preference: LOED > % Effect Concentration (e.g., EC50, LC50, etc.) > NOED.
- <sup>4</sup> NOED = No Observed Effect Dose.
- <sup>5</sup> LD50 = Median Lethal Dose.
- <sup>6</sup> LOED = Lowest Observed Effect Dose.
- <sup>7</sup> mg/kg, ww = milligrams per kilogram, wet weight.

Table 13. Summary of Effects Benchmarks

COC	Individual Chemical, if applicable	Effects Benchmark				
		Value	UOM	Type	Basis for Effects Benchmark	Reference
Copper	--	130	µmol/g <sub>OC</sub>	ESB; Acute and Chronic Toxicity	10 and 14 day mortality to various freshwater and aquatic organisms; Chronic toxicity possible when value exceeded (Pg 3-22 of USEPA (2005a))	USEPA (2005a)
	--	3.1	µg/L	FCV	Chronic toxicity for various aquatic organisms	USEPA (2005a)
Mercury	--	0.19	mg/kg	Tissue effect benchmark	Sublethal tissue benchmark for a variety of benthic organisms (Table 12)	Table 12
HPAH	Fluoranthene	7.109	µg/L	FCV	Chronic toxicity for various aquatic organisms	USEPA (2003b)
	Pyrene	10.11				
	Benzo[a]-anthracene	2.227				
	Chrysene	2.042				
	Benzo[b]-fluoranthene	0.6774				
	Benzo[k]-fluoranthene	0.6415				
	Benzo[a]-pyrene	0.9573				
	Indeno-[1,2,3-cd]-pyrene	0.275				
	Dibenz[a,h]-anthracene	0.2825				
Benzo[ghi]-perylene	0.4391					
PCB	--	0.54	µg/L	FCV	Chronic toxicity for various aquatic organisms	Fuchsman et al. (2006)
TBT	--	0.0658	µg/L	FCV	Chronic toxicity for various aquatic organisms	USEPA (2003a)

**Notes:**

- <sup>1</sup> COC = Contaminant of Concern.
- <sup>2</sup> HPAH = High Molecular Weight Polycyclic Aromatic Hydrocarbons.
- <sup>3</sup> PCB = polychlorinated biphenyl.
- <sup>4</sup> TBT = Tributyltin.
- <sup>5</sup> UOM = Unit of Measure
- <sup>6</sup> µmol/g<sub>OC</sub> = micromoles per gram organic carbon.
- <sup>7</sup> mg/kg = milligram per kilogram.
- <sup>8</sup> µg/L = microgram per liter.
- <sup>9</sup> ESB = Equilibrium Partitioning Sediment Benchmark
- <sup>10</sup> FCV = Final Chronic Value

**Table 14. Benthic Risk for Total PCB Homologs**

Notes	1	2	3	4	5	6	7	8	9	10
Station	Measured Concentration in Surface Sediment (µg/kg <sub>sediment</sub> )	Log K <sub>OW-Total PCB</sub> (L <sub>water</sub> /L <sub>octanol</sub> )	K <sub>OC-Total PCB</sub> (L <sub>porewater</sub> /kg <sub>OC</sub> )	TOC Content (%)	Concentration in Surface Sediment on an Organic Carbon Basis (µg/kg <sub>OC</sub> )	Estimated Concentration in Porewater [EqP model approach] (µg/L)	Estimated Concentration in Porewater [Prediction equation approach] (µg/L)	Measured Concentration in Porewater (µg/L)	Effects Benchmark (µg/L)	Toxic Units
NA19	1,400	6.48	3,002,369	1.84	76,087	0.03	0.28	NR	0.54	0.05
NA22	250	6.42	2,621,879	1.65	15,152	0.01	0.07	NR		0.01
SW01	2,400	6.24	1,756,273	2.24	107,143	0.06	0.44	0.50		0.11
SW04	5,200	6.31	2,041,322	2.28	228,070	0.11	0.88	0.60		0.21
SW05	1,800	6.29	1,938,806	1.55	116,129	0.06	0.34	NR		0.11
SW10	930	6.11	1,290,485	1.21	76,860	0.06	0.20	NR		0.11
SW13	710	6.46	2,914,977	2.33	30,472	0.01	0.16	NR		0.02
SW16	610	6.48	3,036,080	2.24	27,232	0.01	0.14	NR		0.02
SW20	2,600	6.60	3,950,835	2.14	121,495	0.03	0.47	NR		0.06
SW22	1,400	6.51	3,272,133	2.46	56,911	0.02	0.28	NR		0.03
SW23	1,500	6.56	3,607,969	2.52	59,524	0.02	0.29	NR		0.03
SW24	1,500	6.59	3,853,291	1.75	85,551	0.02	0.29	0.67		0.04
SW28	2,600	6.71	5,131,261	2.52	103,311	0.02	0.47	0.29		0.04

**Notes:**

- <sup>1</sup> Measured Concentration in Surface Sediment is referenced from Table 2.B.
- <sup>2</sup> Overall logarithm of octanol-water partition coefficient (Log K<sub>OW</sub>) for the PCB Mixture (Log K<sub>OW-Total PCB</sub>) is referenced from Table 11.
- <sup>3</sup> K<sub>OW-Total PCB</sub> is calculated as 10 to the power of Log K<sub>OW-Total PCB</sub>. The Organic Carbon-Water Partition Coefficient (K<sub>OC</sub>) for nonionic organic chemicals is assumed to be equal to the K<sub>OW</sub> (Fuchsman et al., 2006; Bucheli and Gustafsson, 2001).
- <sup>4</sup> Total organic carbon (TOC) content is referenced from Table 5.B.
- <sup>5</sup> Concentration in Surface Sediment on an Organic Carbon Basis is calculated as the Measured Concentration in Surface Sediment ÷ (TOC Content ÷ 100%).
- <sup>6</sup> Estimated Concentration in Porewater is calculated using the Equilibrium Partitioning (EqP) model approach (Di Toro et al., 1992). Estimated Porewater Concentration = Concentration in Surface Sediment on an Organic Carbon Basis ÷ K<sub>OC</sub>.
- <sup>7</sup> Estimated Concentration in Porewater is calculated using the prediction equation as referenced from 'Table 5-2 Relationships between porewater and sediment' (Exponent, 2003). For PCB homologs, the prediction equation is  $pw = [2.65 + 0.374 \times (\text{Measured Concentration of Contaminant in the Surface Sediment})^{1/2}]^2$ . Porewater units from this equation are provided in ng/L, this value is converted to µg/L.
- <sup>8</sup> Measured Concentration in Porewater is referenced from Table 9.
- <sup>9</sup> Effects Benchmark and reference are provided in Table 13.
- <sup>10</sup> Toxic Units are calculated as the Porewater Concentration divided by the Effects Benchmark. If the Measured Concentration in Porewater is reported, then the lower value of the Measured Concentration in Porewater or the Estimated Concentration in Porewater [EqP model approach] is used to calculate the Toxic Units. Otherwise, the lower value of the Estimated Concentration in Porewater [EqP model approach] or Estimated Concentration in Porewater [Prediction equation approach] is used to calculate the Toxic Units.
- <sup>11</sup> Values indicating risk potential (Toxic Unit > 1) are shown in shaded cells with bold text.
- <sup>12</sup> PCB = polychlorinated biphenyl.
- <sup>13</sup> µg/kg = micrograms per kilogram.
- <sup>14</sup> L<sub>water</sub>/L<sub>octanol</sub> = liter water per liter octanol.
- <sup>15</sup> µg/L = micrograms per liter.



Table 15. Benthic Risk for HPAHs, continued

Notes	4	5	5	5	5	5	5	5	5	5	5	6	6	6	6	6	6	6	6	6	6	7	7	7	7	7	7	7	7	7	7	7	7	
Station	TOC Content (%)	Concentration in Surface Sediment on an Organic Carbon Basis (µg/kg <sub>OC</sub> )										Estimated Concentration in Porewater (µg/L)										Measured Concentration in Porewater (µg/L)												
		Fluoranthene	Pyrene	Benz[a]-anthracene	Chrysene	Benzo[b]-fluoranthene	Benzo[k]-fluoranthene	Benzo[a]-pyrene	Indeno-[1,2,3-cd]-pyrene	Dibenz[a,h]-anthracene	Benzo[ghi]-perylene	Fluoranthene	Pyrene	Benz[a]-anthracene	Chrysene	Benzo[b]-fluoranthene	Benzo[k]-fluoranthene	Benzo[a]-pyrene	Indeno-[1,2,3-cd]-pyrene	Dibenz[a,h]-anthracene	Benzo[ghi]-perylene	Fluoranthene	Pyrene	Benz[a]-anthracene	Chrysene	Benzo[b]-fluoranthene	Benzo[k]-fluoranthene	Benzo[a]-pyrene	Indeno-[1,2,3-cd]-pyrene	Dibenz[a,h]-anthracene	Benzo[ghi]-perylene			
NA19	1.84	10,326	10,870	9,239	17,935	27,174	19,022	25,000	22,283	4,130	17,935	0.10	0.16	0.02	0.04	0.02	0.01	0.02	0.01	0.00	0.01	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	
NA22	1.65	12,727	35,758	10,303	21,212	36,970	28,485	32,727	20,000	4,242	17,576	0.13	0.52	0.03	0.05	0.03	0.02	0.03	0.00	0.00	0.01	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
SW01	2.24	35,268	49,107	25,000	41,071	49,107	35,268	49,107	31,250	6,250	24,107	0.35	0.71	0.07	0.10	0.03	0.02	0.05	0.01	0.00	0.01	0.04	0.13	0.02	0.03	0.09	0.07	0.07	0.04	0.01	0.03			
SW04	2.28	78,947	74,561	57,018	96,491	87,719	61,404	78,947	43,860	10,088	31,140	0.79	1.08	0.15	0.23	0.06	0.04	0.08	0.01	0.00	0.01	0.07	0.11	0.04	0.06	0.10	0.06	0.07	0.05	0.01	0.03			
SW05	1.55	148,387	141,935	58,710	96,774	96,774	77,419	96,774	50,968	12,903	37,419	1.49	2.06	0.16	0.23	0.07	0.05	0.10	0.01	0.00	0.02	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	
SW10	1.21	330,579	289,256	99,174	123,967	132,231	99,174	132,231	65,289	16,529	48,760	3.32	4.19	0.26	0.30	0.09	0.06	0.13	0.02	0.00	0.02	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	
SW13	2.33	94,421	60,086	36,910	81,545	64,378	55,794	60,086	36,910	7,296	27,468	0.95	0.87	0.10	0.20	0.04	0.04	0.06	0.01	0.00	0.01	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	
SW16	2.24	11,607	32,143	9,375	17,857	49,107	35,268	44,643	26,786	5,804	22,321	0.12	0.47	0.02	0.04	0.03	0.02	0.04	0.01	0.00	0.01	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	
SW20	2.14	43,458	56,075	35,514	84,112	70,093	56,075	65,421	45,327	9,346	35,981	0.44	0.81	0.09	0.20	0.05	0.04	0.06	0.01	0.00	0.01	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	
SW22	2.46	36,992	44,715	36,179	77,236	73,171	52,846	69,106	44,715	9,350	33,740	0.37	0.65	0.10	0.19	0.05	0.03	0.07	0.01	0.00	0.01	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	
SW23	2.52	38,095	39,683	33,730	71,429	59,524	47,619	59,524	39,683	8,730	32,540	0.38	0.57	0.09	0.17	0.04	0.03	0.06	0.01	0.00	0.01	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	
SW24	1.75	307,985	370,722	256,654	450,570	410,646	365,019	456,274	188,213	53,042	131,179	3.09	5.37	0.68	1.09	0.28	0.24	0.45	0.05	0.01	0.05	0.07	0.18	0.04	0.07	0.27	0.20	0.25	0.09	0.01	0.06			
SW28	2.52	55,629	59,603	55,629	127,152	103,311	71,523	103,311	59,603	11,126	36,556	0.56	0.86	0.15	0.31	0.07	0.05	0.10	0.01	0.00	0.01	0.04	0.09	0.01	0.01	0.05	0.03	0.03	0.01	0.01	0.01			

Table 15. Benthic Risk for HPAHs, continued

Notes	8	8	8	8	8	8	8	8	8	8	9	9	9	9	9	9	9	9	9	9	10
Station	Effects Benchmark (µg/L)										HPAH, Toxic Units										Total HPAH Toxic Units
	Fluoranthene	Pyrene	Benz[a]-anthracene	Chrysene	Benzo[b]-fluoranthene	Benzo[k]-fluoranthene	Benzo[a]-pyrene	Indeno-[1,2,3-cd]-pyrene	Dibenz[a,h]-anthracene	Benzo[ghi]-perylene	Fluoranthene	Pyrene	Benz[a]-anthracene	Chrysene	Benzo[b]-fluoranthene	Benzo[k]-fluoranthene	Benzo[a]-pyrene	Indeno-[1,2,3-cd]-pyrene	Dibenz[a,h]-anthracene	Benzo[ghi]-perylene	
NA19											0.01	0.02	0.01	0.02	0.03	0.02	0.03	0.02	0.00	0.02	0.18
NA22											0.02	0.05	0.01	0.03	0.04	0.03	0.03	0.02	0.00	0.02	0.25
SW01											0.01	0.01	0.01	0.01	0.05	0.04	0.05	0.03	0.01	0.02	0.23
SW04											0.01	0.01	0.02	0.03	0.09	0.06	0.07	0.04	0.01	0.03	0.37
SW05											0.21	0.20	0.07	0.11	0.10	0.08	0.10	0.05	0.01	0.03	0.97
SW10											0.47	0.41	0.12	0.15	0.14	0.10	0.14	0.06	0.01	0.04	1.6
SW13	7.109	10.11	2.227	2.042	0.6774	0.6415	0.9573	0.275	0.2825	0.4391	0.13	0.09	0.04	0.10	0.07	0.06	0.06	0.03	0.01	0.03	0.61
SW16											0.02	0.05	0.01	0.02	0.05	0.04	0.05	0.02	0.01	0.02	0.28
SW20											0.06	0.08	0.04	0.10	0.07	0.06	0.07	0.04	0.01	0.03	0.56
SW22											0.05	0.06	0.04	0.09	0.07	0.05	0.07	0.04	0.01	0.03	0.53
SW23											0.05	0.06	0.04	0.08	0.06	0.05	0.06	0.04	0.01	0.03	0.48
SW24											0.01	0.02	0.02	0.03	0.40	0.31	0.26	0.17	0.04	0.12	1.4
SW28											0.01	0.01	0.00	0.00	0.08	0.05	0.03	0.04	0.01	0.02	0.25

Table 16. Benthic Risk for Copper

Notes	1	2	2	3	4	5	6	7	8	9	10	11	12	6	13	14
Station	TOC Content (%)	AVS Concentration in Surface Sediment (mg/kg)	SEM for Copper Concentration in Surface Sediment (mg/kg)	AVS Concentration in Surface Sediment (mmol/kg)	SEM for Copper Concentration in Surface Sediment (mmol/kg)	OC Normalized Excess SEM Concentration in Surface Sediment (mmol/kg, OC)	Effects Benchmark [Sediment Approach] (mmol/kg, OC)	Toxic Units [SEM-AVS Sediment Approach]	Measured Copper Concentration in Surface Sediment (mg/kg)	$K_d$ (L-porewater/kg sediment)	Estimated Concentration in Porewater [EqP model approach] (µg/L)	Estimated Concentration in Porewater [Prediction equation approach] (µg/L)	Measured Concentration in Porewater (µg/L)	Effects Benchmark [Porewater Approach] (µg/L, OC)	Toxic Units [Porewater Approach]	Toxic Units
NA19	1.84	130	280	4.1	4.4	19	130	0.15	270	12,589	21	20	NR	3.1	6.5	0.15
NA22	1.65	180	120	5.6	1.9	-226		0.00	150		12	16	NR		3.8	0.00
SW01	2.24	57	350	1.8	5.5	167		1.3	560		44	30	17		5.5	1.3
SW04	2.28	35	1100	1.1	17.3	711		5.5	1,500		119	62	55		18	5.5
SW05	1.55	190	200	5.9	3.1	-179		0.00	230		18	19	NR		5.9	0.00
SW10	1.21	610	140	19.0	2.2	-1390		0.00	160		13	16	NR		4.1	0.00
SW13	2.33	220	1200	6.9	18.9	516		4.0	800		64	38	NR		12	4.0
SW16	2.24	280	500	8.7	7.9	-39		0.00	430		34	26	NR		8.3	0.00
SW20	2.14	370	260	11.5	4.1	-348		0.00	290		23	21	NR		6.7	0.00
SW22	2.46	60	300	1.9	4.7	116		0.89	260		21	20	NR		6.4	0.89
SW23	2.52	160	270	5.0	4.2	-29		0.00	280		22	21	NR		6.6	0.00
SW24	1.753	60	240	1.9	3.8	109		0.84	300		24	21	25		7.7	0.84
SW28	2.517	450	270	14.0	4.2	-389		0.00	265		21	20	19		6.1	0.00

- Notes:**
- Total Organic Carbon (TOC) content is referenced from Table 5.
  - Concentrations of Acid Volatile Sulfide (AVS) and Simultaneously Extracted Metal (SEM) for Copper in Surface Sediment are referenced from Table 4.
  - Molar mass of sulfur is 32.065 mg/mmol.
  - Molar mass of copper is 63.546 mg/mmol.
  - Organic Carbon (OC) Normalized Excess SEM for Copper Concentration in Surface Sediment is calculated as (SEM - AVS) ÷ TOC. The SEM-AVS ÷ TOC methodology was referenced from USEPA (2005a).
  - Effects Benchmark and reference are provided in Table 13.
  - Toxic Units are calculated as the OC Normalized Excess SEM for Copper Concentration (mmol/kg) divided by the Effects Benchmark. If the OC Normalized Excess SEM for Copper Concentration is negative, then the Toxic Unit is 0.
  - Measured Copper Concentration in Surface Sediment is referenced from Table 6.
  - The sediment-porewater partition coefficient ( $K_d$ ) was calculated as 10 to the power Log  $K_d$  as referenced from 'Table 4. Metal partition coefficients (log  $K_d$ ) in kg/L for sediment/porewater' (USEPA, 2005b).
  - Estimated Concentration in Porewater is calculated using the copper sediment-porewater partition coefficient ( $K_d$ ) which is defined as the ratio of sorbed concentration to the dissolved concentration at equilibrium.  $K_d = \text{Sorbed Concentration} / \text{Dissolved Concentration}$ . Therefore, the Dissolved Concentration = Sorbed Concentration /  $K_d$ . Units are converted from mg/L to µg/L.
  - Estimated Concentration in Porewater is calculated using the prediction equation as referenced from 'Table 5-2 Relationships between porewater and sediment' (Exponent, 2003). For Copper, the prediction equation is  $pw = 10.9 + 0.0343 \times (\text{Measured Concentration of Contaminant in the Surface Sediment})$ .
  - Measured Concentration in Porewater is referenced from Table 7.
  - Toxic Units are calculated as the Porewater Concentration divided by the Effects Benchmark. If the Measured Concentration in Porewater is reported, then the lower value of the Measured Concentration in Porewater or the Estimated Concentration in Porewater [EqP model approach] is used to calculate the Toxic Units. Otherwise, the lower value of the Estimated Concentration in Porewater [EqP model approach] or Estimated Concentration in Porewater [Prediction equation approach] is used to calculate the Toxic Units.
  - Toxic Units are the lesser value of the Toxic Units [SEM-AVS Sediment approach] and Toxic Units [Porewater approach].
  - Values indicating risk potential (Toxic Unit > 1) are shown in shaded cells with bold text.
  - mg/kg = milligrams per kilogram.
  - mmol/kg = millimoles per kilogram.

**Table 17. Benthic Risk for TBT**

Notes	1	2	3	4	5	6	7	8	9	10
Station	Measured Concentration in Surface Sediment (µg/kg)	Log K <sub>ow</sub> (L <sub>water</sub> /L <sub>octanol</sub> )	K <sub>oc</sub> (L <sub>porewater</sub> /kg <sub>oc</sub> )	TOC Content (kg <sub>oc</sub> /kg <sub>sediment</sub> )	Concentration in Surface Sediment on an Organic Carbon Basis (µg/kg <sub>oc</sub> )	Estimated Concentration in Porewater [EqP model approach] (µg/L)	Estimated Concentration in Porewater [prediction equation approach] (µg/L)	Measured Concentration in Porewater (µg/L)	Effects Benchmark (µg/L)	Toxic Units
NA19	570	4.4	21,158	1.84	30,978	1.46	0.10	NR	0.0658	<b>1.6</b>
NA22	120			1.65	7,273	0.34	0.03	NR		0.52
SW01	450			2.24	20,089	0.95	0.09	0.04		0.56
SW04	3,250			2.28	142,544	6.74	0.46	0.55		<b>8.4</b>
SW05	170			1.55	10,968	0.52	0.04	NR		0.65
SW10	250			1.21	20,661	0.98	0.06	NR		0.85
SW13	790			2.33	33,906	1.60	0.14	NR		<b>2.1</b>
SW16	1,100			2.24	49,107	2.32	0.18	NR		<b>2.7</b>
SW20	130			2.14	6,075	0.29	0.04	NR		0.55
SW22	190			2.46	7,724	0.37	0.05	NR		0.70
SW23	210			2.52	8,333	0.39	0.05	NR		0.75
SW24	165			1.75	9,411	0.44	0.04	0.07		<b>1.1</b>
SW28	150			2.52	5,960	0.28	0.04	0.02		0.24

**Notes:**

- <sup>1</sup> Measured Concentration in Surface Sediment is referenced from Table 6.
- <sup>2</sup> Logarithm of the octanol-water partition coefficient (K<sub>ow</sub>) is referenced from Meador (2011)
- <sup>3</sup> The Organic Carbon-Water Partition Coefficient (K<sub>oc</sub>) (L/kg<sub>oc</sub>) is calculated as 10 to the power of Log K<sub>oc</sub>. Log K<sub>oc</sub> is calculated using the following equation from Di Toro (1985)  $\log_{10}K_{oc} = 0.00028 + 0.983 \log_{10}K_{ow}$  (USEPA, 2003b).
- <sup>4</sup> Total organic carbon (TOC) content is referenced from Table 5.
- <sup>5</sup> Concentration in Surface Sediment on an Organic Carbon Basis is calculated as the Measured Concentration in Surface Sediment ÷ (TOC Content ÷ 100%).
- <sup>6</sup> Estimated Concentration in Porewater is calculated using the Equilibrium Partitioning (EqP) model approach (Di Toro et al., 1992). Estimated Porewater Concentration = Concentration in Surface Sediment on an Organic Carbon Basis ÷ K<sub>oc</sub>.
- <sup>7</sup> Estimated Concentration in Porewater is calculated using the prediction equation as referenced from 'Table 5-2 Relationships between porewater and sediment' (Exponent, 2003). For TBT, the prediction equation is  $pw = [0.0676 + 0.0107 \times (\text{Measured Concentration of Contaminant in the Surface Sediment})^{1/2}]^2$ .
- <sup>8</sup> Measured Concentration in Porewater is referenced from Table 7.
- <sup>9</sup> Effects Benchmark and reference are provided in Table 13.
- <sup>10</sup> Toxic Units are calculated as the Porewater Concentration divided by the Effects Benchmark. If the Measured Concentration in Porewater is reported, then the lower value of the Measured Concentration in Porewater or the Estimated Concentration in Porewater [EqP model approach] is used to calculate the Toxic Units. Otherwise, the lower value of the Estimated Concentration in Porewater [EqP model approach] or Estimated Concentration in Porewater [Prediction equation approach] is used to calculate the Toxic Units.
- <sup>11</sup> Values indicating risk potential (Toxic Unit > 1) are shown in shaded cells with bold text.
- <sup>12</sup> NR = Not Reported
- <sup>13</sup> TBT = Tributyltin
- <sup>14</sup> µg/kg = micrograms per kilogram.
- <sup>15</sup> L<sub>water</sub>/L<sub>octanol</sub> = liter water per liter octanol.
- <sup>16</sup> L<sub>water</sub>/kg<sub>oc</sub> = liter water per kilogram organic carbon.
- <sup>17</sup> kg<sub>oc</sub>/kg<sub>sediment</sub> = kilograms of organic carbon per kilograms of sediment.
- <sup>18</sup> µg/kg<sub>oc</sub> = micrograms per kilogram of organic carbon.
- <sup>19</sup> µg/L = micrograms per liter.



**Table 18. Benthic Risk for Mercury**

Notes	1	2	3	4	5	6
Station	Measured Concentration in Surface Sediment (mg/kg, dw)	BAF (kg <sub>sediment, dw</sub> /kg <sub>tissue, ww</sub> )	Estimated Concentration in Invertebrate Tissue (mg/kg, ww)	Measured Concentration in Invertebrate Tissue (mg/kg, ww)	Effects Benchmark (mg/kg, ww)	Toxic Units
NA19	0.8	0.026	0.02	0.024	0.19	0.13
NA22	0.4		0.01	NR		0.05
SW01	1.5		0.04	NR		0.20
SW04	1.8		0.05	NR		0.24
SW05	1.0		0.03	NR		0.13
SW10	0.6		0.02	NR		0.08
SW13	0.9		0.02	NR		0.12
SW16	1.0		0.03	NR		0.14
SW20	1.0		0.03	NR		0.14
SW22	1.1		0.03	NR		0.15
SW23	1.0		0.03	NR		0.14
SW24	1.9		0.05	NR		0.26
SW28	0.9		0.02	NR		0.12

**Notes:**

- <sup>1</sup> Concentration in Surface Sediment is referenced from Table 6.
- <sup>2</sup> Bioaccumulation Factor (BAF) is calculated in Table 10.
- <sup>3</sup> Estimated Concentration in Invertebrate Tissue is calculated as the Surface Sediment Concentration multiplied by the BAF.
- <sup>4</sup> Measured concentration in mussel (soft tissue) is referenced from 'Table E-1. Lipid, solids, metal, and butyltin results for tissue samples' in Exponent, 2003. Composite samples of benthic mussel tissues with shell removed (*Musculista senhousiei*) were collected at two stations inside each shipyard, NASSCO and Southwest Marine.
- <sup>5</sup> Effects Benchmark and reference are provided in Table 13.
- <sup>6</sup> Toxic Units are calculated as the Measured Concentration in Invertebrate Tissue divided by the Effects Benchmark. If the Measured Concentration in Invertebrate Tissue is not reported, then the Estimated Concentration in Invertebrate Tissue is used.
- <sup>7</sup> Values indicating risk potential (Toxic Unit > 1) are shown in shaded cells with bold text.
- <sup>8</sup> NR = Not Reported
- <sup>9</sup> mg/kg, ww = milligrams per kilogram, wet weight.
- <sup>10</sup> mg/kg, dw = milligrams per kilogram, dry weight.
- <sup>11</sup> kg<sub>sediment, dw</sub>/kg<sub>tissue, ww</sub> = kilogram of sediment, dry weight per kilogram of tissue, wet weight.

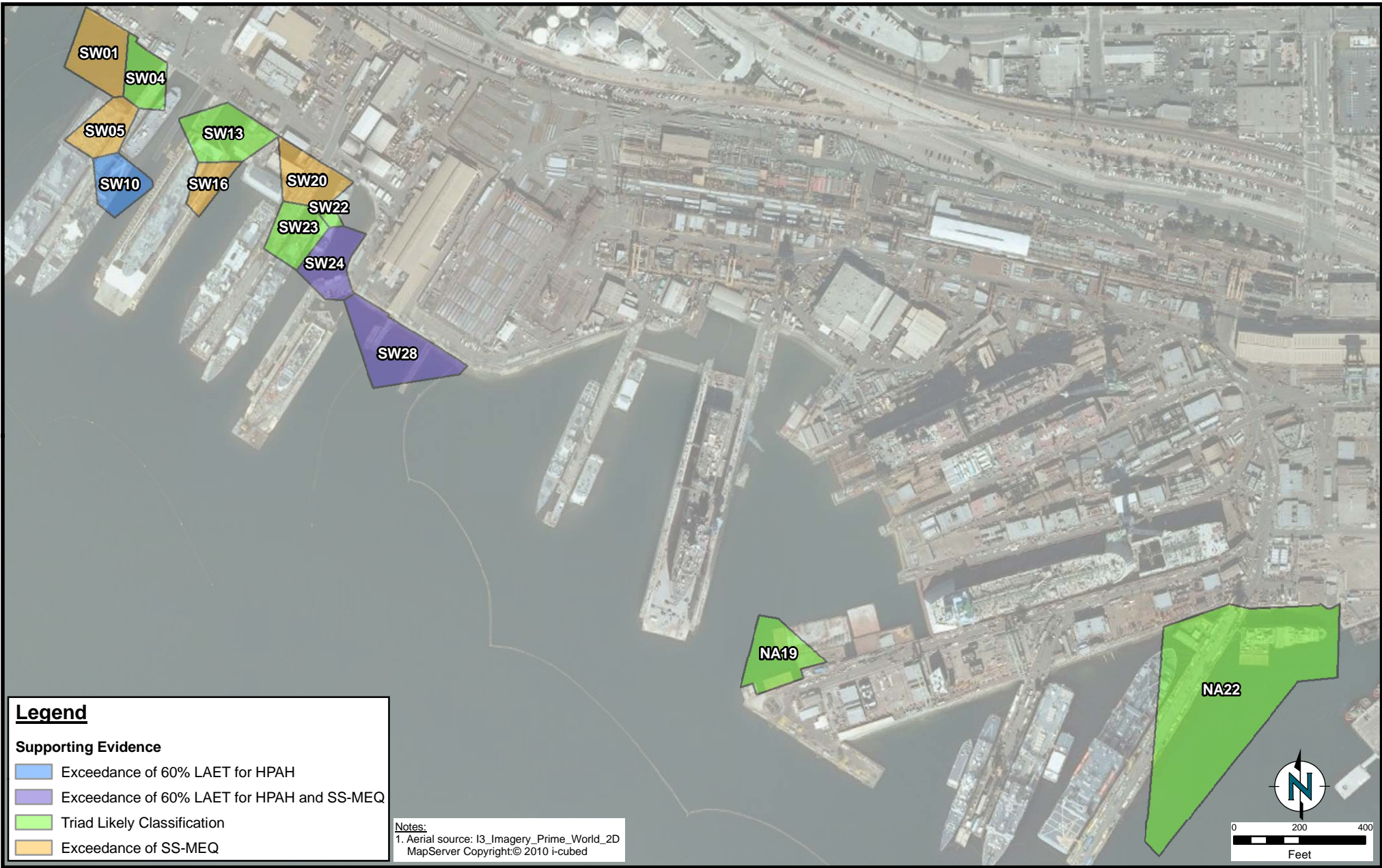
**Table 19. Summary of Benthic Risk for the Five Primary Site Chemicals of Concern**

Notes	1	2	3	4	5	6
Station	Toxic Unit for PCB Homolog	Toxic Unit for Total HPAH	Toxic Unit for Copper	Toxic Unit for TBT	Toxic Unit for Mercury	Chemicals Posing Risk
NA19	0.05	0.18	0.15	<b>1.6</b>	0.13	TBT
NA22	0.01	0.25	0.00	0.52	0.05	
SW01	0.11	0.23	<b>1.3</b>	0.56	0.20	Copper
SW04	0.21	0.37	<b>5.5</b>	<b>8.4</b>	0.24	Copper, TBT
SW05	0.11	0.97	0.00	0.65	0.13	
SW10	0.11	<b>1.6</b>	0.00	0.85	0.08	HPAH
SW13	0.02	0.61	<b>4.0</b>	<b>2.1</b>	0.12	Copper, TBT
SW16	0.02	0.28	0.00	<b>2.7</b>	0.14	TBT
SW20	0.06	0.56	0.00	0.55	0.14	
SW22	0.03	0.53	0.89	0.70	0.15	
SW23	0.03	0.48	0.00	0.75	0.14	
SW24	0.04	<b>1.4</b>	0.84	<b>1.1</b>	0.26	HPAH, TBT
SW28	0.04	0.25	0.00	0.24	0.12	

**Notes:**

- <sup>1</sup> The Toxic Unit for PCB Homologs is referenced from Table 14.
- <sup>2</sup> The Toxic Unit for Total HPAH is referenced from Table 15.
- <sup>3</sup> The Toxic Unit for Copper is referenced from Table 16.
- <sup>4</sup> The Toxic Unit for TBT is referenced from Table 17.
- <sup>5</sup> The Toxic Unit for Mercury is referenced from Table 18.
- <sup>6</sup> Toxic units greater than 1 pose potential risk to the benthic community.
- <sup>7</sup> HPAH = High Molecular Weight Polycyclic Aromatic Hydrocarbons.
- <sup>8</sup> PCB = polychlorinated biphenyl.
- <sup>9</sup> TBT = Tributyltin.

## Figures

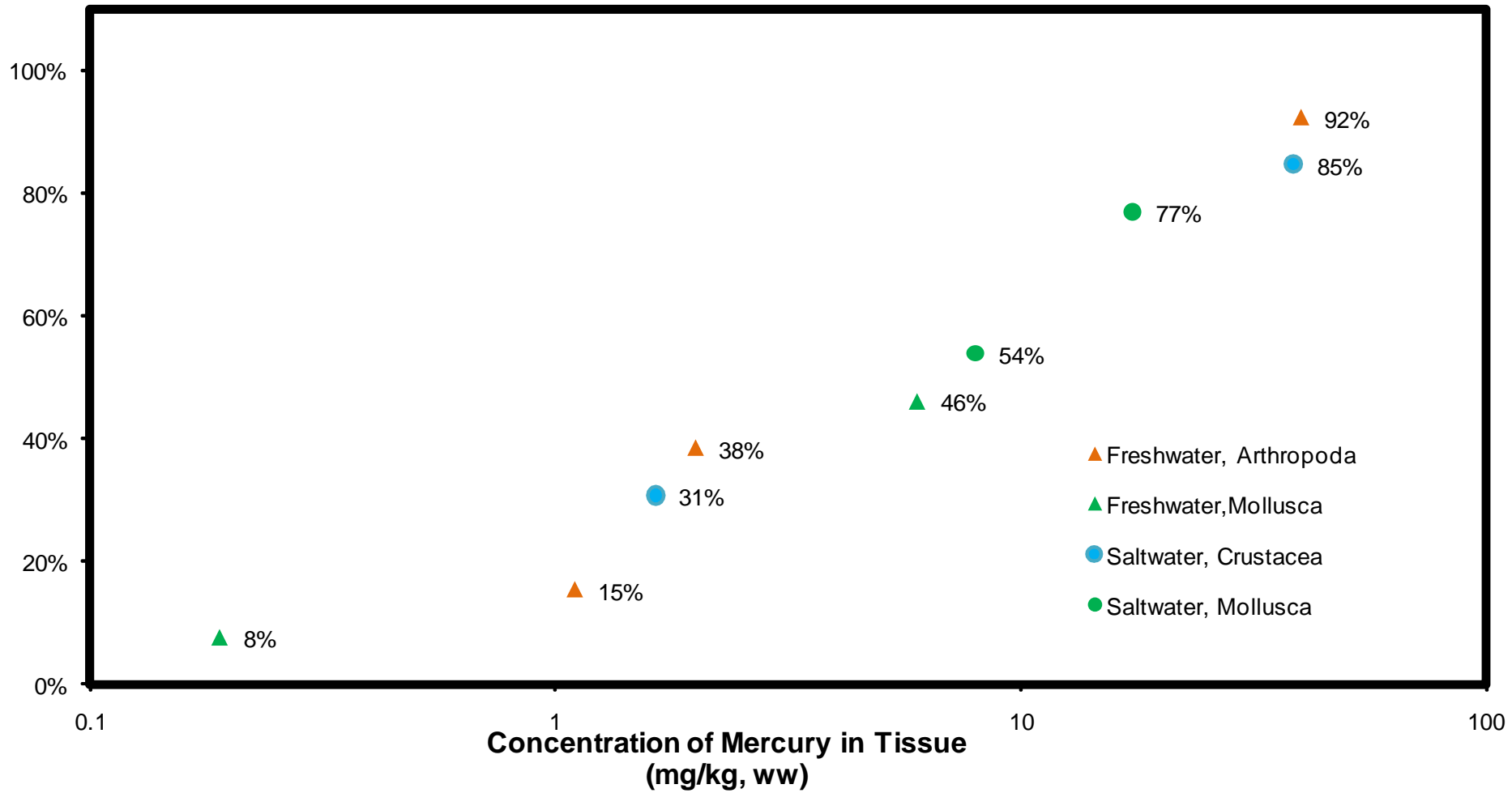


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# Stations with Aquatic Life Beneficial Use Impairment

San Diego Shipyard Sediment Site  
 San Diego, California

% Species Affected



## Mercury Tissue Residue Effect Data for Benthic Invertebrates

San Diego Shipyard Sediment Site  
San Diego, California

Figure  
2



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**Appendix A**  
**Curriculum Vitae for Jason Conder**

## Jason M. Conder, PhD

### EDUCATION

- 2004 PhD, Environmental Science, University of North Texas (UNT)
- 2000 MS, Zoology, Oklahoma State University (OSU)
- 1997 BS, Wildlife and Fisheries Ecology, OSU

### EXPERIENCE

Dr. Jason M. Conder is a Manager in the Ecological/Sediment Practice at ENVIRON. He has over 10 years research and consulting experience in environmental toxicology, ecological risk assessment, bioaccumulation and bioavailability of environmental contaminants, environmental chemistry, environmental monitoring technology, wildlife ecology and management, plant and animal taxonomy, and statistics. Project-related experience includes the assessment of ecotoxicity, bioaccumulation, and bioavailability of organic compounds and metals to aquatic and terrestrial invertebrates, plants, mammals, reptiles, and fish exposed to contaminated soils, sediments, and water. A key focus of his expertise is contaminant bioavailability. Jason has extensive experience with the measurement and interpretation of environmental contaminants in soil, sediment, water, and biological tissues, including innovative methods to predict contaminant bioavailability and toxicity.

Jason has published over 20 peer-reviewed articles in the primary scientific literature in environmental toxicology and chemistry, including several book chapters on contaminant bioavailability and sediment quality assessment. He serves as a peer reviewer for scientific journals, including: Environmental Toxicology and Chemistry, Integrated Environmental Assessment and Management, Chemosphere, Archives of Environmental Contamination and Toxicology, and Journal of Soils and Sediments.

Since joining ENVIRON in 2004, Jason has led ecological risk assessments, ecological/biological investigations, ecotoxicological studies, environmental fate and transport studies, and human health risk assessments. Representative experience includes:

- Fish Bioaccumulation Assessment, Metropolitan Council, Upper Mississippi River, MN. Evaluated bioaccumulation of PFOS in benthic and pelagic fish from water column and sediment PFOS sources. Investigated chemical fate and source issues relevant to exposure of fish to PFOS.
- Ecological Risk Assessment, Private Client, Augusta Bay, Sicily. Prepared an Ecological Risk Assessment and Sediment Quality Triad Evaluation for an industrial pier impacted with a variety of organics and metals, including mercury, methylmercury, and PAHs. Managed a team of 3 ecotoxicologists in providing a full assessment using various lines of evidence, including habitat information and chemical measurements in sediment, sediment porewater, fish, mussels, and benthic invertebrates. Key components of the assessment included food chain and bioavailability modeling and risk assessment to evaluate risks to invertebrates, fish, and piscivorous birds. Geospatial modeling was also conducted to identify areas of Augusta Bay that are associated with potentially-elevated chemical exposures.
- Contaminated Sediment Risk and Chemical Fate and Transport Evaluation, San Diego Gas & Electric (Subsidiary of Sempra Energy), San Diego Bay, CA. Evaluation of human health and ecological risks, sediment cleanup values, remedial strategies, sediment hydrodynamics, chemical fate and transport, remedial cost allocation, and chemical sources and uses in San Diego Bay. Served as project manager and technical advisor in proceedings with the California Regional Water Quality Control Board (CRWQCB) and other parties named in the CRWQCB's Cleanup and Abatement Order.



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- Monitored Natural Recovery (MNR) Guidance, United States Department of Defense (DoD). Technical advisor on a resource document used to guide DoD remedial project managers on the evaluation and application of MNR for contaminated sediment.
- Evaluation of Water Quality Impacts from Terrestrial Burn Dump, Private Client, San Francisco Bay, CA. With hydrogeologists, evaluated the ecological and human risks associated with the hypothetical transport of metals and organic chemicals to San Francisco Bay via ground water flow from a former burn dump site located 0.25 miles upland of the Bay. With considerations of appropriate aquatic life screening values and sediment geochemistry conditions, the evaluation demonstrated insignificant risk associated with the site.
- Quantico Bay Thin-Layer Cap Demonstration Project, United States Department of Defense (DoD). Led evaluation of a Thin-Layer Cap remediation project for 14-acres of sediments impacted with chlorinated pesticides (DDT, DDD, and DDE). The 5-year study is evaluating a variety of endpoints involving chemical fate and transport, chemical bioavailability measurements via *in situ* organism deployment and SPME measurements, cap physical stability, and degree of ecological risk reduction. Responsibilities included project management, coordination of field work, and interpretation and presentation of results.
- Ecotoxicological Data Review, The Dow Chemical Company, Saginaw River and Bay Watershed, MI. Review and synthesis of 30+ years of environmental data to support the avian and aquatic ecological risk assessment of dioxins and furans present in the Tittabawassee River, Saginaw River, and Saginaw Bay.
- Contaminated Sediment Management Decision-making Framework, The Dow Chemical Company. Led the development of a decision-making framework for evaluating the cause-effect relationships between chemically-impacted sediments and 16 different Beneficial Use Impairments identified by the State of Michigan. Using a tiered approach, frameworks begin with simple and resource-efficient screening steps using sediment quality guidelines and ecological benchmarks, then proceeds to considerations of more site-specific factors and determinations of probable linkages between sediments and specific Beneficial Use Impairments. Higher tiers in the frameworks utilize more advanced, but scientifically rigorous and agency-accepted approaches utilizing tools such as chemical fate and transport modeling, risk assessment, and Sediment Quality Triad, complete with decision rules for the interpretation of results with respect to resource impairment. The frameworks place screening and investigative tools in the proper context and facilitate a more efficient characterization of natural resources suspected to be affected by chemically-impacted sediment.
- Ecological Risk Assessment, Honeywell, NY/NJ Estuary System, Jersey City, NJ. Avian and aquatic ecological risk assessment of 66-acre area offshore of a former chromium ore processing facility. In addition to evaluation of chemicals in sediment, pore water, and surface water and wildlife species and habitat at the Site, responsibilities included TrophicTrace modeling to predict chemical bioaccumulation in avian and human food chains and application of the Sediment Quality Triad (SQT), a line-of-evidence approach that integrates chemistry data, laboratory toxicity results, and benthic community surveys to understand ecological risk. Using the SQT with equilibrium partitioning modeling to quantify risks, revealed that benthic community impacts and sediment toxicity were associated with widespread background PAH contamination in the local estuary, not site-related chromium releases. Key work also included evaluation of the effectiveness and risks associated with application of 11 sediment remedial alternatives, highlighting the ability of cost-effective remedies to reduce risk to ecological and human receptors.

- Sediment Monitoring Guidance and Web Portal, US Navy Space and Naval Warfare Systems Center, San Diego, CA. Prepared a guidance document and online web portal/database (<http://www.ISRAP.org>) of monitoring needs and tools associated with sediment remediation (dredging, capping, and monitored natural recovery). The guidance and online web portal assists Navy remedial project managers in developing efficient and effective monitoring plans and includes a decision-making framework to aid in selecting effective monitoring tools to assess all phases of remediation, including short-term monitoring (construction and remedial design performance) and long-term monitoring (ecological and human health risk).
- Landscape-level Ecological Risk Assessment, ICF Consulting/US Department of Energy, Bakersfield, CA. Developed a unique landscape-level approach for performing a California Department of Toxic Substances Control (DTSC) Part B Scoping Ecological Risk Assessment at a 75-square mile petroleum reserve located in southern California. The novel approach used landscape ecology and population indices to discern potential effects of active and historical petroleum exploration and production activities on the habitat and populations of endangered species and other sensitive receptors. The first step in this assessment included the site-wide investigation of the spatial co-occurrence of soil contamination and ecological receptors, as predicted by landscape-level models integrating historical ecological monitoring data, topography, and soil type.
- Ecological Risk Assessment, Private Client, CA. Conducted a DTSC Part B Scoping Ecological Risk Assessment for a former 996-acre munitions, explosives, and solid rocket fuel manufacturing facility located in southern California. Project responsibilities included the compilation of generic ecological risk-based soil screening benchmarks, preparation of a technical brief on the ecotoxicity of perchlorate, and development of a site-specific ecological risk-based soil screening level for perchlorate.
- Ecological Risk Assessment, Private Client, CA. Prepared a DTSC Part B Scoping and Phase I Predictive Ecological Risk Assessment for a 429-acre site in southern California at which explosives, solid rocket motor fuel, cryogenics, petroleum hydrocarbons, hypergolic fuels, and solvents were used. Project responsibilities have included the compilation of generic ecological risk-based soil screening benchmarks, field inspection of the Site, interpretation of biological survey information for development of the conceptual site model, food chain modeling to predict chemical bioaccumulation, and ecological risk calculations, including estimation of inhalation risks to burrowing mammals and development of toxicity reference values. Through interpretation of historical site use and the spatial pattern of chemical impacts and projected future land uses, narrowed the focus of the assessment to an undeveloped riparian area comprising approximately 5-10% of the site, enabling a more efficient and realistic approach to characterizing long-term ecological risk.
- Human Health Risk Assessment for Perchlorate Associated with Homegrown Produce, Private Client, CA. Designed and managed a laboratory plant-uptake study to determine bioconcentration factors for perchlorate accumulation by garden crops from perchlorate-impacted soils at a site in southern California. Results from the three-species study were used to generate site-specific, risk-based perchlorate concentrations associated with the consumption of homegrown garden produce by future residents. Responsibilities included experimental design and management, collection of site soils, and analysis and interpretation of data. Risk-based concentrations estimated with site-specific data developed in this study were approximately 100-fold higher than concentrations estimated using data from previous studies, which were shown to be unrealistic and overly conservative.
- Food-chain Modeling of Perfluorinated Compounds, E.I. du Pont de Nemours and Company (DuPont), Canadian Arctic. With a multi-disciplinary team of environmental chemists, engineers,

## Jason M. Conder, PhD

and risk assessors, assessed of the global fate and transport of perfluorinated carboxylic acids (PFCAs) to the Canadian Arctic. As lead technical advisor in ecotoxicology, responsibilities included development of a 5-tier food chain bioaccumulation model. The model integrated biological receptor life history and behavior, toxicokinetics of PFCAs, and environmental fate and transport processes in the Arctic Ocean to predict concentrations of PFCAs in polar bear liver tissue. Key challenges of the project included developing a model that did not rely on octanol-water partition coefficients (KOW values). Model development included Monte Carlo analysis to account for uncertainty and variability associated with model parameters and predictions.

- Critical Review of the Bioaccumulative Potential of Perfluorinated Compounds, E.I. du Pont de Nemours and Company (DuPont). Performed a survey of environmental monitoring and laboratory data on the bioaccumulation, bioconcentration, and biomagnification of perfluorinated carboxylic acids (PFCAs) and perfluorinated sulfonates (PFASs). Results were synthesized in a scientific manuscript submitted to a peer-reviewed scientific journal (Environmental Science & Technology) that summarized the bioaccumulative potential of these compounds according to guidance from current US and European chemical regulatory frameworks.

## PROFESSIONAL AFFILIATIONS & ACTIVITIES

Member, Society of Environmental Toxicology and Chemistry (1997-Present)

Member, American Chemical Society (2005-Present)

## PUBLICATIONS & PRESENTATIONS

### *Publications*

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