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9/16/08 Bd. Hrng. Item 9
**Sediment Quality Objectives**
Deadline: 9/5/08 by 12:00 p.m.

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September 4, 2008

**VIA FEDEX**

Ms. Jeanine Townsend, Clerk to the Board
Executive Office
State Water Resources Control Board
1001 I Street
Sacramento, CA 95814

File No. 019304-0023

Re: 9/16/08 BOARD MEETING (Public Comment on Agenda Item No. 9: Enclosed Bays and Estuaries – Part 1 Sediment Quality)

Dear Ms. Townsend:

We are submitting the enclosed documents in support of the comments submitted by the California Chamber of Commerce ("CalChamber"), General Electric Company, Montrose Chemical Corporation of California, and National Steel and Shipbuilding Company under separate cover on September 5, 2008. Please find enclosed the following documents:

1. Field LJ, MacDonald DD, Norton SB, Severn CG, Ingersoll CG. 1999. Evaluating Sediment Chemistry and Toxicity Data Using Logistic Regression Modeling. *Environ Toxicol Chem* 18:1311-1322.

2. Field LJ, MacDonald DD, Norton SB, Ingersoll CG, Severn CG, Smorong D, Lindskoog R. 2002. Predicting Amphipod Toxicity From Sediment Chemistry Using Logistic Regression Models. *Environ Toxicol Chem* 21:1993-2005.

3. Fuchsman PC, Barber TR, Lawton JC, Leigh KB. 2006. An Evaluation of Cause-Effect Relationships Between Polychlorinated Biphenyl Concentrations and Sediment Toxicity to Benthic Invertebrates. *Environ Toxicol Chem* 25:2601-2612.

4. May 23, 2008 electronic mail from Daniel Fuchs, Deputy Attorney General, to Eric Katz, Latham & Watkins LLP, copied to Caryn Craig, regarding SQOs, with attachments – e-mail chain with oal.pdf; Withdrawal_Ntc.pdf.

5. Letter received June 3, 2008 from Dale Mentink, Office of Administrative Law, to Eric Katz, Latham & Watkins LLP, regarding Sediment Quality Objectives for Enclosed Bays and Estuaries OAL file no. SWRCB_2008-0229-07S, with enclosures.

Please feel free to contact me at (714) 755-8256 if you have any questions regarding this submittal.

Very truly yours,

Robert L. Dickson, Jr.
Senior Paralegal

Enclosures

cc: Eric M. Katz, Esq.
Paul Singarella, Esq.

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# EVALUATING SEDIMENT CHEMISTRY AND TOXICITY DATA USING LOGISTIC REGRESSION MODELING

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**Abstract**—This paper describes the use of logistic-regression modeling for evaluating matching sediment chemistry and toxicity data. Contaminant-specific logistic models were used to estimate the percentage of samples expected to be toxic at a given concentration. These models enable users to select the probability of effects of concern corresponding to their specific assessment or management objective or to estimate the probability of observing specific biological effects at any contaminant concentration. The models were developed using a large database ($n$ = 2,524) of matching saltwater sediment chemistry and toxicity data for field-collected samples compiled from a number of different sources and geographic areas. The models for seven chemicals selected as examples showed a wide range in goodness of fit, reflecting high variability in toxicity at low concentrations and limited data on toxicity at higher concentrations for some chemicals. The models for individual test endpoints (e.g., amphipod mortality) provided a better fit to the data than the models based on all endpoints combined. A comparison of the relative sensitivity of two amphipod species to specific contaminants illustrated an important application of the logistic model approach.

**Keywords**—Sediment toxicity Logistic regression Sediment-quality guidelines

## INTRODUCTION

Recognition of the importance of sediment quality relative to the protection of aquatic biota in freshwater, estuarine, and marine ecosystems has led to the increased use of sediment toxicity testing to support contaminated sediment assessment and management programs [1]. Although such toxicity test results typically have been used to support site-specific sediment management decisions, these data can also be used in broader applications. For example, a regional database of matching sediment chemistry and toxicity data for Puget Sound (Washington, USA) was used to develop sediment quality guidelines (SQGs) using the apparent effects threshold (AET) approach [2]. These SQGs were subsequently used to establish sediment management standards for the state of Washington, USA [3]. Similarly, effect ranges, low (ERLs) and effect ranges, median (ERMs) for priority contaminants in coastal sediments were developed for use in the National Oceanic and Atmospheric Administration (NOAA) National Status and Trends Program (NSTP) from a large database that contained matching sediment chemistry and biological effects data from field studies with a variety of test endpoints. The NSTP database also used results from spiked sediment bioassays and equilibrium-partitioning models [4–6]. A similar approach was used to develop threshold effect levels (TELs) and probable effect levels (PELs) for priority substances in coastal sediments in Florida, USA [7] and in freshwater sediments [8]. The SQGs developed from these databases have been used in a variety of applications, including screening contaminants of concern in ecological risk assessments, setting priorities for further investigations, designing monitoring programs, and establishing target cleanup levels at contaminated sites.

Establishment of direct cause-and-effect relationships between sediment-associated contaminant concentrations and toxicity to aquatic organisms using field-collected data that contain mixtures of contaminants is difficult. Nonetheless, the results of previous studies have demonstrated that such associative data can be used to develop SQGs that reliably predict the presence or absence of adverse effects of sediment-associated contaminants [2,6–9]. Although this high level of predictability makes the SQGs relevant for use in numerous applications, it may be possible to make further use of matching sediment chemistry and toxicity data in various site-specific and general applications.

The primary objective of this study is to present an alternative approach for using matched sediment toxicity and chemistry data in evaluations of sediment quality. Rather than developing another set of SQG values, our approach uses a large database of matching sediment chemistry and toxicity data to develop simple logistic-regression models that estimate the probability of observing specific toxic effects at a wide range of concentrations of individual contaminants for selected toxicity test endpoints. These models enable users to select the probability of effects of concern corresponding to their specific

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The views expressed in this paper are those of the authors and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

Table 1. List of data sources, geographic areas, and number of test samples in the saltwater database

| Data source[a] | Geographic area | Number of samples[b] | Reference |
|---|---|---|---|
| NSTP | Long Island Sound, New York | 252 | [15] |
| NSTP | Hudson–Raritan Estuary, New York/New Jersey | 170 | [16] |
| NSTP | Boston Harbor, Massachusetts | 90 | [17] |
| NSTP | Tampa Bay, Florida | 138 | [18] |
| NSTP | San Diego Bay, California | 173 | [19] |
| NSTP | Los Angeles/Long Beach Harbors, California | 90 | [20] |
| EMAP | Virginian Province (9 selected estuaries) | 239 | [21,22] |
| SEDQUAL | Puget Sound, Washington (17 studies) | 671 | [23] |
| BEDS | All 3 coasts of North America (22 studies) | 701 | [7] |

[a] NSTP = NOAA National Status and Trends Program; EMAP = U.S. EPA Environmental Monitoring and Assessment Program; SEDQUAL = State of Washington Department of Ecology Puget Sound Database; BEDS = MacDonald Environmental Sciences, Biological Effects Database for Sediments.
[b] Total number of samples 2,524.

objective or to estimate the probability of observing specific biological effects at any contaminant concentration.

## METHODS

### *Data acquisition and evaluation*

The matching sediment chemistry and toxicity data used in this study were obtained from four primary sources, including the NSTP, U.S. Environmental Protection Agency (U.S. EPA) Environmental Monitoring and Assessment Program (EMAP), State of Washington Department of Ecology Puget Sound Database (SEDQUAL), and MacDonald Environmental Sciences Biological Effects Database for Sediments (BEDS) (Table 1). The data from these sources encompassed many geographic areas along the Atlantic, Gulf, and Pacific coasts of North America and included information on many different toxicity endpoints. The standard 10-d amphipod survival toxicity test [1], with either *Ampelisca abdita* or *Rhepoxynius abronius*, was the most commonly used sediment toxicity test endpoint (Table 2). Although the database includes freshwater, estuarine, and marine data, only estuarine and marine data are discussed in this paper.

Data collected as part of the NSTP, EMAP, and SEDQUAL programs followed the standard protocols for sediment chemistry and toxicity testing developed for each program. Candidate data sets identified from the marine and estuarine BEDS included studies that used a variety of protocols and test endpoints [7]. Individual studies were evaluated for acceptance based on their experimental designs and endpoints, test protocols and environmental conditions, sample handling procedures, control responses, and analytical methods that were consistent with general established procedures [6,7]. All of the data that met the above screening criteria were incorporated into the project database.

Table 2. Number of test samples for standard amphipod sediment bioassays (*Ampelisca abdita* and *Rhepoxynius abronius*, 10-d survival endpoints only) and other test species and endpoints in the saltwater database

| Data source[a] | *Ampelisca* | *Rhepoxynius* | Other |
|---|---|---|---|
| BEDS | 36 | 125 | 540 |
| NSTP | 212 | 166 | 535 |
| EMAP | 239 | NA[b] | NA |
| SEDQUAL | NA | 406 | 265 |
| Total | 487 | 697 | 1,340 |

[a] NSTP = NOAA National Status and Trends Program; EMAP = U.S. EPA Environmental Monitoring and Assessment Program; SEDQUAL = State of Washington Department of Ecology Puget Sound Database; BEDS = MacDonald Environmental Sciences, Biological Effects Database for Sediments.
[b] NA = no data available.

### *Data analysis*

The first step in the process of determining relationships between contaminant concentrations and sediment toxicity involved extracting data from the database into separate tables for individual contaminants. The substances used in this evaluation included lead, mercury, nickel, zinc, fluoranthene, phenanthrene, and total PCBs. Contaminants were selected to include representative chemicals from three major classes of environmental contaminants (i.e., metals, PAHs, and PCBs). The contaminant-specific data tables included the chemical concentration in each sample (normalized either to dry weight or total organic carbon) and the toxicity test results (toxic or nontoxic) for each toxicity test endpoint. The data were sorted in order of increasing concentration, similar to the ascending data tables described by other investigators [4,7,10].

Evaluating the relationship between the concentration of an individual contaminant and toxicity in field-collected sediments is frequently complicated by the presence of multiple contaminants, many of which may be present at very low concentrations. Consequently, the data on samples that were identified as toxic were further screened before inclusion in the analyses for each individual contaminant, so as to minimize the potential for including samples in which the selected contaminant did not contribute substantially to the observed toxicity. Following the screening approach used by Ingersoll et al. [8] and similar to that used by others [4,5,7], the concentration of the selected chemical in each toxic sample was compared with the mean of the concentration of that substance in the nontoxic samples collected in the same study and geographic area. If the concentration of a chemical in an individual toxic sample was less than or equal to the mean concentration of that chemical in the nontoxic samples from that study area, it was considered highly unlikely that the observed toxicity could be attributed to that chemical. Therefore, these toxic samples were not included in the screened data set used for model development for that chemical. All nontoxic samples were included. Samples from field reference stations were treated the same as any other sample and included in the analysis.

In most cases, data analyses were conducted using two primary data groupings: the entire marine database (including samples tested using all species and all endpoints), and marine

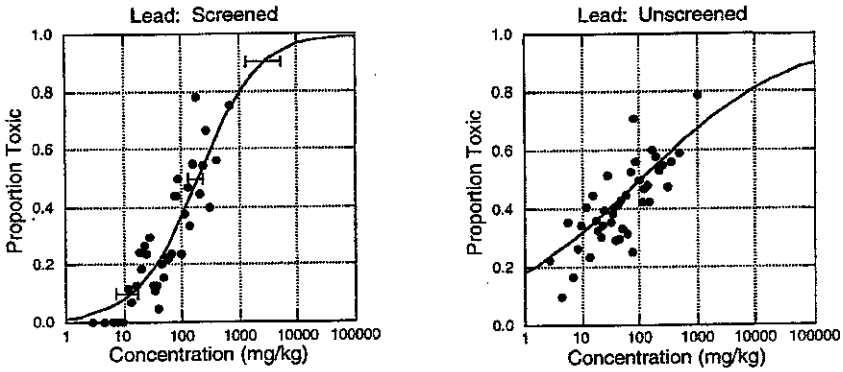


Fig. 1. Proportion of toxic samples in concentration intervals and logistic model plots for screened and unscreened data for lead (mg/kg, dry weight), amphipod survival endpoints only. The individual points represent the median of the sample concentrations within the interval and the fraction of the samples within the interval that was identified as toxic for the selected test endpoint. Each point represents a minimum of 15 individual samples, except for the highest concentration interval, which may have included as few as 10 samples. The horizontal bars for the screened data represent the 95% CI for T10, T50, and T90 values.

amphipod survival only (limited to those samples tested with either *A. abdita* or *R. abronius* in 10-d lethality tests). Data for the two amphipod species were generally combined to increase the sample size for the data analysis.

Individual samples were designated as toxic or nontoxic according to the results reported by the investigator in the original study. This was usually based on a statistical comparison to a negative control. For the data from the Puget Sound database, however, toxicity was determined statistically by comparing toxicity test results with those for field reference stations rather than a negative control. In addition, the Puget Sound amphipod (*R. abronius*) toxicity data required both statistical significance and a minimum of 25% mortality before a sample was defined as toxic. For the other data sets, no additional evaluations (such as defining a minimum effect level or comparing control-adjusted values to a minimum effect level) were conducted for any of the tests. Because the primary focus of this study was to illustrate the potential applications of the logistic model method, differences in the approach used to classify toxic samples were not considered in most of our analyses. Potential implications of ignoring these differences are discussed in a later section of the paper.

### *Concentration-interval plots*

Concentration-interval plots, which display a summary of the matching data on sediment chemistry and toxicity for selected contaminants, were prepared by calculating the proportion of toxic samples within discrete concentration intervals. Unless otherwise noted, these plots were based on the screened data sets. The points shown on the concentration-interval plots represent the median of the sample concentrations within the interval and the fraction of the samples within the interval that was identified as toxic for the selected test endpoint(s) (i.e., Fig. 1). Each point on the figures represents a minimum of 15 individual samples, except for the highest concentration interval, which may have included as few as 10 samples. In many cases, more than 15 samples were included within a concentration interval (i.e., if multiple samples had the same reported concentration of the contaminant). The range represented by each concentration interval was determined from an ascending list of unique sample concentrations for each contaminant, with the number of intervals determined by the total number of unique sample concentrations for the selected contaminant.

### *Logistic modeling*

Fitting statistical models to these screened data provides a means of deriving a relationship between the probability of toxicity ($p$) and the concentration of the chemical of interest. The concentration-interval plots generated data distributions that resembled typical sigmoidal dose–response curves. The shape of these curves indicated that it might be appropriate to use a logistic-regression model, which is commonly used for dose–response data with a binary outcome (i.e., toxic or nontoxic), to model these data distributions [11].

Logistic regression is typically applied to dose–response data, such as that generated by spiked-sediment bioassays. The data screening procedures used in this study were intended to identify the chemicals that plausibly could be considered to be associated with the observed toxic response in individual sediment samples. Therefore, the screening procedures provide a basis for transforming the underlying data into a form that is more consistent with traditional uses of logistic regression modeling.

In its simplest form (i.e., with a single independent variable), the logistic model can be described using the following equation:

$$p = \frac{\exp[B0 + B1(x)]}{1 + \exp[B0 + B1(x)]}$$

where $p$ = probability of observing a toxic effect, B0 = intercept parameter, B1 = slope parameter, and $x$ = chemical concentration or log chemical concentration.

This logistic model was applied to the complete screened data (not the concentration-interval summarized data) for a number of substances to evaluate its utility for modeling matching sediment chemistry and toxicity data. For each of these substances, the intercept (B0), slope (B1), and chi-square statistic (−2 log likelihood) were determined. The chi-square statistic provides valuable information on the statistical significance of the slope parameter and a measure of how well the model fits the data. All of the logistic regression analyses were conducted using the SAS Institute's logistic procedure [12].

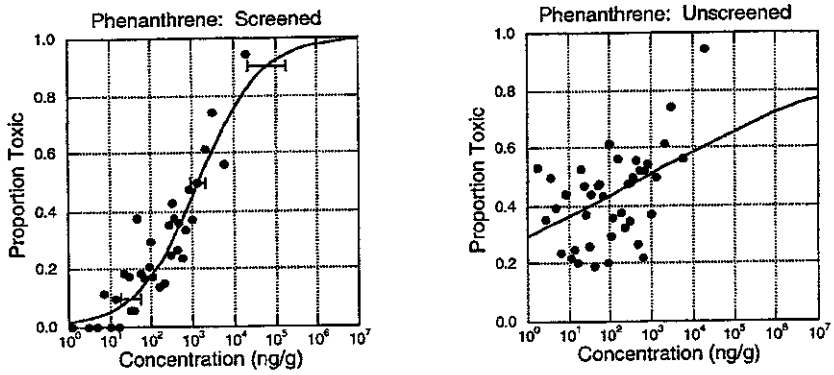


Fig. 2. Proportion of toxic samples in concentration intervals and logistic model plots for screened and unscreened data for phenanthrene (ng/g, dry weight), amphipod survival endpoints only. The horizontal bars for the screened data represent the 95% CI for T10, T50, and T90 values.

Data for each chemical were modeled independently, rather than building a model that analyzed the concentrations of multiple chemicals simultaneously. Thus, only a single concentration variable ($x$) was used in each individual chemical model. However, for each model, it is possible to consider the addition of various covariates, such as test species or endpoints. If such a covariate is considered, then a separate intercept term, a separate slope term, or both a distinct intercept and a distinct slope term can be fit for each level of the covariate.

In the logistic model, the slope parameters describe how the probability of a positive response (toxic effect) increases as the independent variable increases. In the simplest application of the model with a slope, $B1(x)$, an increase in the concentration ($x$) by a specific amount ($dx$) will increase the probability of observing a positive (toxic) response ($p/(1 - p)$) by a factor of $\exp[B1(dx)]$. The slope and intercept parameters for the model were estimated using the maximum-likelihood approach. After the parameters are estimated, the model can be inverted to estimate the concentrations that yield a certain response probability. The notation Tp (e.g., T50) is used to denote the concentration that would give a response of "p" percent according to the model (e.g., the probability that 50% of the samples would be toxic). Confidence intervals for Tp were derived using the delta method, which is based on a truncated Taylor series expansion that utilizes the variance–covariance matrix derived from the maximum-likelihood fit and the derivative of the function of interest (in this case, T10, T50, and T90 were used as examples) with respect to each parameter [11].

The chi-square statistic provides useful information for interpreting the results of the logistic modeling. Specifically, the chi-square statistic was used to determine if the slope parameter, B1, was significantly different from zero. For all of the models generated, the probability ($p$ value) associated with the slope parameter was <0.0001; therefore, the null hypothesis (slope = 0) can be rejected. Additionally, the chi-square statistic can be used to assess how well the model fits the data. For data sets with similar sample sizes, a larger chi-square statistic indicates a better fit of the model to the data. Note, however, that for a similar fit, the chi-square statistic increases with sample size, and thus cannot be used to compare the fit of data sets that are not roughly the same size. For this reason, the chi-square statistic was normalized to the sample size ($N$) to provide a goodness-of-fit measure that could be applied across all the data sets.

## RESULTS

### *Database composition*

The assembled database includes data on a large number of samples with a range of chemical concentrations and many different bioassay endpoints. In total, the database includes data for nearly 1,200 samples tested with the amphipod 10-d survival toxicity tests (Table 2). In addition to the amphipod tests, the compiled studies also evaluated numerous other endpoints, including the responses of bioluminescent bacteria (Microtox®), the survival and normal development of bivalve and echinoderm larvae, and the fertilization success of gastropods and echinoderm gametes. The combined saltwater database included information from studies conducted in more than 40 geographic areas on the Atlantic, Gulf, and Pacific coasts of North America.

Toxic effects were reported in 43% of 487 samples tested with *A. abdita* and in 38% of 697 samples tested with *R. abronius*. Of the 1,340 samples tested with other endpoints, approximately 40% were identified as toxic. Percent total organic carbon in test sediments ranged between 0.02 and 29.4, with a mean concentration of 1.9 (95% CI of 1.79–2.01) for 1,265 test samples. This combined database was used for the analyses discussed in the following sections.

### *Effects of data-screening procedure*

Before applying the logistic model or plotting the data for individual contaminants, the data for each contaminant were screened (as described previously). The data-screening procedure was designed to exclude toxic samples for which the observed toxicity could not be reasonably considered to be associated with the chemical under consideration. The effects of this data-screening procedure on the data distributions for lead (Fig. 1) and phenanthrene (Fig. 2) are illustrated in the concentration-interval plots, which summarize the percentage of toxic samples within discrete concentration intervals. In each case, the respective logistic regression model for the unscreened and screened data was plotted to provide a frame of reference for comparing the data distributions.

Visual inspection of these plots reveals that inclusion of the data for those toxic samples in which the chemical of concern is unlikely to contribute to the observed response tends to scatter and skew the data distributions. The plots of the unscreened data for lead (Fig. 1) and phenanthrene (Fig. 2) show very few intervals with <20% effects. Importantly, the unscreened data for phenanthrene show no relationship be-

tween chemistry and toxicity for concentrations <1,000 ng/g dry weight and the incidence of effects were approximately 20 to 60% below that concentration (Fig. 2). However, few differences were found between the distributions of the screened and unscreened data at higher concentrations. These results suggest that the data-screening technique used in this study helps to clarify the relationship between contaminant concentrations and biological effects in field-collected sediments, particularly at low and intermediate concentrations.

*Logistic model with no covariates*

The logistic model applied to the screened data sets was used to evaluate relationships between contaminant concentrations and adverse effects for selected contaminants. Contaminants were selected to represent different chemical classes (metals, PAHs, and PCBs) and a range of expected quality of concentration–response relationship, that is, based on earlier studies [6,7]. In general, using log(concentration) as the independent variable, rather than concentration, resulted in higher chi-square values; thus only the results for the models using log(concentration) are presented.

Based on the normalized chi-square statistic, the logistic model provided the best fits of the amphipod toxicity data for phenanthrene (Fig. 2), zinc (Fig. 3), and mercury (Fig. 3). The data for nickel (Fig. 3), fluoranthene (Fig. 3), and total PCBs (Fig. 3) provided poorer fits with the logistic model based on chi-square analyses (Table 3). The organic carbon-normalized data models for fluoranthene (Fig. 4) and phenanthrene (Fig. 4) did not fit the data as well as the dry weight-normalized model. The chi-square statistics (normalized for sample size) were similar for both dry weight- and total organic carbon-normalized models for total PCBs (Figs. 3 and 4) and were among the lowest overall.

The concentration-interval plots provide additional insight into the relationship between concentration and the probability of toxicity. For example, although the normalized chi-square for zinc was relatively high, only 50 to 60% of the samples were toxic in the three highest concentration intervals. For fluoranthene, the rather flat slope of the model appeared to underestimate the percentage of toxic samples for the highest concentration intervals, probably because of the high variability at the low concentrations. Most analyses of the selected chemicals appeared to have only limited data from the higher concentration ranges, as evidenced by the limited number of concentration intervals in which >80% of the samples were toxic.

The chemical concentrations that corresponded to the 10, 50, and 90% effects levels (designated as the T10, T50, and T90 levels) for amphipod survival were determined, along with their confidence intervals (Table 4) and are plotted in Figures 1 to 4. Note that the confidence intervals for the T90 values are wider for most chemicals compared with the T50 or T10 values, with several ranging over an order of magnitude. The concentration-interval plots for virtually all of the chemicals examined indicated that the analyses were limited by the amount of data available for the higher chemical concentrations (i.e., toxic samples were >80% in only a few concentration intervals).

Among all of the selected chemical-specific models, except for PCBs, the logistic model for amphipod toxicity data alone provided a better fit than for all test endpoints combined (Table 5). The T50 effect-level concentrations derived using all endpoints and tests combined were higher than those calculated for amphipod survival alone for all of the chemicals tested, except PCBs. This suggests that most of the other information in the database included endpoints and species that were less sensitive than amphipod survival.

*Comparison of toxicity test endpoints*

The database used in this study contains information on the effects of sediment-associated contaminants for a wide range of aquatic organisms and test endpoints. Compilation of this information into a single database facilitates comparisons of the relative sensitivities of the organisms tested and the various endpoints investigated. To demonstrate how the logistic-regression model approach can be used to compare the sensitivity of different toxicity test species or endpoints, the model was applied to the data on the toxicity (10-d survival endpoints only) of various chemicals to the amphipods *R. abronius* and *A. abdita.* with the test species treated as a covariate (Table 6). Note that these comparisons do not require that the same sample be tested concurrently with both amphipod species.

Using the logistic-regression model, it is possible to fit a separate slope, intercept, or both for each of the two amphipod species. The results of this analysis indicated that the additional intercept term was statistically significant (at the $\alpha = 0.05$ level) for all of the chemicals considered (Table 6). However, the distinct slope term was not significant at this $\alpha$-level for lead, mercury, zinc, and phenanthrene. For nickel, fluoranthene, and PCBs, both terms (distinct slope and intercept) were significant, but most of the increase in the goodness of fit came from the addition of the intercept term. Thus, for all of these chemicals the results of the common slope-distinct intercept model were used to calculate the specific Tp concentrations for each species (Table 7). These results can be compared with the results in Tables 3 and 4, which apply to the combined data for both species.

These analyses suggest differences between the two marine amphipod species in their sensitivities to sediment-associated contaminants. For example, the logistic models developed for lead, mercury, zinc, and phenanthrene (Fig. 5) illustrate this apparent difference between the two species. Comparison of the T10, T50, and T90 values derived for each of these species also suggests that *A. abdita* is more sensitive to the effects of sediment-associated contaminants than is *R. abronius.* These apparent differences in sensitivity were relatively small (a difference of less than a factor of three in T50 values) for the metals tested (e.g., lead, mercury, nickel, and zinc), but were almost an order of magnitude different for fluoranthene, phenanthrene, and total PCBs.

Although the results of these evaluations suggest systematic differences in the sensitivities of these two amphipod species, a closer examination of the data revealed that the differences may be attributable to differences in classifying marginally toxic samples. A sample classified as toxic in the Puget Sound *R. abronius* data, which represented more than 400 samples, was required to have <75% survival. In contrast, almost 40% of the *A. abdita* samples classified as toxic, based on statistical comparisons to the negative control, had >80% control-adjusted survival (where control-adjusted survival represents the ratio of the mean test-sample survival to the mean control survival). These samples almost certainly would have been classified as nontoxic in the Puget Sound study.

To further investigate this apparent difference, we conducted two additional comparisons using the data for the four contaminants (lead, mercury, zinc, and phenanthrene) with the

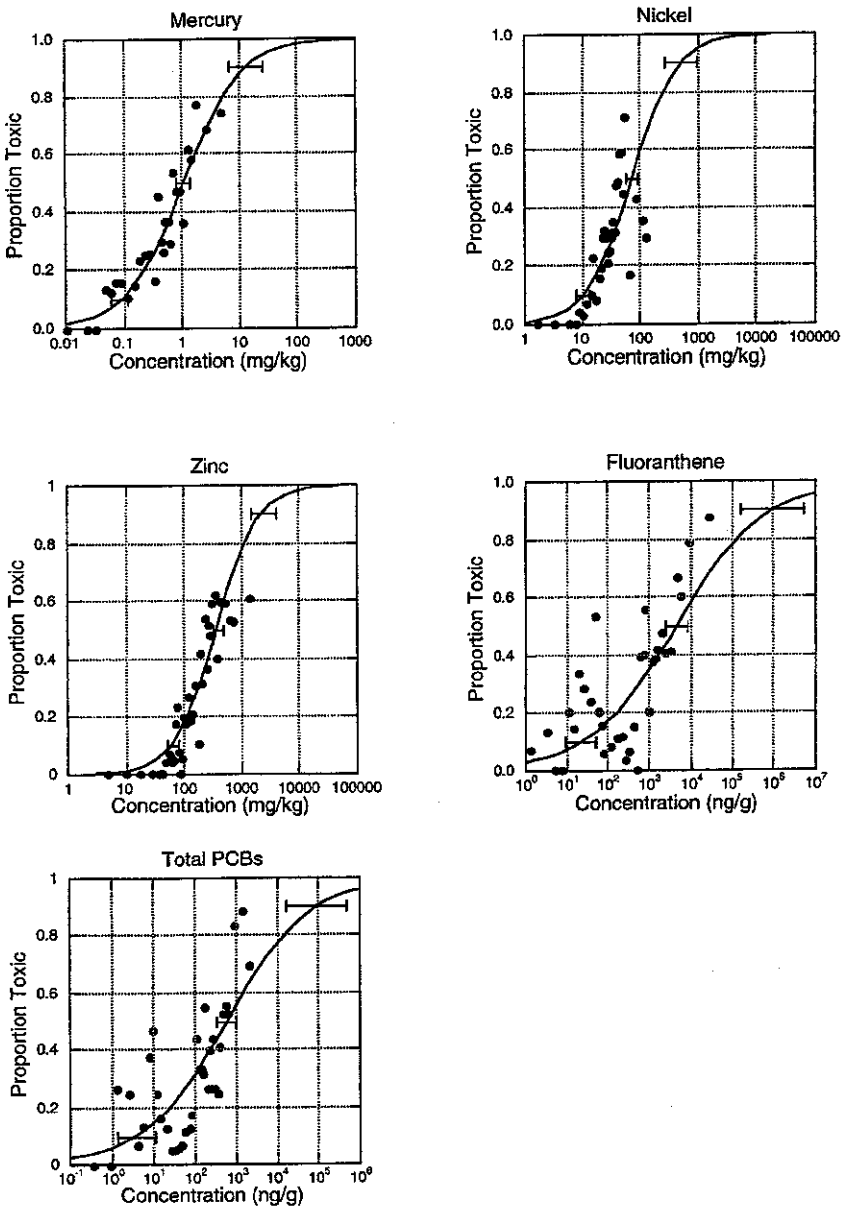


Fig. 3. Logistic model with 95% CI for T10, T50, and T90 values and proportion of samples toxic in concentration intervals for amphipod survival endpoints only: mercury (mg/kg), nickel (mg/kg), zinc (mg/kg), fluoranthene (ng/g), and total PCBs (ng/g).

Table 3. Results of logistic model for log chemical concentration (dry wt. and organic carbon, both normalized) and toxicity data for amphipod (*Ampelisca abdita* and *Rhepoxynius abronius*) 10-d survival test endpoints

| Chemical | Units[a] | Intercept (B0) | Slope (B1) | Chi-square value | No. of samples | Chi-square value/*N* |
|---|---|---|---|---|---|---|
| Lead | mg/kg, DW | −4.35 | 1.91 | 156.31 | 911 | 0.17 |
| Mercury | mg/kg, DW | −0.06 | 2.03 | 172.86 | 796 | 0.22 |
| Nickel | mg/kg, DW | −4.84 | 2.61 | 97.88 | 890 | 0.11 |
| Zinc | mg/kg, DW | −7.15 | 2.75 | 179.49 | 925 | 0.19 |
| Fluoranthene | ng/g, DW | −3.46 | 0.95 | 94.23 | 743 | 0.13 |
| Phenanthrene | ng/g, DW | −4.19 | 1.33 | 145.11 | 751 | 0.19 |
| PCBs, total | ng/g, DW | −2.78 | 1.01 | 78.43 | 606 | 0.13 |
| Fluoranthene | mg/kg, TOC | −2.49 | 1.01 | 75.43 | 715 | 0.11 |
| Phenanthrene | mg/kg, TOC | −2.72 | 1.38 | 109.51 | 721 | 0.15 |
| PCBs, total | mg/kg, TOC | −1.72 | 1.12 | 63.59 | 584 | 0.11 |

[a] DW = Dry weight, normalized; TOC = Total organic carbon, normalized.

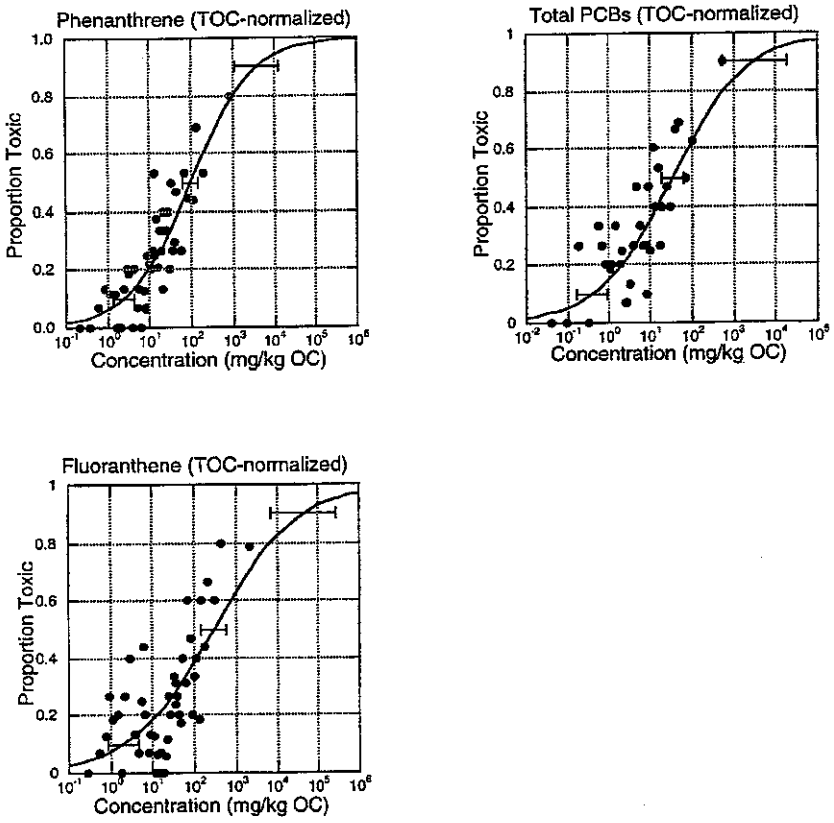


Fig. 4. Logistic model with 95% CI for T10, T50, and T90 values and proportion of samples toxic in concentration intervals for amphipod survival endpoints only for TOC-normalized concentrations (mg/kg, OC) of fluoranthene, phenanthrene, and total PCBs.

highest normalized chi-square statistics in the original models. To make the treatment of the data more similar, we reclassified the *A. abdita* data using a control-adjusted minimum significant difference (MSD) approach [13]. In this approach, samples are classified as toxic if they significantly differ from the negative control and if the control-adjusted survival is <80%; all other samples are classified as nontoxic. Because of limitations in the available data, we were unable to reclassify the toxicity data using the same approach for classifying samples as toxic and nontoxic for both species. However, the MSD approach is similar to the approach that was used for *Rhepoxynius* in Puget Sound. Logistic-regression models were used to compare the original classification method (based on a statistically significant difference from the negative control) with the control-adjusted MSD approach for three chemicals (Table 8). The results showed that the control-adjusted MSD approach yielded T50 values that were higher than those derived using the original method. The difference between the models was largest at lower concentrations (Fig. 6).

Comparison of models for *Ampelisca* (using the control-adjusted MSD approach to classification of toxic samples) and *R. abronius* (using the Puget Sound classification approach) reveals that neither the separate intercept nor the separate intercept and slope model fit the data statistically better than the

Table 4. Logistic model point estimates of T10, T50, and T90 effect concentrations with 95% CIs for amphipod (*Ampelisca abdita* and *Rhepoxynius abronius*) 10-d survival test endpoints[a]

| | | T10 | | T50 | | T90 | |
|---|---|---|---|---|---|---|---|
| Chemical | Units | Value | (95% CI) | Value | (95% CI) | Value | (95% CI) |
| Lead | mg/kg, DW | 13.4 | (9.5–19.1) | 191 | (146–249) | 2,700 | (1,400–5,300) |
| Mercury | mg/kg, DW | 0.09 | (0.06–0.12) | 1.1 | (0.82–1.39) | 13 | (7–25) |
| Nickel | mg/kg, DW | 10.2 | (7.7–13.7) | 71 | (57–90) | 495 | (262–935) |
| Zinc | mg/kg, DW | 63 | (50–80) | 395 | (328–477) | 2,500 | (1,550–3,975) |
| Fluoranthene | ng/g, DW | 21.5 | (9.0–51) | 4,450 | (2,400–8,300) | 920,000 | (160,000–5,200,000) |
| Phenanthrene | ng/g, DW | 31.1 | (17.6–55) | 1,400 | (930–2,100) | 61,000 | (22,00–175,000) |
| PCBs, total | ng/g, DW | 3.8 | (1.4–10.1) | 581 | (328–1,030) | 88,500 | (16,000–490,000) |
| Fluoranthene | mg/kg, TOC | 1.9 | (0.9–4.4) | 292 | (149–572) | 44,100 | (7,100–275,000) |
| Phenanthrene | mg/kg, TOC | 2.4 | (1.4–4.2) | 95 | (60–150) | 3,750 | (1,150–12,200) |
| PCBs, total | mg/kg, TOC | 0.4 | (0.2–0.9) | 35 | (19–63) | 3,150 | (550–18,000) |

[a] The notation Tp (e.g., T50) is used to denote the concentration that would give a response of "p" percent according to the model (e.g., the probability that 50% of the samples would be toxic. DW = dry weight, normalized; TOC = total organic carbon, normalized.

Table 5. Comparison of logistic model estimates of T50 effect concentrations with 95% CIs and normalized chi-square statistic (chi-sq/*N*) for amphipod (*Ampelisca abdita* and *Rhepoxynius abronius*) 10-d survival test endpoints and for all test endpoints combined[a]

| | | Amphipod survival only | | | All test endpoints | | |
|---|---|---|---|---|---|---|---|
| Chemical | Units | T50 | (95% CI) | Chi-sq/*N* | T50 | (95% CI) | Chi-sq/*N* |
| Lead | mg/kg, DW | 191 | (146–249) | 0.17 | 217 | (177–266) | 0.14 |
| Mercury | mg/kg, DW | 1.1 | (0.8–1.4) | 0.22 | 1.6 | (1.2–2.1) | 0.11 |
| Nickel | mg/kg, DW | 71 | (57–90) | 0.11 | 111 | (81–152) | 0.06 |
| Zinc | mg/kg, DW | 395 | (328–477) | 0.19 | 481 | (403–573) | 0.14 |
| Fluoranthene | ng/g, DW | 4,450 | (2,400–8,300) | 0.13 | 4,750 | (3,150–7,200) | 0.12 |
| Phenanthrene | ng/g, DW | 1,400 | (930–2,100) | 0.19 | 2,100 | (1,450–3,000) | 0.15 |
| PCBs, total | ng/g, DW | 581 | (328–1,030) | 0.13 | 440 | (319–607) | 0.13 |

[a] See Table 4 for definition of T50. DW = dry weight, normalized.

common intercept and common slope model for all four chemicals evaluated (Table 9). This indicates that the null hypothesis (i.e., that the two species have the same logistic concentration–response curve) cannot be rejected (at the $\alpha = 0.05$ level) for these chemicals. The results of these comparisons suggest that the two amphipod species have essentially the same sensitivities to these four chemicals when comparable methods of classifying toxic samples were used.

### *Using the models to compare existing guidelines*

The contaminant-specific logistic models provide a means of comparing the performance of existing SQGs using a large database of matching sediment chemistry and toxicity data. By using the individual SQG values as input parameters, the logistic models for individual chemicals can be used to estimate the proportion of samples expected to be toxic at specific contaminant concentrations (Table 10). For example, 15 to 31% of the samples with contaminant concentrations equal to ERLs would be toxic to amphipods, based on the results obtained for six selected substances. By comparison, 38 to 53% of the samples would be toxic to amphipods at contaminant concentrations equal to the ERMs. A higher incidence of toxicity to amphipods would be expected at contaminant concentrations equal to the amphipod AETs (i.e., 66–94%).

## DISCUSSION

### *Evaluation and application of logistic-model approach*

The statistical analyses conducted in this study indicate that logistic-regression models can be used to describe the relationships between sediment chemistry and measures of biological effects. As a result, it may be possible to use models like these to predict the probability of observing adverse effects at measured concentrations of sediment-associated contaminants. An evaluation of the predictive ability of these models with independent data is an essential next step.

Table 6. Statistical comparisons between three logistic-regression models for survival toxicity test of two amphipods (*Ampelisca abdita* and *Rhepoxynius abronius*) using the chi-square statistic (–2 log likelihood). Model A = common slope, common intercept combined model for both species; Model B = common slope, separate intercept model; Model C = separate slope, separate intercept model

| | | | | Model comparison | |
|---|---|---|---|---|---|
| | Chi-square statistic | | | Model B versus Model A | Model C versus Model B |
| Chemical | Model A | Model B | Model C | (*p*-value) | (*p*-value) |
| Lead | 156.313 | 176.101 | 177.006 | <0.001 | 0.34 |
| Mercury | 172.858 | 179.497 | 179.501 | 0.010 | 0.95 |
| Nickel | 97.876 | 133.081 | 153.238 | <0.001 | <0.001 |
| Zinc | 179.494 | 205.266 | 205.267 | <0.001 | 0.97 |
| Fluoranthene | 94.231 | 124.518 | 133.294 | <0.001 | 0.003 |
| Phenanthrene | 145.113 | 177.189 | 177.195 | <0.001 | 0.94 |
| PCBs, total | 78.425 | 104.277 | 122.536 | <0.001 | <0.001 |

Despite the benefits attributable to the database and associated analytical procedures, some limitations could restrict the application of these tools. Importantly, only a limited amount of data is available for high concentrations of many of the chemicals represented in the database. Hence, the upper portions of the curves fitted by the logistic model are poorly defined, which generates substantial uncertainty in the highest effect concentrations calculated (e.g., T90s).

Another challenge associated with these analyses concerns the inconsistent treatment of marginally toxic samples, particularly the many marginally toxic samples classified as toxic in the *A. abdita* survival assays. Classifying these samples either as toxic or nontoxic in the analyses may cause toxicity to be over- or underestimated and thus create a bias in the resultant model. To adequately compare the sensitivity of different toxicity test species or endpoints, it is imperative to standardize the treatment of the toxicity classification. The results presented here do not suggest that one approach is superior to the other, because the models for the different approaches to classification of toxic samples for *A. abdita* had similar goodness-of-fit measures.

The presence of contaminant mixtures makes it difficult to use matching field-collected data on sediment chemistry and toxicity to derive universally applicable concentration–response relationships for individual chemicals. For a given chemical, toxicity in field-collected samples may be observed at low concentrations where toxicity would not be expected (false negatives) or may not be observed at high concentrations where toxicity would be expected according to the model (false positives). In the former case, where the observed toxicity may be caused by other contaminants, the data-screening method (comparing the concentration of the toxic samples with the mean of the concentrations of the nontoxic samples) was designed to filter out many of those samples for a particular chemical before performing the logistic regressions. However, for chemicals such as fluoranthene and PCBs, this screening process did not effectively eliminate high variability at low concentrations, and the resultant models did not provide as good a fit to the data compared with the other models based on screened data.

For several of the contaminants evaluated, some samples with very high concentrations were not toxic to amphipods.

Table 7. Logistic model point estimates of effect concentrations (T10, T50, and T90) and chi-square statistic for selected chemicals using the common slope/separate intercept species covariate model for two amphipod species (*Ampelisca abdita* and *Rhepoxynius abronius*)[a]

| Chemical | Units | *Rhepoxynius* | | | *Ampelisca* | | | Chi-square value[b] |
|---|---|---|---|---|---|---|---|---|
| | | T10 | T50 | T90 | T10 | T50 | T90 | |
| Lead | mg/kg, DW | 22 | 257 | 3,027 | 9.3 | 110 | 1,293 | 19.8 |
| Mercury | mg/kg, DW | 0.1 | 1.3 | 15.5 | 0.1 | 0.8 | 9.0 | 6.6 |
| Nickel | mg/kg, DW | 17 | 88 | 446 | 8.1 | 41 | 210 | 35.2 |
| Zinc | mg/kg, DW | 91 | 509 | 2,857 | 46 | 256 | 1,439 | 25.8 |
| Fluoranthene | ng/g, DW | 73 | 8,734 | 1,046,099 | 8.0 | 957 | 114,597 | 30.3 |
| Phenanthrene | ng/g, DW | 80 | 2,316 | 67,271 | 15 | 437 | 12,682 | 32.1 |
| PCBs, total | ng/g, DW | 18 | 1,120 | 70,578 | 2.57 | 162 | 10,222 | 25.9 |

[a] See Table 4 for definition of T10, T50, and T90. DW = dry weight, normalized.
[b] Chi-square statistic is for logistic model with separate intercept term for each species (1 df).

These may represent samples where the contaminants were not in a bioavailable form (e.g., PAHs in coal particles or metals present in slag or paint chips). Compiling a large database helps to reduce the influence of a small number of samples with highly elevated but nonbioavailable contaminant concentrations. Compiling data from many different areas with sufficient numbers of samples with high concentrations may make it possible to minimize the effect of these samples on the results. However, for most of the chemicals selected as examples, only limited data were available for highly contaminated samples.

### *Using the models to estimate effect concentrations*

The logistic-model approach described in this report provides a statistical method for estimating effect concentrations using the information contained in the database. The model permits users to select appropriate effect concentrations for their specific application. For example, T10, T15, or T20 values could be calculated and used to identify contaminant concentrations for which the predicted incidence of effects would be 10, 15, or 20%, respectively. Similarly, contaminant concentrations for which a high probability of observing adverse effects exists could be estimated by calculating, for example,

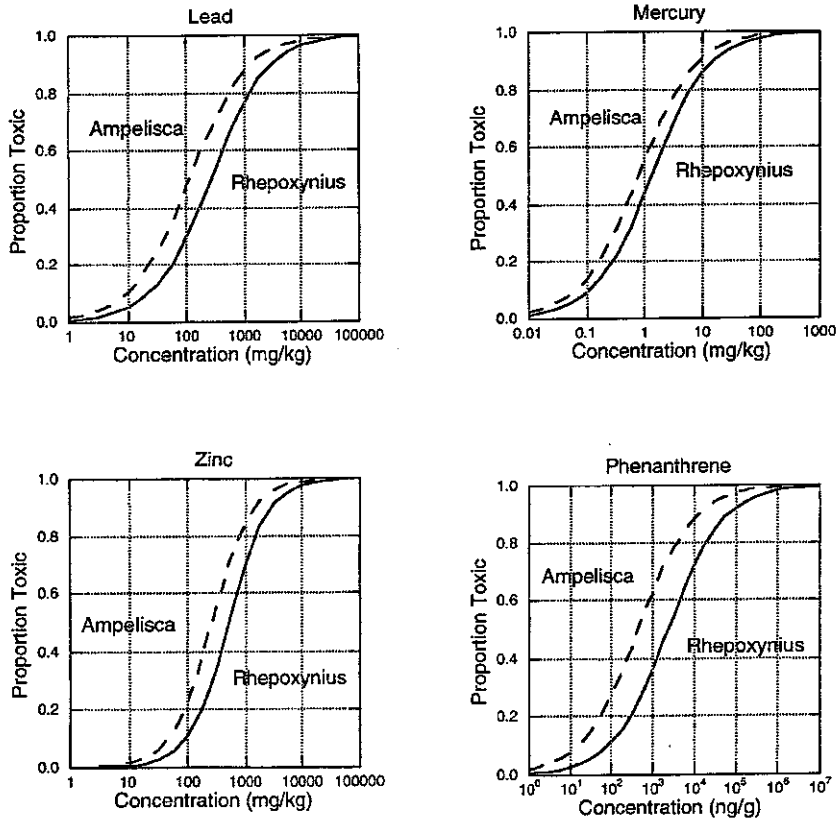


Fig. 5. Comparison of common slope/different intercept logistic model results for lead (mg/kg, dry weight), mercury (mg/kg, dry weight), zinc (mg/kg, dry weight), and phenanthrene (ng/g, dry weight) with toxicity data for the two amphipod species (*Ampelisca abdita* and *Rhepoxynius abronius*) treated as covariates.

Table 8. Comparison of logistic model estimates of T50 effect concentrations with 95% confidence intervals and normalized chi-square statistic (chi-sq/*N*) for two methods of classifying samples as toxic for *Ampelisca abdita* 10-d survival test endpoints[a]

| | | Significance only | | | Significance and >MSD | | |
|---|---|---|---|---|---|---|---|
| Chemical | Units | T50 | (95% CI) | Chi-sq/*N* | T50 | (95% CI) | Chi-sq/*N* |
| Lead | mg/kg, DW | 118 | (82–171) | 0.17 | 210 | (150–292) | 0.21 |
| Mercury | mg/kg, DW | 0.8 | (0.5–1.1) | 0.28 | 1.4 | (1.0–2.0) | 0.32 |
| Zinc | mg/kg, DW | 256 | (205–320) | 0.24 | 437 | (338–565) | 0.21 |
| Phenanthrene | ng/g, DW | 434 | (273–690) | 0.28 | 1,145 | (704–1,860) | 0.27 |

[a] See Table 4 for definition of T50. MSD = minimum signficant difference; DW = dry weight, normalized.

T70, T80, or T90 values. An important advantage of this approach is its ability to calculate confidence intervals for a selected effect concentration, making it easier to evaluate uncertainty associated with fitting the model. However, it should be noted that greater uncertainty occurs away from the median (T50) response.

*Evaluation of site-specific data*

Hazardous waste-site evaluations often involve the collection of sediment chemistry and toxicity data. Logistic-regression models, derived from a large database of samples from many geographic regions, can serve as a frame of reference for evaluating site-specific data. That is, the logistic models provide a basis for determining if evidence exists of differences in the bioavailability of contaminants due to regional geochemistry. If the site-specific data are consistent with the data from other areas, it would be reasonable to use the models derived from a large database to further evaluate the hazards posed by sediment-associated contaminants at the site under consideration.

The existing SQGs (e.g., ERLs, ERMs, TELs, PELs, AETs) provide threshold values for evaluating sediment quality. For a selected contaminant, the concentration in a sediment sample is either above or below the threshold; however, it is difficult to determine the increase in risk associated with the extent to which the threshold is exceeded. Although use of these existing SQGs is possible to calculate a hazard quotient (HQ; the ratio of the sample concentration to the threshold concentration), the concentration–response relationships for each contaminant cannot be assumed to be necessarily the same [14]. Thus, depending on which chemicals were considered, an HQ of 3 might have different implications for the likelihood of toxicity. In contrast to the existing SQGs, the logistic-model approach allows a user to establish the risk level of concern (i.e., the

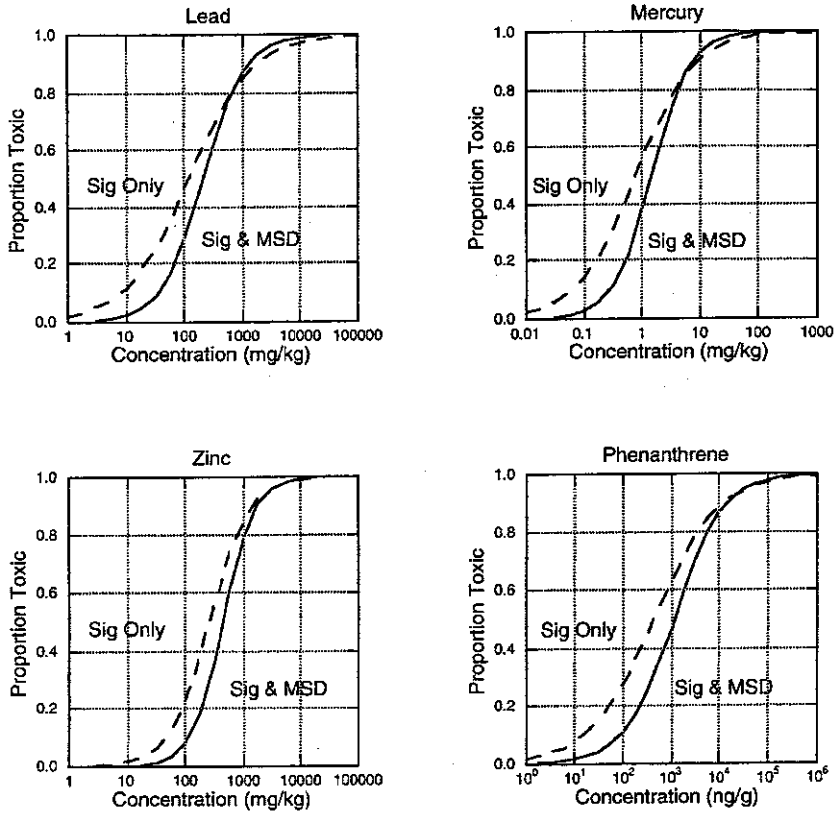


Fig. 6. Comparison of logistic model results for lead (mg/kg, dry weight), mercury (mg/kg, dry weight), zinc (mg/kg, dry weight), and phenanthrene (ng/g, dry weight) with toxicity data for *Ampelisca abdita*, with toxic samples based on significance only (Sig Only) or significance and control-adjusted mortality greater than the minimum significant difference (Sig & MSD).

Table 9. Statistical comparisons of the logistic regression models for *Ampelisca abdita* (toxic samples with control-adjusted survival less than 80%) and *Rhepoxynius abronius* (toxic samples with survival less than 75%) using the chi-square statistic (–2 log likelihood). Model A = common slope, common intercept combined model for both species; Model B = common slope, separate intercept model; Model C = separate slope, separate intercept model

| | Chi-square statistic | | | Model comparison | |
|---|---|---|---|---|---|
| Chemical | Model A | Model B | Model C | Model B versus Model A ($p$-value) | Model C versus Model B ($p$-value) |
| Lead | 197.4 | 198.2 | 201.8 | 0.371 | 0.059 |
| Mercury | 217.3 | 220.8 | 224.5 | 0.062 | 0.055 |
| Zinc | 202.3 | 202.3 | 204.8 | 0.865 | 0.114 |
| Phenanthrene | 196.1 | 198.3 | 199.9 | 0.133 | 0.207 |

probability that a sample would be toxic) and estimate the corresponding concentration for each chemical of concern.

### *Using the models to compare sensitivities of species and endpoints*

With sufficient data, contaminant-specific logistic models can be derived for individual toxicity test species and endpoints. Although not the same as comparing endpoints based on toxicity tests conducted on the same samples, the logistic-model comparisons provide an objective basis for evaluating the relative sensitivity of different tests (including both freshwater and saltwater tests) to different contaminants. The same approach could also be used to compare models for the same toxicity test from different geographic regions to evaluate regional differences in conditions that could affect toxicity test results. The comparisons between the two amphipod species also demonstrate the importance of using a consistent approach to classify samples as toxic or nontoxic.

### *Framework for evaluating existing SQGs*

Available SQGs have different narrative objectives. For example, ERLs are intended to represent chemical concentrations below which toxicity would occur only rarely. Endpoint-specific AET values represent concentrations above which toxicity would be always expected for that endpoint. Although the SQG derivation methods are well described and are consistent for all of the chemicals within a given type of SQG, no straightforward way exists for the user to either evaluate the degree to which individual SQGs meet their objectives or to evaluate the reliability of individual SQGs. The logistic-model approach provides a way to put the individual SQG values into perspective with a large amount of field-collected data and a measure of goodness of fit.

## CONCLUSIONS

This paper illustrates the potential applications of the logistic-modeling approach for evaluating matched sediment chemistry and sediment toxicity data. Compilation of a large database, including data from different geographic areas and contamination gradients, provides sufficient numbers of samples and ranges of concentrations to test the logistic-model approach. This database also makes it possible to evaluate individual toxicity test endpoints to reduce the variability associated with combining data from different tests. However, the lack of consistency in classifying toxic samples among the studies included in the database increased the variability in the data. The models for amphipod survival endpoints alone consistently provided a better fit to the data than did models based on multiple toxicity test species and endpoints. Likewise, the models for dry weight-normalized values for three nonpolar organic contaminants appeared to be more reliable than the equivalent organic carbon-normalized models. Further work is needed to evaluate the ability of logistic-regression models to predict toxicity in environmental samples using independent data sets.

The combined concentration-interval and logistic-model approach provides several advantages relative to previous approaches: concentration-interval plots display a summary of the screened matching data on sediment chemistry and toxicity for the selected contaminant, allowing the user to see the distribution of the data; the user can either select the desired effect level(s) (e.g., T50) or use the model to predict the probability of effects associated with specific sediment concentrations; the methods permit the calculation of confidence intervals for a selected effect level; the comparative reliability of the individual models can be assessed using the goodness-of-fit statistic; different toxicity test endpoints and sediment-normalization approaches can be compared in a consistent and unbiased manner; and the method provides a framework to independently evaluate existing or proposed SQGs, using a large database of matching sediment chemistry and toxicity data.

Based on our preliminary results, the logistic-modeling approach is a useful tool for evaluating matching sediment chemistry and toxicity data. However, several refinements would improve the applicability of the database and analytical procedures in the future. For example, a consistent definition of toxic and nontoxic should be established for each endpoint. Because of the importance of the screening procedure, further investigation into the effects of variations in the screening

Table 10. Logistic model (amphipod only) estimated proportion of samples with toxic effects for ERL, ERM, and AET (1994 amphipod) concentrations for selected chemicals[a]

| | | SQG values | | | Estimated proportion toxic | | |
|---|---|---|---|---|---|---|---|
| Chemical | Units | ERL | ERM | AET | ERL | ERM | AET |
| Lead | mg/kg, DW | 46.7 | 218 | 1,200 | 0.24 | 0.53 | 0.82 |
| Mercury | mg/kg, DW | 0.15 | 0.7 | 2.3 | 0.15 | 0.41 | 0.66 |
| Zinc | mg/kg, DW | 150 | 410 | 3,800 | 0.24 | 0.51 | 0.94 |
| Fluoranthene | ng/g, DW | 600 | 5,100 | 30,000 | 0.31 | 0.52 | 0.69 |
| Phenanthrene | ng/g, DW | 240 | 1,500 | 21,000 | 0.26 | 0.51 | 0.83 |
| PCBs | ng/g, DW | 22.7 | 180 | 3,100 | 0.20 | 0.38 | 0.68 |

[a] ERL = effect ranges, low; ERM = effect ranges, medium; AET = apparent effects threshold; SQG = sediment quality guideline.

approach is needed. Including additional data on the biological effects of sediment-associated contaminants to freshwater organisms would facilitate the development of logistic models that apply to freshwater sediments. Also, data from both freshwater and marine locations with high concentrations of contaminants would help to define the upper portions of the data distributions. Similarly, calculation of alternative ecotoxicologic thresholds may help to overcome some of the existing limitations of the data (i.e., minimal data on upper range of contaminant concentrations). Finally, the development of simple toxic-unit models may provide a means of accounting for contaminant mixtures in sediment quality assessments. We hope to explore these and several other questions during the next stages of this study.

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# PREDICTING AMPHIPOD TOXICITY FROM SEDIMENT CHEMISTRY USING LOGISTIC REGRESSION MODELS

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**Abstract**—Individual chemical logistic regression models were developed for 37 chemicals of potential concern in contaminated sediments to predict the probability of toxicity, based on the standard 10-d survival test for the marine amphipods *Ampelisca abdita* and *Rhepoxynius abronius*. These models were derived from a large database of matching sediment chemistry and toxicity data, which includes contaminant gradients from a variety of habitats in coastal North America. Chemical concentrations corresponding to a 20, 50, and 80% probability of observing sediment toxicity (T20, T50, and T80 values) were calculated to illustrate the potential for deriving application-specific sediment effect concentrations and to provide probability ranges for evaluating the reliability of the models. The individual chemical regression models were combined into a single model, using either the maximum ($P_{Max}$ model) or average ($P_{Avg}$ model) probability predicted from the chemicals analyzed in a sample, to estimate the probability of toxicity for a sample. The average predicted probability of toxicity (from the $P_{Max}$ model) within probability quartiles closely matched the incidence of toxicity within the same ranges, demonstrating the overall reliability of the $P_{Max}$ model for the database that was used to derive the model. The magnitude of the toxic effect (decreased survival) in the amphipod test increased as the predicted probability of toxicity increased. Users have a number of options for applying the logistic models, including estimating the probability of observing acute toxicity to estuarine and marine amphipods in 10-d toxicity tests at any given chemical concentration or estimating the chemical concentrations that correspond to specific probabilities of observing sediment toxicity.

**Keywords**—Sediment toxicity Sediment guidelines Logistic regression

## INTRODUCTION

The contribution of contaminated sediments to effects on sediment-dwelling organisms (including plants and invertebrates), aquatic-dependent wildlife (amphibians, reptiles, fish, birds, and mammals), and human health has become more apparent in recent years [1,2]. Many toxic contaminants (such as metals, polycyclic aromatic hydrocarbons [PAHs], polychlorinated biphenyls [PCBs], chlorophenols, and pesticides) are found in only trace amounts in water but can accumulate to elevated levels in sediments [3]. As such, sediments can serve both as reservoirs and as potential sources of contaminants to the water column. In addition, sediment-associated contaminants can adversely affect sediment-dwelling organisms by causing direct toxicity or altering benthic invertebrate community structure [4]. Furthermore, contaminated sediments can adversely affect fish and wildlife species, either through direct exposure or through bioaccumulation in the food web. While carefully designed monitoring programs can detect effects on sediment communities, fish, and wildlife, the concentrations of chemicals in sediments can provide useful information for evaluating risks to sediment-dwelling organisms, wildlife, or human health from releases of toxic or bioaccumulative substances into the environment.

A variety of indicators provide information on the status of marine and estuarine sediments relative to ecological receptors [3]. These indicators include sediment chemistry, sediment toxicity, benthic invertebrate community status, and bioaccumulation assessments. While the results of sediment toxicity tests and benthic invertebrate community assessments can be used directly to evaluate or infer effects on sediment-dwelling organisms, effective interpretation of sediment chemistry data requires tools that link chemical concentrations to the potential for observing adverse biological effects [5]. Numerical sediment quality guidelines (SQGs) are commonly used in this capacity [6–8].

The logistic regression model (LRM) approach described in this paper is similar to other empirical approaches for deriving SQGs because it relies on matching field-collected sediment chemistry and biological effects (e.g., sediment toxicity or benthic invertebrate community structure effects) data. In contrast to other approaches to developing SQGs, however, the LRM approach does not develop threshold values. Instead, the approach develops models that enable users to select the probability of observing sediment toxicity that corresponds to their specific objectives or to estimate the probability of observing effects at a particular chemical concentration [5].

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Table 1. Number of samples and percentage of samples toxic summarized by amphipod species and data source. Samples were classified as toxic if significantly different from control and less than 90% survival (SIG only) and if significantly different from control and less than 80% control-adjusted survival (SIG and MSD). NA = no data

| | *Ampelisca abdita* | | | *Rhepoxynius abronius* | | |
|---|---|---|---|---|---|---|
| | | Percentage toxic | | | Percentage toxic | |
| Data source | No. of samples | SIG only | SIG and MSD | No. of samples | SIG only | SIG and MSD |
| EMAP[a] | 1,203 | 22.2 | 9.5 | NA | NA | NA |
| NST[b] | 649 | 23.7 | 15.6 | NA | NA | NA |
| MLML[c] | 43 | 11.6 | 7.0 | 465 | 72.7 | 52.3 |
| SEDQUAL[d] | NA | NA | NA | 594 | 63.5 | 34.0 |
| BEDS[e] | 117 | 41.0 | 30.8 | 152 | 36.8 | 32.2 |
| Total | 2,012 | 23.6 | 12.6 | 1,211 | 63.7 | 40.8 |

[a] EMAP = U.S. Environmental Protection Agency Estuarine Monitoring and Assessment Project.
[b] NST = National Oceanic and Atmospheric Administration Status and Trends program.
[c] MLML = Moss Landing Marine Laboratory, California, USA.
[d] SEDQUAL = Washington State Department of Ecology, Sediment Quality Database.
[e] BEDS = MacDonald Environmental Sciences, Biological Effects Database for Sediments.

Logistic regression models require a large database of matching sediment chemistry and toxicity data that includes a broad range of concentrations. Since the preliminary logistic models were developed [5], the underlying sediment toxicity database has been substantially expanded from 1,200 to 3,200 samples. The primary objectives of this paper are to describe the development of individual chemical logistic regression models for an expanded list of analytes, based on the standard marine and estuarine amphipod 10-d lethality toxicity test endpoint, and to combine these individual models into a single model for predicting toxicity in field-collected sediment samples. In addition, the paper illustrates the applications of the individual logistic models for evaluating sediment quality guidelines and the use of the combined models to predict toxicity for an independent data set.

## METHODS

### *Data acquisition and evaluation*

This investigation compiled synoptically collected sediment chemistry and sediment toxicity data from throughout North America. The primary sources of these data included the National Oceanic and Atmospheric Administration's (NOAA) National Status and Trends program (NST), U.S. Environmental Protection Agency's (U.S. EPA) Environmental Monitoring and Assessment Program (EMAP), Moss Landing Marine Laboratory (MLML, which compiled data for the state of California), State of Washington Department of Ecology's Puget Sound Database (SEDQUAL), and MacDonald Environmental Sciences' Biological Effects Database for Sediments (BEDS; Table 1). Many geographic areas along the Atlantic, Gulf, and Pacific coasts are represented in the database. Although the database includes information on a variety of toxicity endpoints, only data from the American Society for Testing Materials (ASTM) standard 10-d amphipod survival toxicity tests with *Ampelisca abdita* and *Rhepoxynius abronius* were used in the analyses discussed in this paper.

All the candidate data sets considered for inclusion in the database were critically evaluated. Acceptance criteria applied to individual studies provided a basis for determining whether experimental designs and measurement endpoints, sample collection and handling procedures, toxicity testing protocols and environmental conditions, control responses, and analytical methods were consistent with established procedures [5–7,9]. In the case of the data sets from the NST, EMAP, and SEDQUAL sources, the standard protocols established under each program were evaluated, and individual studies were generally examined to identify possible deviations from these protocols. All the data that met the acceptance criteria were incorporated into the project database. Toxicity data were not included in the database if negative control survival was less than 85%. Data that were compiled in the database were verified against the original data source to ensure that project data quality objectives were met. For data that were acquired electronically (the majority of the data), a minimum of 10% of the data were compared to the source files. All data that were obtained from hard-copy materials (reports and journal articles) were compared to the source documents.

### *Data treatment and analysis*

To support subsequent data analysis, the total concentrations of PCB was calculated for each sediment sample represented in the database. In certain studies, only total concentrations of one or more of these substances were reported; in these cases, the reported values were used directly. The concentrations of total PCBs were determined using procedures that depended on the data reported in the original study. If the concentrations of Aroclors (e.g., Aroclor 1242, Aroclor 1248) were reported, then the concentrations of the individual Aroclors were summed to determine the concentration of total PCBs. When the concentrations of individual congeners were reported, these values were summed to determine the total PCB concentration. If fewer than 20 congeners were reported, the sum of the congeners was multiplied by 2, following the approach used by NST [10]. If both Aroclors and congeners were measured, total PCBs were based on the congener concentrations. In calculating the total PCB concentration, below-detection-limit values were treated as zero values. If all the individual chemicals to be summed were below detection or if the detection limit of any one nondetected chemical exceeded the sum of detected values, the highest detection limit of the chemical constituents for the sample was used as the total value and qualified as a below-detection-limit value.

Various methods have been used to designate individual sediment samples as toxic or nontoxic. In this study, individual

sediment samples were designated as toxic if the sample was statistically significant compared to the negative control and survival was less than 90% (the upper limit of the response level based on acceptable negative control response in 10-d marine amphipod toxicity tests [9]). We also evaluated a second commonly used approach for designating individual sediment samples as toxic, based on statistical significance compared to a negative control and a minimum significant difference (MSD) from the control (control adjusted survival of <80% [11]). The individual chemical models that were derived using the significance-only approach described previously are used for the analyses presented in this paper. These models consistently provided slightly better fits to the matching sediment chemistry and toxicity data.

The presence of multiple contaminants, many of which may be present at very low concentrations, frequently complicates evaluating the relationship between the concentration of an individual contaminant and toxicity in field-collected sediments. Consequently, the data for samples that were identified as toxic in this investigation were further screened before being used to develop the logistic models for each individual contaminant [5]. This screening process excluded toxic samples in which the selected contaminant was unlikely to contribute substantially to the observed toxicity. Following the general screening approach used by Ingersoll et al. [12] and similar to that used by others [1,7,13], the concentration of the selected chemical in each toxic sample was compared with the mean of the concentration of that substance in the nontoxic samples collected in the same study and geographic area. If the concentration of a chemical in an individual toxic sample was less than or equal to the mean concentration of that chemical in the nontoxic samples from that study area, it was considered unlikely that the observed toxicity could be attributed to that chemical. Therefore, these toxic samples were not included in the screened data set used for developing the logistic model for that chemical. All nontoxic samples were included in these analyses. Samples from reference stations were treated the same as other samples and included in the analysis. The data for chemical concentrations that were less than the reported detection limit were not used to develop the logistic models.

### *Concentration-interval plots*

Concentration-interval plots summarize the matching sediment chemistry and toxicity data for individual contaminants. These plots were prepared by calculating the proportion of toxic samples within discrete concentration intervals. The individual points represent the median of the sample concentrations within the interval and the fraction of the samples classified as toxic within the interval. Each point on the plots represents a minimum of 15 individual samples (a greater number of samples was included in the interval if more than one sample had the same concentration). The range represented by each concentration interval was determined from an ascending list of unique sample concentrations for each contaminant, with the number of intervals determined by the total number of unique sample concentrations for the selected contaminant.

### *Logistic regression modeling*

The individual chemical logistic regression models were developed from the screened data set for each chemical, according to the methods described in Field et al. [5]. The data for each chemical consist of the chemical concentration and the toxicity test result (toxic or nontoxic). The model parameters (slope, intercept) define the shape of relationship between the chemical concentration (log10) and probability of a toxic result. In its simplest form, the logistic model can be described using the following equation:

$$p = \frac{\exp(B_0 + B_1(x))}{1 + \exp(B_0 + B_1(x))}$$

where $p$ = probability of observing a toxic effect, $B_0$ = intercept parameter, $B_1$ = slope parameter, and $x$ = chemical concentration or log chemical concentration.

This logistic model was applied to the complete screened data (not the concentration-interval summarized data) for a number of substances to develop relationships between the sediment chemistry and the sediment toxicity data. For each of these substances, the intercept ($B_0$), slope ($B_1$), and chi-square statistic (−2 log likelihood) were determined. The data for each chemical were modeled independently rather than building a model that analyzed the concentrations of multiple chemicals simultaneously. Thus, only a single concentration variable ($x$) was used in each individual chemical model. All the logistic regression analyses were conducted using the Statistical Analyses System (SAS®) Institute's logistic procedure [14].

After estimating the model parameters, the model was inverted to estimate the concentrations that yield a certain response probability [5]. The notation $T_p$ (e.g., T50) is used to denote the concentration that would give a response of $p$ percent according to the model (e.g., the T50 represents the chemical concentration at which 50% of the samples would be predicted to be toxic). Confidence intervals for these effect concentrations were derived to describe the uncertainty associated with fitting the model.

The chi-square statistic provides useful information for interpreting the results of the logistic modeling. Specifically, the chi-square statistic was used to determine whether the slope parameter, $B_1$, was significantly different from zero. For all the models generated, the probability ($p$ value) associated with the slope parameter was <0.0001; therefore, the null hypothesis (slope = 0) can be rejected. Additionally, the chi-square statistic can be used to assess how well the model fits the data. Normalizing the chi-square statistic to the sample size ($N$) provides a goodness-of-fit measure that could be applied across all the data sets. Models that had a normalized chi-square value of <0.15 were considered a poor fit and were not used [5].

### *Multichemical models*

Individual chemical logistic regression models were combined to provide a single probability of observing toxicity using two approaches: the maximum probability model ($P_{Max}$), which was derived from the individual chemical model with the highest probability for a sample, and the average probability model ($P_{Avg}$), which was derived from the arithmetic mean of the probabilities for all the chemicals with models measured for a sample. The results for both approaches were plotted using the interval approach described earlier, with the difference that the x-axis for these plots represented the median of either the maximum or the mean probability (instead of the concentration) for each interval and included a minimum number of 50 samples per interval. The relationship between the maximum (or mean) probability of toxicity from the individual chemical models and the proportion of toxic samples was described by a least-squares binomial regression model of the

interval data. These binomial models were used to estimate the probability of toxicity for individual samples.

*Model evaluation*

We used three approaches to evaluate models with acceptable goodness-of-fit values (normalized chi-square value of <0.15). First, we evaluated the internal reliability of the chemical-specific models by comparing the model predictions of the probability of the toxicity to all the information contained in the project database on the toxicity of contaminated sediments to marine amphipods (the data that were screened out of the logistic model development process were included in the reliability evaluation). A similar approach was used to evaluate the reliability of the $P_{Max}$ model. Second, we compared spiked sediment bioassay median lethal concentration (LC50) values from the literature with individual model results. Third, we calculated the probability of observing toxicity for an independent data set (one not used to derive the models) using the $P_{Max}$ model. Model results were compared with toxicity test outcomes by comparing the proportion of toxicity test results observed with that predicted from the models.

## RESULTS

*Database composition*

The database is composed of matching sediment chemistry and toxicity data from the Atlantic, Gulf, and Pacific coasts of North America. The database includes data from 10-d toxicity tests with two species of amphipods (*R. abronius* and *A. abdita*), for which survival is the endpoint that was measured (Table 1). Overall, roughly 39% of the 3,223 sediment samples in the database with matching chemistry and toxicity were toxic to amphipods (percentage survival was <90% and significantly different from the negative control). For *A. abdita*, 24% of the 2,012 samples were toxic in 10-d tests (Table 1). A higher proportion of the samples (64% of 1,211 samples) tested with *R. abronius* were identified as toxic (Table 1). Using the MSD approach to classifying samples as toxic (percentage control-adjusted survival was <80% and significantly different from the negative control), 12.6% of the *A. abdita* samples and 40.8% of the *R. abronius* samples were classified as toxic.

The sediment toxicity database includes information on the concentrations of over 300 chemicals of potential concern at contaminated sediment sites. For many of these chemicals, the assembled data span a broad range of chemical concentrations. Table 2 presents the distributions of the chemistry data (10th, 50th, and 90th percentiles) for metals, PAHs, PCBs, and several organochlorine pesticides. These data show that the 10th- to 90th-percentile concentrations of the individual contaminants typically span two to three orders of magnitude, with ranges often spanning four to six orders of magnitude. Percentage total organic carbon in test sediments averaged 1.92% (standard deviation = 2.05, $n$ = 3,117) and ranged from 0.01 to 29.4%.

*Logistic regression models for individual chemicals*

We derived logistic regression models for individual chemicals to evaluate the relationships between chemical concentrations and sediment toxicity. Data to generate acceptable logistic models were available for 37 substances, including 10 trace metals, 22 individual PAHs, total PCBs, and four organochlorine pesticides (Table 3). In this paper, all the logistic models were generated using dry weight–normalized chemical concentration data because previous analyses indicated that such models fit the amphipod toxicity data as well or better than the organic carbon–normalized models for nonpolar organic contaminants [5]. The data for arsenic, chromium, nickel, and *p,p'*-DDE (dichlorodiphenyldichloroethylene) provided relatively poorer fits with the logistic model. Nevertheless, the normalized chi-square statistic exceeded 0.15 for 37 substances for which logistic models were generated, indicating that the models provide good fits of the amphipod toxicity data.

Concentration-interval plots provide additional information for evaluating the relationships between chemical concentration and the probability of observing sediment toxicity in the screened data set used to derive the model. For example, the plots for lead and mercury confirm that logistic models provide good fits of the underlying amphipod toxicity data (Fig. 1). Importantly, the range of concentrations represented in the database appears to span the effects range, as demonstrated by the low incidence of toxicity (0%) at the lowest concentrations and the high incidence of toxicity (90–100%) at the highest chemical concentrations. Similar results were obtained for fluoranthene and phenanthrene; however, the incidence of toxicity tended to be somewhat lower (roughly 90%) at the highest concentrations of these substances.

While the logistic models provide effective tools for estimating the probability of observing sediment toxicity at various chemical concentrations, point estimates of sediment effect concentrations are also useful for assessing sediment quality conditions. As an example, the chemical concentrations that corresponded to the 20, 50, and 80% proportion of toxic samples for amphipod survival were determined and designated as the T20, T50, and T80 values, respectively (Table 4). These values provide a framework for evaluating the reliability of the individual models.

*Reliability of individual chemical logistic models*

The reliability of the chemical-specific logistic models was evaluated by comparing the model predictions of the probability of the toxicity to all the information contained in the project database on the toxicity of contaminated sediments to amphipods. For example, the data screened out of the logistic model development process were included in the reliability evaluation. In this evaluation, the T values derived for each substance were used to define four ranges of chemical concentrations (≤T20, >T20–T50, >T50–T80, and >T80), and the percentage of samples that were toxic within each concentration range was determined (Table 5). The logistic models and associated point estimates were considered reliable if the observed incidence of toxicity was consistent with the predicted incidence of toxicity. In this evaluation, chemical concentrations below the T20 value were predicted to be associated with a low incidence of toxicity (<20%). Similarly, a high incidence of toxicity (>80%) was expected when chemical concentrations exceeded the T80 values. Moderately low (20–50%) and moderately high (50–80%) incidences of toxicity were expected at concentrations between the T20 and T50 values and between the T50 and T80 values, respectively.

The results of this evaluation indicate that the logistic models and associated point estimates of sediment effect concentrations generally provide a reliable basis for estimating the incidence of sediment toxicity in the project database (Table 5). The largest number of samples for each chemical had concentrations in the range below the T20 value, and the number of samples decreased within each of the subsequent ranges.

Table 2. Distribution of the sediment chemistry concentrations for the sediment samples with matching toxicity data

| Chemical | No. of samples | 10th percentile | 50th percentile | 90th percentile |
|---|---|---|---|---|
| Metals (mg/kg dry wt) | | | | |
| Antimony | 2,173 | 0.17 | 0.67 | 2.9 |
| Arsenic | 2,844 | 2.2 | 7.4 | 19 |
| Cadmium | 2,958 | 0.05 | 0.3 | 1.9 |
| Chromium | 2,827 | 9.1 | 50 | 130 |
| Copper | 3,091 | 2.6 | 26 | 160 |
| Lead | 3,010 | 5.5 | 23 | 130 |
| Mercury | 2,788 | 0.02 | 0.11 | 0.79 |
| Nickel | 2,916 | 2.4 | 19 | 44 |
| Silver | 2,552 | 0.03 | 0.21 | 1.9 |
| Zinc | 3,013 | 16 | 89 | 300 |
| Polycyclic aromatic hydrocarbons (μg/kg dry wt) | | | | |
| 1-Methylnaphthalene | 1,677 | 0.50 | 6.5 | 60 |
| 1-Methylphenanthrene | 1,697 | 0.30 | 11 | 130 |
| 2,6-Dimethylnaphthalene | 1,505 | 0.40 | 6.3 | 67 |
| 2-Methylnaphthalene | 2,077 | 0.77 | 12 | 130 |
| Acenaphthene | 1,795 | 0.20 | 7.0 | 130 |
| Acenaphthylene | 1,747 | 0.18 | 7.0 | 120 |
| Anthracene | 2,268 | 0.47 | 20 | 410 |
| Benz[*a*]anthracene | 2,574 | 1.2 | 40 | 760 |
| Benzo[*a*]pyrene | 2,526 | 1.2 | 53 | 910 |
| Benzo[*b*]fluoranthene | 1,645 | 0.86 | 48 | 1,000 |
| Benzo[*ghi*]perylene | 2,210 | 1.1 | 46 | 550 |
| Benzo[*k*]fluoranthene | 1,691 | 0.50 | 26 | 620 |
| Biphenyl | 1,507 | 0.47 | 6.8 | 54 |
| Chrysene | 2,650 | 1.7 | 53 | 1,000 |
| Dibenz[*a,h*]anthracene | 1,886 | 0.20 | 14 | 170 |
| Fluoranthene | 2,734 | 3.0 | 81 | 1,400 |
| Fluorene | 2,011 | 0.35 | 11 | 160 |
| Indeno[1,2,3-*cd*]pyrene | 2,212 | 0.94 | 47 | 600 |
| Naphthalene | 2,201 | 1.8 | 16 | 220 |
| Perylene | 2,174 | 1.5 | 36 | 370 |
| Phenanthrene | 2,688 | 1.7 | 44 | 660 |
| Pyrene | 2,768 | 3.2 | 87 | 1,500 |
| Polychlorinated biphenyls (PCBs; μg/kg dry wt) | | | | |
| Total PCBs | 1,989 | 2.0 | 39 | 640 |
| Organochlorine pesticides (μg/kg dry wt) | | | | |
| Dieldrin | 770 | 0.04 | 0.79 | 5.1 |
| *p,p'*-DDD[a] | 1,672 | 0.084 | 1.8 | 20 |
| *p,p'*-DDE[b] | 1,899 | 0.080 | 2.2 | 59 |
| *p,p'*-DDT[c] | 1,176 | 0.058 | 1.2 | 18 |

[a] DDD = dichlorodiphenyldichloroethane.
[b] DDE = dichlorodiphenyldichloroethylene.
[c] DDT = dichlorodiphenyltrichloroethane.

The incidence of amphipod toxicity was underestimated for concentrations below the T20 value for all 37 chemicals, although 35 of 37 were within 10% of the top of the range. Between the T20 and T50 values, the incidence of toxicity for most (30 of 37) of the chemicals was within the predicted range of 20 to 50% toxicity. The incidence of toxicity between the T50 and T80 values was within the predicted range 50 to 80% toxicity for all 37 chemicals. The observed incidence of toxicity was below the T80 value for 11 chemicals, but most were within 10% of the predicted range. The exceptions included arsenic and *p,p'*-DDE. The models for these chemicals would be most likely to overestimate toxicity.

Among the various classes of contaminants, the logistic models for PAHs were the most reliable. For 16 of 22 PAHs, the actual incidence of toxicity to amphipods was correctly predicted within three of the four concentration ranges defined by the T values; however, a higher-than-predicted incidence of toxicity was observed above the T20 values for all PAHs (Table 5). Among the trace metals, the logistic models for chromium, copper, lead, mercury, and zinc were the most reliable, as indicated by the level of agreement between the predicted and observed incidence of toxicity to amphipods. Likewise, the logistic model for total PCBs provided an accurate basis for predicting toxicity to amphipods in the database. A somewhat lower level of reliability was observed for the organochlorine pesticide models.

### *Individual chemical models and spiked-sediment bioassay LC50 values*

Dose–response data from laboratory-spiked sediment bioassays provide perspective on the concentrations of individual chemicals that can be considered to cause toxicity. Most of the studies in the literature on spiked-sediment toxicity report LC50s. An LC50 value represents the concentration corresponding to 50% survival of test organisms. In this study, many samples were classified as toxic with much higher test survival.

Table 3. Logistic regression model parameters, normalized chi-square values, and number of samples in the screened database for individual chemicals

| Chemical | No. of samples | Intercept ($B_0$) | Slope ($B_1$) | Chi-square value/*N* |
|---|---|---|---|---|
| Metals (mg/kg dry wt) | | | | |
| Antimony | 1,718 | −0.90 | 2.41 | 0.25 |
| Arsenic | 2,336 | −4.14 | 3.17 | 0.17 |
| Cadmium | 2,413 | −0.34 | 2.51 | 0.31 |
| Chromium | 2,399 | −6.44 | 3.00 | 0.20 |
| Copper | 2,580 | −5.79 | 2.93 | 0.38 |
| Lead | 2,481 | −5.45 | 2.77 | 0.27 |
| Mercury | 2,296 | 0.80 | 2.55 | 0.32 |
| Nickel | 2,450 | −4.61 | 2.77 | 0.18 |
| Silver | 2,103 | −0.11 | 1.97 | 0.25 |
| Zinc | 2,516 | −7.98 | 3.34 | 0.28 |
| Polycyclic aromatic hydrocarbons (μg/kg dry wt) | | | | |
| 1-Methylnaphthalene | 1,368 | −4.14 | 2.10 | 0.24 |
| 1-Methylphenanthrene | 1,401 | −3.59 | 1.75 | 0.28 |
| 2,6-Dimethylnaphthalene | 1,249 | −4.05 | 1.90 | 0.20 |
| 2-Methylnaphthalene | 1,704 | −3.76 | 1.78 | 0.25 |
| Acenaphthene | 1,424 | −3.62 | 1.75 | 0.33 |
| Acenaphthylene | 1,447 | −2.96 | 1.38 | 0.23 |
| Anthracene | 1,823 | −3.66 | 1.49 | 0.29 |
| Benz[*a*]anthracene | 2,099 | −4.20 | 1.58 | 0.30 |
| Benzo[*a*]pyrene | 2,053 | −4.30 | 1.58 | 0.30 |
| Benzo[*b*]fluoranthene | 1,348 | −4.54 | 1.49 | 0.27 |
| Benzo[*ghi*]perylene | 1,818 | −4.28 | 1.59 | 0.25 |
| Benzo[*k*]fluoranthene | 1,376 | −4.28 | 1.57 | 0.29 |
| Biphenyl | 1,226 | −4.11 | 2.21 | 0.26 |
| Chrysene | 2,126 | −4.32 | 1.54 | 0.29 |
| Dibenz[*a,h*]anthracene | 1,546 | −3.63 | 1.77 | 0.33 |
| Fluoranthene | 2,189 | −4.46 | 1.48 | 0.26 |
| Fluorene | 1,668 | −3.71 | 1.81 | 0.32 |
| Indeno[1,2,3-*cd*]pyrene | 1,837 | −4.37 | 1.62 | 0.27 |
| Naphthalene | 1,816 | −3.78 | 1.62 | 0.24 |
| Perylene | 1,823 | −4.68 | 1.76 | 0.22 |
| Phenanthrene | 2,173 | −4.46 | 1.68 | 0.30 |
| Pyrene | 2,240 | −4.71 | 1.59 | 0.29 |
| Polychlorinated biphenyls (PCBs; μg/kg dry wt) | | | | |
| Total PCBs | 1,617 | −3.46 | 1.35 | 0.27 |
| Organochlorine pesticides (μg/kg dry wt) | | | | |
| Dieldrin | 633 | −1.17 | 2.56 | 0.35 |
| *p,p'*-DDD[a] | 1,360 | −1.90 | 1.49 | 0.27 |
| *p,p'*-DDE[b] | 1,552 | −1.84 | 0.91 | 0.16 |
| *p,p'*-DDT[c] | 931 | −1.77 | 1.68 | 0.34 |

[a] DDD = dichlorodiphenyldichloroethane.
[b] DDE = dichlorodiphenyldichloroethylene.
[c] DDT = dichlorodiphenyltrichloroethane.

Thus, for comparisons with the individual chemical logistic regression models, LC20 values would be more consistent with the method used in this study to classify samples as toxic, but these values were not generally available. Consequently, reported LC50 values for 10-d spiked-sediment toxicity test conducted with marine amphipods were compared to the probability of toxicity estimated from the individual chemical models (Table 6). Using the logistic models, the probability of toxicity at the reported LC50 values ranged from 0.54 for zinc to 0.97 for mercury, with most estimates of the probability of toxicity within the range of 0.8 to 0.9.

### *Multichemical models*

The logistic regression models for individual chemicals were combined in two ways to estimate the probability of toxicity for the mixture of contaminants in a given sample:

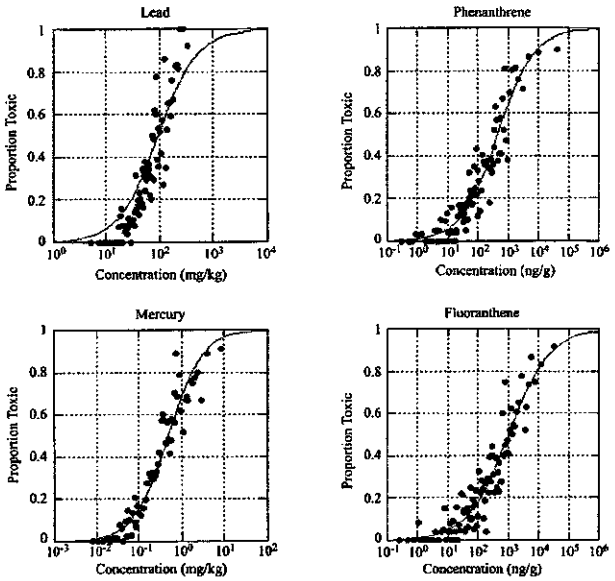


Fig. 1. Logistic regression models and proportion of samples toxic in concentration intervals in the screened database for lead (mg/kg), mercury (mg/kg), phenanthrene (ng/g), and fluoranthene (ng/g). The individual points represent the median of the sample concentrations within the interval and the fraction of the samples toxic within the interval.

using either the maximum ($P_{Max}$ model) or the mean ($P_{Avg}$ model) probabilities from the individual models (Fig. 2). The combined models are derived from the probability-interval plots, which summarize all the data in the database. In order to minimize the potential impact of samples with partial chemistry, only samples with at least 10 chemicals analyzed were included in the data set used to derive the multichemical models. Unlike the methods used for deriving the individual chemical models, all the samples with matching chemistry and toxicity were included in the evaluation (no additional data screening procedures were employed). The multichemical models are derived from the probability-interval plots, which summarize all of the data in the database. The parameter estimates shown in Table 3 were used to develop the multichemical models, except for PCBs. A correction in PCB units for 15 samples resulted in a minor change in the PCB model. However, because the effects of the correction on the multichemical models were extremely small (the maximum differences in predicted probability of toxicity were 0.0025 for the $P_{Max}$ model and 0.029 for the $P_{Avg}$ model), the multichemical models were not changed.

The results of this evaluation show that both combined models accurately predict toxicity to amphipods. Above a maximum probability of about 0.2, the maximum probability is somewhat higher than the corresponding proportion toxic (Fig. 2). The $P_{Max}$ model calibrates the maximum probability to account for the difference between the maximum probability from the individual chemical models and the observed proportion toxic for samples within the same probability interval. For example, for a maximum probability of 1.0 from the individual chemical models (x-axis), the observed proportion toxic (and the predicted probability from the $P_{Max}$ model) is 0.84. The data used to derive the $P_{Avg}$ model show the opposite situation, where mean probability values correspond to a higher proportion toxic (underestimate toxicity). Thus, using the

Table 4. Logistic model point estimates of T20, T50, and T80 concentrations and 95% confidence intervals (CI) for individual chemicals. The notation $T_p$ (e.g., T50) is used to denote the concentration that would give a response of $p$ percent according to the model (e.g., the probability that 50% of the samples would be toxic)

| | T20 | | | T50 | | | T80 | | |
|---|---|---|---|---|---|---|---|---|---|
| Chemical | Lower CI | T value | Upper CI | Lower CI | T value | Upper CI | Lower CI | T value | Upper CI |
| Metals (mg/kg dry wt) | | | | | | | | | |
| Antimony | 0.55 | 0.63 | 0.72 | 2.0 | 2.4 | 2.8 | 6.6 | 8.9 | 12 |
| Arsenic | 6.8 | 7.4 | 8.1 | 18 | 20 | 23 | 45 | 56 | 69 |
| Cadmium | 0.34 | 0.38 | 0.43 | 1.2 | 1.4 | 1.5 | 4.0 | 4.9 | 6.0 |
| Chromium | 44 | 49 | 53 | 126 | 141 | 158 | 329 | 410 | 510 |
| Copper | 29 | 32 | 35 | 86 | 94 | 103 | 239 | 280 | 328 |
| Lead | 27 | 30 | 33 | 84 | 94 | 104 | 244 | 297 | 360 |
| Mercury | 0.12 | 0.14 | 0.15 | 0.43 | 0.48 | 0.54 | 1.4 | 1.7 | 2.1 |
| Nickel | 13 | 15 | 16 | 42 | 47 | 52 | 118 | 147 | 185 |
| Silver | 0.19 | 0.23 | 0.26 | 0.98 | 1.1 | 1.3 | 4.4 | 5.8 | 7.6 |
| Zinc | 87 | 94 | 102 | 224 | 245 | 267 | 542 | 636 | 746 |
| Polycyclic aromatic hydrocarbons (μg/kg dry wt) | | | | | | | | | |
| 1-Methylnaphthalene | 17 | 21 | 25 | 73 | 94 | 122 | 281 | 433 | 669 |
| 1-Methylphenanthrene | 15 | 18 | 23 | 88 | 112 | 143 | 454 | 696 | 1,067 |
| 2,6-Dimethylnaphthalene | 20 | 25 | 31 | 96 | 133 | 185 | 413 | 713 | 1,231 |
| 2-Methylnaphthalene | 18 | 21 | 26 | 102 | 128 | 161 | 514 | 767 | 1,145 |
| Acenaphthene | 15 | 19 | 24 | 90 | 116 | 148 | 469 | 714 | 1,085 |
| Acenaphthylene | 11 | 14 | 18 | 102 | 140 | 194 | 799 | 1,418 | 2,517 |
| Anthracene | 27 | 34 | 42 | 228 | 290 | 369 | 1,630 | 2,486 | 3,792 |
| Benz[*a*]anthracene | 50 | 61 | 75 | 382 | 466 | 567 | 2,491 | 3,535 | 5,017 |
| Benzo[*a*]pyrene | 57 | 69 | 85 | 428 | 520 | 633 | 2,754 | 3,908 | 5,546 |
| Benzo[*b*]fluoranthene | 100 | 130 | 169 | 814 | 1,107 | 1,506 | 5,525 | 9,413 | 16,035 |
| Benzo[*ghi*]perylene | 54 | 67 | 82 | 395 | 497 | 625 | 2,444 | 3,710 | 5,631 |
| Benzo[*k*]fluoranthene | 55 | 70 | 90 | 405 | 537 | 713 | 2,541 | 4,121 | 6,685 |
| Biphenyl | 14 | 17 | 21 | 57 | 73 | 93 | 206 | 310 | 466 |
| Chrysene | 67 | 82 | 99 | 529 | 650 | 799 | 3,595 | 5,186 | 7,479 |
| Dibenz[*a,h*]anthracene | 15 | 19 | 23 | 92 | 113 | 139 | 475 | 685 | 988 |
| Fluoranthene | 98 | 119 | 146 | 832 | 1,034 | 1,284 | 6,066 | 8,952 | 13,212 |
| Fluorene | 16 | 19 | 24 | 92 | 114 | 140 | 465 | 665 | 951 |
| Indeno[1,2,3-*cd*]pyrene | 56 | 68 | 84 | 393 | 488 | 607 | 2,350 | 3,482 | 5,159 |
| Naphthalene | 25 | 30 | 37 | 170 | 217 | 278 | 1,022 | 1,569 | 2,409 |
| Perylene | 62 | 74 | 89 | 358 | 453 | 572 | 1,819 | 2,767 | 4,209 |
| Phenanthrene | 57 | 68 | 81 | 377 | 455 | 550 | 2,191 | 3,056 | 4,263 |
| Pyrene | 103 | 125 | 150 | 768 | 932 | 1,132 | 4,942 | 6,982 | 9,865 |
| Polychlorinated biphenyls (PCBs; μg/kg dry wt) | | | | | | | | | |
| Total PCBs | 27 | 35 | 44 | 282 | 368 | 481 | 2,412 | 3,926 | 6,393 |
| Organochlorine pesticides (μg/kg dry wt) | | | | | | | | | |
| Dieldrin | 0.65 | 0.83 | 1.0 | 2.3 | 2.9 | 3.6 | 6.9 | 10 | 15 |
| *p,p'*-DDD[a] | 1.7 | 2.2 | 2.8 | 14 | 19 | 25 | 95 | 159 | 267 |
| *p,p'*-DDE[b] | 2.2 | 3.1 | 4.4 | 61 | 103 | 176 | 1,278 | 3,414 | 9,119 |
| *p,p'*-DDT[c] | 1.3 | 1.7 | 2.2 | 8.3 | 11 | 15 | 45 | 76 | 129 |

[a] DDD = dichlorodiphenyldichloroethane.
[b] DDE = dichlorodiphenyldichloroethylene.
[c] DDT = dichlorodiphenyltrichloroethane.

$P_{Avg}$ model, mean probabilities of 0.5 and 0.75 correspond to proportion toxic of 0.7 and 0.9, respectively.

A major advantage of the $P_{Max}$ model is that it is less sensitive to the number of chemicals measured in each sample. For example, the $P_{Avg}$ model incorporates the output from models for 22 individual PAHs. Since individual PAHs are likely to co-occur in environmental samples, the $P_{Avg}$ may be influenced more by the concentrations of PAHs than by the concentrations of other chemicals. Although both the $P_{Max}$ and $P_{Avg}$ models provide a good fit to the data, we will discuss only the $P_{Max}$ model in the remainder of this paper.

The average predicted probability of toxicity (from the $P_{Max}$ model) within probability quartiles closely matches the incidence of toxicity within the same probability quartiles (Fig. 3), demonstrating the overall reliability of the $P_{Max}$ model within the database that was used to derive the model. In addition, the magnitude of the effect (decreased survival) in the amphipod test increases as the probability of toxicity increases (Fig. 4). Toxic samples with a probability of toxicity less than or equal to 0.25 have an average control-adjusted survival of greater than 75%, while samples with a probability of toxicity greater than 0.75 have an average control-adjusted survival of less than 50%. This demonstrates that samples that are estimated to have the highest probability of toxicity are also likely to be extremely toxic.

The number of chemicals in a sample that have a high probability of toxicity (e.g., $p > 0.75$) makes a difference in how well the model predictions match the observed percentage of the samples that are toxic (Fig. 5). As shown, when only one chemical in a sample has a probability of toxicity greater than 0.75, the $P_{Max}$ model tends to overestimate the probability of toxicity. The difference between the predicted probability of toxicity and incidence of toxicity can be considered to be a measure of the false-positive rate. As the number of chem-

Table 5. The percentage of toxic samples within ranges defined by logistic model T20, T50, and T80 values and the number of samples in the database used to derive the logistic model for each chemical

| Chemical | <T20 | T20–T50 | T50–T80 | >T80 | No. of samples |
|---|---|---|---|---|---|
| Metals (mg/kg dry wt) | | | | | |
| Antimony | 30.3 | 48.5 | 67.9 | 82.5 | 2,173 |
| Arsenic | 30.0 | 43.8 | 56.3 | 69.7 | 2,844 |
| Cadmium | 27.6 | 50.9 | 62.7 | 78.7 | 2,958 |
| Chromium | 24.5 | 42.7 | 55.7 | 80.0 | 2,827 |
| Copper | 22.1 | 50.6 | 64.9 | 85.0 | 3,091 |
| Lead | 28.5 | 45.0 | 60.6 | 90.0 | 3,010 |
| Mercury | 25.5 | 49.0 | 66.1 | 79.4 | 2,788 |
| Nickel | 25.3 | 44.7 | 60.4 | NA[a] | 2,916 |
| Silver | 25.6 | 52.5 | 60.7 | 73.7 | 2,552 |
| Zinc | 23.6 | 47.3 | 67.8 | 71.3 | 3,013 |
| Polycyclic aromatic hydrocarbons (μg/kg dry wt) | | | | | |
| 1-Methylnaphthalene | 23.7 | 48.3 | 60.3 | 75.0 | 1,677 |
| 1-Methylphenanthrene | 24.7 | 45.8 | 65.5 | 80.0 | 1,697 |
| 2,6-Dimethylnaphthalene | 23.5 | 42.2 | 57.4 | NA | 1,505 |
| 2-Methylnaphthalene | 25.4 | 47.5 | 61.1 | 88.0 | 2,077 |
| Acenaphthene | 25.2 | 50.3 | 67.7 | 91.4 | 1,795 |
| Acenaphthylene | 24.0 | 44.7 | 67.6 | NA | 1,747 |
| Anthracene | 26.5 | 48.8 | 66.9 | 77.1 | 2,268 |
| Benz[*a*]anthracene | 28.5 | 45.3 | 65.0 | 82.9 | 2,574 |
| Benzo[*a*]pyrene | 27.7 | 48.5 | 64.2 | 83.8 | 2,526 |
| Benzo[*b*]fluoranthene | 24.0 | 46.3 | 67.4 | NA | 1,645 |
| Benzo[*ghi*]perylene | 25.3 | 46.6 | 63.6 | 86.7 | 2,210 |
| Benzo[*k*]fluoranthene | 25.5 | 44.2 | 68.3 | 93.3 | 1,691 |
| Benzofluoranthenes, total | 25.2 | 46.3 | 56.7 | 83.3 | 1,507 |
| Biphenyl | 28.7 | 47.8 | 64.8 | 86.1 | 2,650 |
| Chrysene | 23.9 | 49.0 | 65.3 | 85.7 | 1,886 |
| Dibenz[*a,h*]anthracene | 28.4 | 47.1 | 64.9 | 87.9 | 2,734 |
| Fluoranthene | 22.1 | 48.4 | 68.2 | 87.2 | 2,011 |
| Fluorene | 25.6 | 44.5 | 64.9 | 90.5 | 2,212 |
| Indeno[1,2,3-*cd*]pyrene | 26.4 | 43.8 | 64.1 | 89.7 | 2,201 |
| Naphthalene | 26.9 | 39.4 | 59.8 | NA | 2,174 |
| Perylene | 28.2 | 47.3 | 64.1 | 85.7 | 2,688 |
| Phenanthrene | 28.3 | 46.2 | 65.0 | 87.2 | 2,768 |
| Pyrene | 23.7 | 48.3 | 60.3 | 75.0 | 1,677 |
| Polychlorinated biphenyls (PCBs; μg/kg dry wt) | | | | | |
| Total PCBs | 26.8 | 46.3 | 72.7 | 81.5 | 1,989 |
| Organochlorine pesticides (μg/kg dry wt) | | | | | |
| Dieldrin | 20.2 | 53.8 | 66.7 | 78.8 | 770 |
| *p,p'*-DDD[b] | 25.9 | 49.4 | 64.7 | 80.5 | 1,672 |
| *p,p'*-DDE[c] | 22.5 | 53.4 | 54.9 | 57.6 | 1,899 |
| *p,p'*-DDT[d] | 25.6 | 56.5 | 66.7 | 76.7 | 1,176 |

[a] NA = fewer than 10 samples.
[b] DDD = dichlorodiphenyldichloroethane.
[c] DDE = dichlorodiphenyldichloroethylene.
[d] DDT = dichlorodiphenyltrichloroethane.

Table 6. Estimated probability of toxicity from chemical-specific logistic regression models for median lethal concentration (LC50) values (dry wt) reported from 10-d spiked sediment amphipod toxicity tests

| Chemical | LC50 | Probability of toxicity | Source |
|---|---|---|---|
| Cadmium (mg/kg) | 9.81 | 0.90 | Mearns et al. [24] |
| | 8.8–10.0 | 0.88–0.90 | Kemp et al. [25] |
| | 8.2–11.5 | 0.88–0.91 | Robinson et al. [26] |
| | 6.9 | 0.85 | Swartz et al. [27] |
| Mercury (mg/kg) | 13.1 | 0.97 | Swartz et al. [17] |
| Zinc (mg/kg) | 276 | 0.54 | Swartz et al. [17] |
| Fluoranthene (mg/kg) | 4.2 | 0.71 | Swartz et al. [17] |
| | 3.3–10.5 | 0.68–0.82 | Swartz et al. [28] |
| Phenanthrene (mg/kg) | 3.68 | 0.82 | Swartz et al. [29] |
| Total PCBs[a] (mg/kg) | 8.8 | 0.87 | Swartz et al. [17] |
| *p,p'*-DDT[b] (ng/g) | 11.2–125 | 0.50–0.85 | Word et al. [30] |

[a] PCBs = polychlorinated biphenyls.
[b] DDT = dichlorodiphenyltrichloroethane.

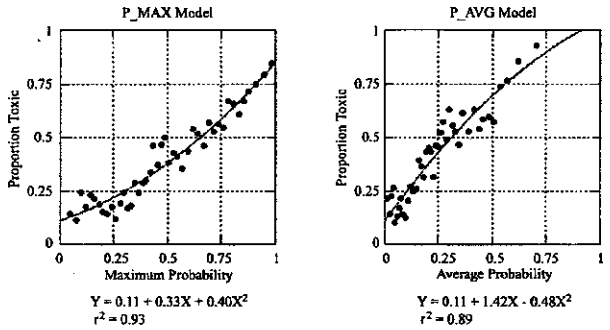


Fig. 2. Maximum ($P_{Max}$) and average ($P_{Avg}$) probability models (curves) and probability interval plots (points) where each point represents the median sample probability of a minimum of 50 individual samples within the interval and the fraction of the samples toxic within the interval.

icals in a sample with a high probability of toxicity increases, the false-positive rate decreases.

Application of the models to independent data (data not used in model derivation) is an important step in evaluation of the models. The $P_{Max}$ model was applied to a small independent data set ($n$ = 65) consisting of two studies from the Calcasieu Estuary in Louisiana, USA, that had matching sediment chemistry and toxicity data for *A. abdita* [16], unpublished data set provided electronically by P. Crocker, U.S. EPA, Region 6, Dallas, TX, USA]. These data were not included in the database used to derive the models. Although the data set has a limited number of samples, data from a wide range of contaminant concentrations are represented. The average predicted probability of toxicity within probability quartiles was within 25% of the measured incidence of toxicity for each quartile, indicating that the $P_{Max}$ model generally was able to successfully classify the samples as toxic or not toxic (Fig. 6). With the exception of the third quartile ($0.5 < p < 0.75$), the $P_{Max}$ model underestimated the toxicity observed in Calcasieu Estuary samples.

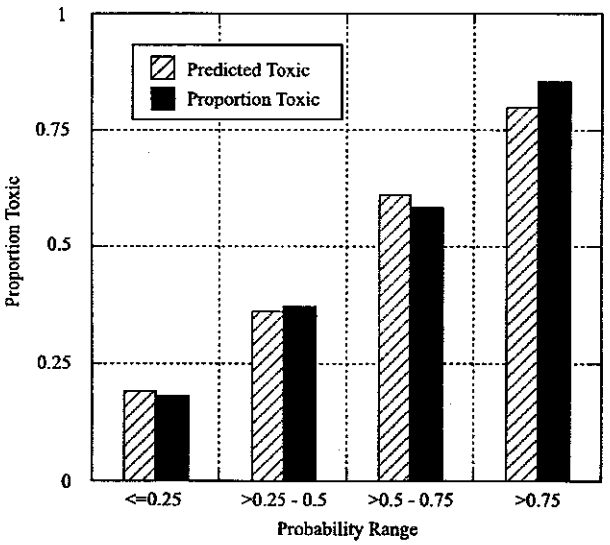


Fig. 3. Average predicted and proportion toxic within probability quartiles for $P_{Max}$ model.

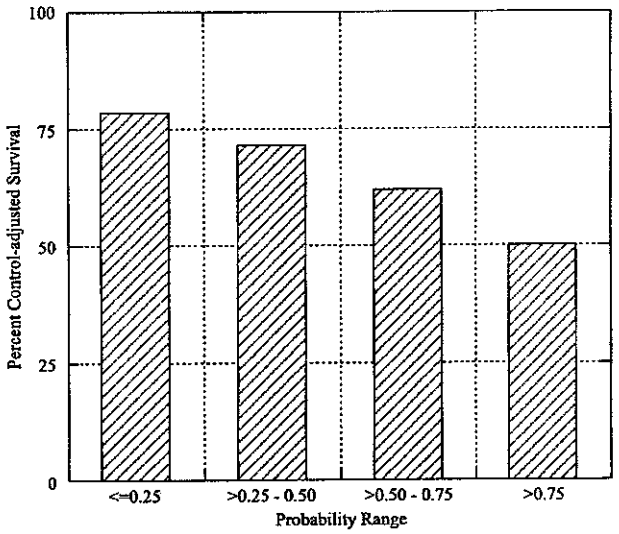


Fig. 4. Percentage control-adjusted survival for toxic samples within probability quartiles for $P_{Max}$ model.

## DISCUSSION

### *Evaluation of the logistic regression modeling approach*

The results of this investigation indicate that logistic regression models provide an effective basis for describing relationships between the concentrations of sediment-associated contaminants and toxicity to two species of marine amphipods. The chemical-specific models that were derived in this investigation provide a basis for estimating the proportion of samples expected to be toxic for 37 individual contaminants over a wide range of contaminant concentrations. As such, these models help users select sediment effect concentrations that most directly meet the needs of their specific application. For

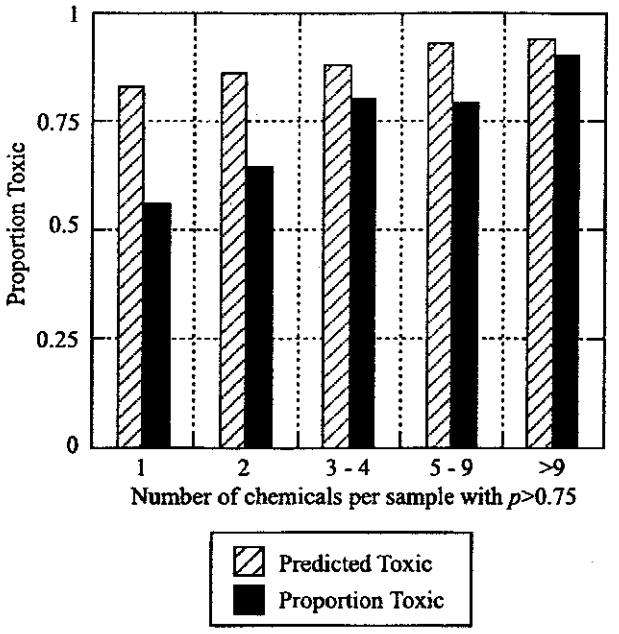


Fig. 5. Average predicted and proportion toxic by number of chemicals per sample with probability of toxicity from individual chemical models >0.75.

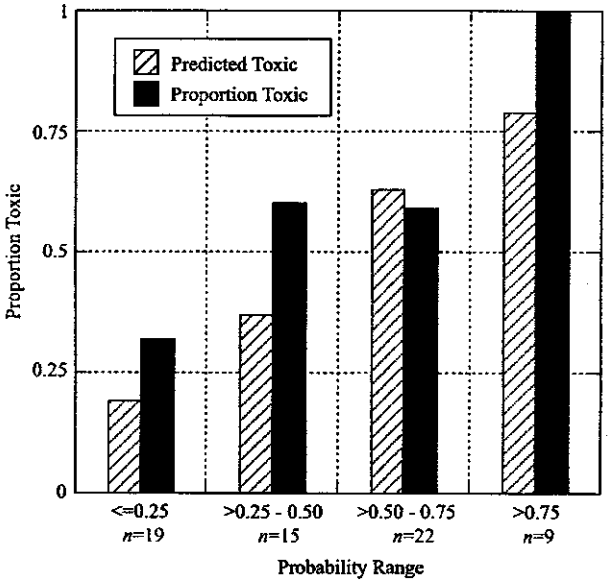


Fig. 6. Average predicted and proportion toxic within probability quartiles for an independent data set from the Calcasieu Estuary (LA, USA) ($n$ = 65).

example, T10, T15, or T20 values could be calculated and used to identify concentrations for individual contaminants that are likely to be associated with a relatively low incidence of sediment toxicity (10, 15, or 20%, respectively). Such point estimates of minimal effect concentrations might be used in a screening assessment to identify sediments that are relatively uncontaminated and have a low probability of sediment toxicity. Similarly, contaminant concentrations for which there is a high probability of observing adverse effects could be estimated by calculating T70, T80, or T90 values. These higher point estimates could be used to identify sediments that are highly likely to be toxic to amphipods and have a greater magnitude of effect (higher percentage mortality). The T values can be used in much the same way as other sediment guidelines, with the difference that the T value is associated with a specific probability of observing toxicity and an estimate of variance based on the fit of the model.

The individual models derived in this study have lower T50 values for all the seven chemicals that were modeled in our earlier study [5]. The most likely explanations for this difference include standardization of the approach used to classify samples as toxic and the large increase in the size of the database. In the previous study, toxicity was classified by the original investigators. For example, in the earlier study, samples from the Puget Sound database were classified as toxic if significantly different from a field reference and less than 75% survival; in the present study, these samples were classified as toxic if significantly different from the negative control and less than 90% survival. This change resulted in a greater number of samples classified as toxic in data from Puget Sound than in the earlier study.

The $P_{Max}$ model is based on the individual chemical with the highest probability of toxicity. For approximately 70% of the samples, one of the 10 metals for which individual regression models had been developed had the maximum probability used in the $P_{Max}$ model. This should not be construed to imply that metals are causing toxicity in these samples. It indicates only that metals appear to be a good predictor of toxicity in field-collected samples where mixtures of contaminants are likely to be present. It is not possible to determine from the models alone whether the metals or the other chemicals considered in this evaluation make a substantial contribution to the observed toxicity in any individual sample.

Comparison of the model results to spiked-sediment bioassays reported in the literature indicates that LC50 values are equivalent to T80 to T90 values for several chemicals. This is consistent with the observation that control-adjusted survival is approximately 50% in samples with a high probability of toxicity. It may be more appropriate to use LC15 or LC20 values for comparisons to the chemical-specific models since the models are based on whether samples are classified as toxic, which does not require 50% mortality. Unfortunately, the LC15 or LC20 values are rarely reported in the literature.

### *Estimating the probability of toxicity for sediment quality guidelines*

The individual chemical logistic models can be used to estimate the probability of observing toxicity to amphipods at the chemical concentrations that are defined by the SQGs. Examples are shown in Table 7 for three commonly used sets of SQGs that represent a range of threshold values: threshold effect levels (TEL), probable effect levels (PEL [7]; effect range low [ERL] and effect range median [ERM] [13]), and apparent effect thresholds (AET) for marine amphipods [15]. Both ERLs and TELs represent chemical concentrations below which toxicity would occur infrequently (<25%) [1,7], while effects are expected to be frequently observed at concentrations exceeding PEL and ERM concentrations. In contrast, endpoint-specific AET values represent concentrations above which toxicity is always expected for that endpoint.

The results are generally consistent with the narrative intent of the SQGs for most of the chemicals for which SQGs had been derived. For the TELs, the probability of observing sediment toxicity at these concentrations ranged from 10 to 41% (depending on the chemical under consideration), with the probability of toxicity below 25% for 24 of 27 chemicals considered (Table 7). The probability of observing sediment toxicity was generally higher at the ERL concentrations (ranging from 11–47%), with a median value of 33%. The predicted incidence of toxicity was higher for the PEL and ERM values, with median values of 55 and 72%, respectively. The highest probability of observing toxicity to amphipods was noted for the amphipod AETs, with the estimated proportion of the toxic samples ranging from 45 to 99% and the median value of 90%.

Although derivation methods for the different SQGs are well described and are consistent for all the chemicals within a given type of SQG, no straightforward method exists that enables the user to either evaluate the degree to which individual SQGs meet their objectives or evaluate the reliability of individual SQGs. The logistic model approach provides a way to put the individual SQG values into perspective by estimating the probability of toxicity to amphipods. In addition, the goodness of fit for each model provides an objective measure of the quality of the models for individual chemicals.

### *Multichemical models*

One of the major challenges in assessing the ecological risk associated with exposure to contaminated sediments is the presence of chemical mixtures. Swartz et al. [17] demonstrated that mixtures of two to four contaminants produced greater toxicity to a marine amphipod than the individual chemicals

Table 7. Estimated proportion of samples toxic to amphipods at the chemical concentrations defined by sediment quality guidelines (SQG)

| Chemical | ERL[a] | ERM[b] | TEL[c] | PEL[d] | AET[e] |
|---|---|---|---|---|---|
| Metals | | | | | |
| Antimony | NA[f] | NA | NA | NA | 0.99 |
| Arsenic | 0.22 | 0.85 | 0.20 | 0.73 | 0.99 |
| Cadmium | 0.46 | 0.89 | 0.32 | 0.77 | 0.93 |
| Chromium (total) | 0.33 | 0.78 | 0.22 | 0.54 | 0.94 |
| Copper | 0.21 | 0.79 | 0.11 | 0.54 | 0.97 |
| Lead | 0.30 | 0.73 | 0.20 | 0.55 | 0.96 |
| Mercury | 0.22 | 0.60 | 0.19 | 0.60 | 0.85 |
| Nickel | 0.28 | 0.53 | 0.22 | 0.48 | 0.92 |
| Silver | 0.47 | 0.73 | 0.41 | 0.59 | 0.81 |
| Zinc | 0.33 | 0.68 | 0.27 | 0.54 | 0.98 |
| Polycyclic aromatic hydrocarbons | | | | | |
| 2-Methylnaphthalene | 0.39 | 0.78 | 0.19 | 0.59 | 0.89 |
| Acenaphthene | 0.18 | 0.75 | 0.10 | 0.45 | 0.90 |
| Acenaphthylene | 0.33 | 0.71 | 0.13 | 0.49 | 0.79 |
| Anthracene | 0.31 | 0.70 | 0.24 | 0.47 | 0.92 |
| Benz[*a*]anthracene | 0.40 | 0.70 | 0.22 | 0.57 | 0.84 |
| Benzo[*a*]pyrene | 0.47 | 0.68 | 0.23 | 0.57 | 0.79 |
| Benzo[*ghi*]perylene | NA | NA | NA | NA | 0.78 |
| Chrysene | 0.41 | 0.73 | 0.23 | 0.54 | 0.91 |
| Dibenz[*a,h*]anthracene | 0.39 | 0.66 | 0.10 | 0.53 | 0.90 |
| Fluoranthene | 0.41 | 0.74 | 0.19 | 0.56 | 0.90 |
| Fluorene | 0.20 | 0.77 | 0.21 | 0.55 | 0.94 |
| Indeno[1,2,3-*cd*]pyrene | NA | NA | NA | NA | 0.83 |
| Naphthalene | 0.45 | 0.83 | 0.22 | 0.60 | 0.84 |
| Phenanthrene | 0.39 | 0.70 | 0.23 | 0.53 | 0.94 |
| Pyrene | 0.44 | 0.67 | 0.22 | 0.57 | 0.88 |
| Polychlorinated biphenyls (PCBs) | | | | | |
| Total PCBs | 0.16 | 0.40 | 0.16 | 0.40 | 0.78 |
| Organochlorine pesticides | | | | | |
| Dieldrin | NA | NA | 0.18 | 0.61 | 0.55 |
| *p,p'*-DDD[g] | NA | NA | 0.15 | 0.36 | 0.69 |
| *p,p'*-DDE[h] | 0.18 | 0.37 | 0.17 | 0.62 | 0.45 |
| *p,p'*-DDT[i] | NA | NA | 0.16 | 0.35 | 0.91 |

[a] ERL = effect range low [1].
[b] ERM = effect range median [1].
[c] TEL = threshold effect level [7].
[d] PEL = probable effect level [7].
[e] AET = apparent effect threshold for amphipod survival [15].
[f] NA = no SQG value available.
[g] DDD = dichlorodiphenyldichloroethane.
[h] DDE = dichlorodiphenyldichloroethylene.
[i] DDT = dichlorodiphenyltrichloroethane.

alone. Most evaluations of the effects of mixtures on aquatic toxicity endpoints such as survival and growth have focused on two empirical models of noninteractive joint action: concentration addition and response addition [18]. Concentration addition, which is also referred to as simple similar action, assumes that contaminants act independently but by similar mode of action. Toxic unit models, which are a specialized case of concentration addition, have been applied to the assessment of the toxicity of PAH mixtures in sediment [19–21] but are unlikely to be applicable to complex mixtures of contaminants commonly found in the environment that may represent different modes of action. Response addition, or independent action, is expected to apply to cases where contaminants have a different mode of action and toxicity would be predicted only when one or more contaminants exceeds their toxicity threshold.

The $P_{Max}$ model can be considered to be similar to a response addition model, where toxicity is predicted on the basis of the individual chemical model that has the highest probability of toxicity. However, because the individual models themselves were derived from field-collected sediments that include mixtures of contaminants rather than individual dose–response relationships, to some extent the individual models incorporate the overall toxicity of the mixtures.

The multichemical models provide an estimate of toxicity for individual samples based on the output from the individual chemical models. The $P_{Max}$ model, which is based on the highest predicted probability from any of the individual chemical models, is less sensitive to the number of chemicals analyzed than a model based on the mean value ($P_{Avg}$ model). However, because the predicted probability of toxicity is based on a single chemical, a greater potential may exist for false positives because of the application of less reliable individual models for some chemicals or unusual conditions. The individual models for nickel, *p,p'*-DDE, and a few other chemicals were shown (Table 5) to have a lower incidence of toxicity at concentrations exceeding T80 values. Thus, the probability of toxicity could be overestimated for any sample where the chemical with the maximum probability value had a high rate of false positives. In addition, a greater tendency exists for false positives when high probability predictions are based on only one or two chemicals having a high probability of toxicity. These results are similar to the analysis presented by Long et al. [22] that demonstrated an increasing incidence of toxicity as the number of ERMs or PELs exceeded in individual samples increased. This supports the concept that empirical approaches, such as the one described in this paper, are not defining dose–response relationships for individual chemicals but serve as indicators of toxicity based on chemical mixtures.

### *Application of models to evaluations of site-specific or regional data*

Hazardous waste-site evaluations often involve the collection of substantial quantities of sediment chemistry data. For example, information on the magnitude and areal extent of sediment contamination is frequently collected to support screening-level ecological risk assessments. In the past, sediment assessors have used numerical SQGs (e.g., ERLs, ERMs, TELs, PELs, AETs, and/or others) to evaluate such data. While such SQGs are useful for identifying thresholds below which sediment toxicity is unlikely to be observed and above which sediment toxicity is likely to occur, it is difficult to determine the extent to which risk increases with the magnitude of exceedance of the SQGs. Calculation of hazard quotients (HQ, the ratio of the measured concentration of a contaminant in sediments to the corresponding toxicity threshold) for each chemical using the SQGs can provide additional information for assessing risk to sediment-dwelling organisms. Several investigators have applied mean SQG quotients to evaluate mixtures of contaminants in field-collected sediment samples [8,22,23]. However, such evaluations are based on an assumption that concentration–response relationships for each chemical are similar. The logistic regression models approach avoids this assumption and provides a way to apply separate concentration response relationships for each chemical.

The models described in this paper were derived from a large database of matching sediment chemistry and toxicity that included data from many different coastal areas of North America and many different contamination gradients. Because the models require a large amount of data for their derivation, data from an individual site are rarely sufficient to derive site-specific models. Rather than deriving site- or regional-specific models, we recommend applying the models to data from the

site or region to determine how well the models fit the local data. The evaluation of the independent data set from the Calcasieu Estuary provides an example of how this could be accomplished. By comparing the percentage of toxic samples with the mean predicted probability of toxicity within discrete probability ranges (e.g., probability quartiles as shown in Fig. 6), the performance of the models with data from the site can be evaluated. If the models predict a higher percentage of toxic samples than observed (false positives), then issues related to bioavailability may be investigated further. The individual chemical models could be used to determine whether specific chemical models are associated with the high false-positive rate. If toxicity occurs at a much higher frequency than predicted (false negatives), then it may be important to consider chemicals not accounted for (no models available) or issues related to the sediment matrix (e.g., grain size effects).

## CONCLUSIONS

A large database of matching sediment chemistry and toxicity data for marine amphipod survival, which includes many different contaminant gradients from a wide variety of habitats in coastal North America, was used to derive logistic regression models for 37 individual chemicals. The logistic regression models do not represent dose–response relationships for individual chemicals but can be considered to be indicators of toxicity based on field-collected sediment chemical mixtures. Combining the individual models into a single model, using either the maximum or the average probability predicted from the chemicals analyzed in a sample, provides a single value for estimating the probability that a sample will be toxic. These models enable users to select sediment quality guidelines that match the level of protectiveness (as measured by the probability of toxicity) appropriate for the objectives of their assessment and to estimate the uncertainty associated with the chosen level of protectiveness.

The LRM approach provides a useful framework for conducting screening-level assessments that require classifying or prioritizing samples on the basis of sediment chemistry but should not be considered as a substitute for direct effects assessment (e.g., toxicity tests). Because the models do not consider potential differences in bioavailability or exposure, the probability of toxicity may be over- or underestimated for some locations. Application of the models to additional independent data sets will provide the best measure of the ability of the models to predict toxicity in different environments and contaminant gradients.

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# AN EVALUATION OF CAUSE–EFFECT RELATIONSHIPS BETWEEN POLYCHLORINATED BIPHENYL CONCENTRATIONS AND SEDIMENT TOXICITY TO BENTHIC INVERTEBRATES

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**Abstract**—Cause–effect sediment-quality benchmarks for the protection of benthic invertebrates are needed for polychlorinated biphenyls (PCBs) to support predictive risk assessments and retrospective evaluations of the causes of observed sediment toxicity. An in-depth evaluation of PCB aquatic toxicity and organic carbon partitioning was conducted to predict sediment effect concentrations using the equilibrium partitioning (EqP) approach. This evaluation was limited to invertebrate toxicity data, because PCBs may exert toxicity to invertebrates and fish via different toxicological mechanisms. As a result of differences in organic carbon partitioning among PCBs of differing levels of chlorination, the estimated EqP benchmarks increase with increasing degree of chlorination for various commercial and environmental PCB mixtures. Studies of spiked sediment toxicity using PCBs were reviewed, and their results generally were consistent with EqP predictions. Additionally, toxicity and benthic community data were reviewed for eight PCB-contaminated sites; these data also showed agreement with EqP predictions. None of these lines of evidence supports previously proposed, empirical sediment-quality guidelines for PCBs. Reasons for the lack of agreement between cause–effect and association-based benchmarks are discussed, and areas of future research to further refine EqP predictions for PCBs are identified.

**Keywords**—Polychlorinated biphenyls Sediment toxicity Benthic invertebrates Equilibrium partitioning Sediment-quality guidelines

## INTRODUCTION

The development of cause–effect sediment-quality benchmarks to assess potential effects on benthic invertebrates from polychlorinated biphenyls (PCBs) has lagged behind similar efforts regarding polycyclic aromatic hydrocarbons and metals [1,2]. This can be attributed, in part, to a focus on PCB bioaccumulation and toxicity to piscivorous wildlife, arguably the most sensitive endpoint for PCB-related effects. However, in our experience, cause–effect benchmarks for PCBs are needed specifically for benthic invertebrates to support both predictive ecological risk assessments and retrospective evaluations of the causes of observed sediment toxicity. For example, in smaller bodies of water, PCB bioaccumulation is limited because of the lower trophic status of the aquatic community, and wildlife exposure also may be limited by the size of the affected area relative to overall wildlife foraging ranges. In such cases, benthic invertebrates often are the most exposed organisms, and their protection may determine cleanup decisions. Because the assessment and remediation of even small sites with contaminated sediment can cost millions of dollars and involve significant disruption of physical habitat, it would be valuable to improve the efficiency and accuracy of PCB risk assessments for benthic invertebrates by better understanding expected concentration–response relationships. Similarly, improved methods are needed to support identification of causative agents in sediments that are found to be toxic to benthic invertebrates. The consequences of incorrectly identifying causes of toxicity include failure to target source control efforts appropriately (e.g., resulting in recontamination of remediated areas) as well as potential misallocation of liability.

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Most sediment-quality guidelines for PCBs have been based on interpretations of large data sets of paired biological and chemistry data collected from sites contaminated primarily with chemicals other than PCBs [3–14]. These empirical guidelines were developed using methods that cannot discriminate between noncausal associations and cause–effect, concentration–response relationships [15]. MacDonald et al. [16] proposed "consensus" sediment effect concentrations for PCBs based on a review and synthesis of such guidelines, including a threshold value of 0.040 mg/kg, a midrange value of 0.40 mg/kg, and an extreme value of 1.7 mg/kg. Increasingly, the guidelines of MacDonald et al. [16] are being used by sediment assessors and regulatory agencies, and in the present paper, they serve as a focus for comparisons to empirical PCB sediment guidelines in general. It is worth noting that the same methods used to derive consensus guidelines for PCBs also have been applied to metals [17], resulting in "threshold effect concentrations" that are at or below naturally occurring background concentrations of metals in sediments and, thus, obviously underestimate plausible, individual-chemical toxicity thresholds [18].

MacDonald et al. [16] tried to demonstrate that their consensus PCB guidelines reflect cause–effect relationships between PCB concentrations and benthic toxicity by comparing the consensus values to equilibrium partitioning (EqP) [19] benchmarks and results of spiked sediment toxicity tests. However, in their review, the sole EqP benchmark identified [20] is actually based on the U.S. Environmental Protection Agency (U.S. EPA) water-quality criterion of 0.014 μg/L. This criterion is not relevant to the prediction of benthic invertebrate toxicity, because it is calculated from a fish tissue PCB concentration for the protection of wildlife [21]. Also, MacDonald et al. [16], in their evaluation of the results of spiked sediment

tests, did not account for biases resulting from limited sediment–chemical contact time. Spiked sediment studies are capable of characterizing cause–effect relationships provided that the bioavailability of the spiked chemical is understood and can be compared to other sediments. Current guidance for sediment spiking recommends a stabilization period to allow partitioning (i.e., bioavailability) of the spiked chemical to approach equilibrium [22]; otherwise, chemical bioavailability and toxicity will tend to be overestimated compared to field conditions [23]. Recent PCB-spiked sediment studies incorporating a sediment stabilization period were not included in the review by MacDonald et al. [16].

In the present paper, we endeavor to provide a more thorough causal assessment of PCB-related risks to benthic invertebrates, including the application of EqP methods using aquatic life toxicity values and a review of spiked sediment toxicity studies. Measures of biological effects for sites contaminated primarily with PCBs are then compared with both the consensus PCB guidelines and the cause–effect benchmarks to evaluate the predictive ability of these screening values. Uncertainties in our assessment also are discussed, both to guide its application and to identify future research needs.

## EQUILIBRIUM PARTITIONING ASSESSMENT

### *Overview*

The EqP approach [19] uses the mass fraction of organic carbon in sediment ($f_{oc}$) and the chemical-specific partition coefficient between water and organic carbon ($K_{oc}$) to calculate sediment-quality benchmarks as follows:

$$\text{sediment-quality benchmark} = \text{water-quality benchmark} \cdot K_{oc} \cdot f_{oc} \quad (1)$$

This approach is based on observations indicating that the bioavailable fraction of nonpolar organic chemicals is equivalent to the fraction of the sediment concentration that is freely dissolved in interstitial water, and that the freely dissolved fraction is determined primarily by the extent of partitioning to organic carbon. The $K_{oc}$ parameter typically is estimated based on octanol–water partition coefficient ($K_{ow}$) values. Advantages of the EqP approach include the ability to incorporate extensive aquatic toxicity data from controlled experiments and the ability to interpret differences in bioavailability among different sediments. The applicability of the EqP approach to nonionic organic chemicals has been extensively validated (see, e.g., [1,19,24,25]).

Application of the EqP approach to PCBs is complicated by the occurrence in sediment of as many as 209 individual PCB congeners, representing up to 10 levels of chlorination (mono- through decachlorobiphenyl homologues). Analyses of all possible congeners are uncommon in both environmental characterization and toxicity studies. Rather, PCB concentrations typically are measured as total PCBs (quantified using technical Aroclor standards) or as homologues. Thus, toxicity thresholds and chemical partitioning properties identified for Equation 1 must apply to PCB mixtures. Toward that end, we developed an EqP model for PCBs, as summarized in Figure 1. In the absence of adequate toxicity data for PCB homologues or congeners, the aquatic toxicity component is addressed by identifying a single water-quality benchmark applicable to total PCB concentrations in pore water. Chemical partitioning is addressed on a homologue basis to account for site-specific differences in PCB mixture composition. Normalization of

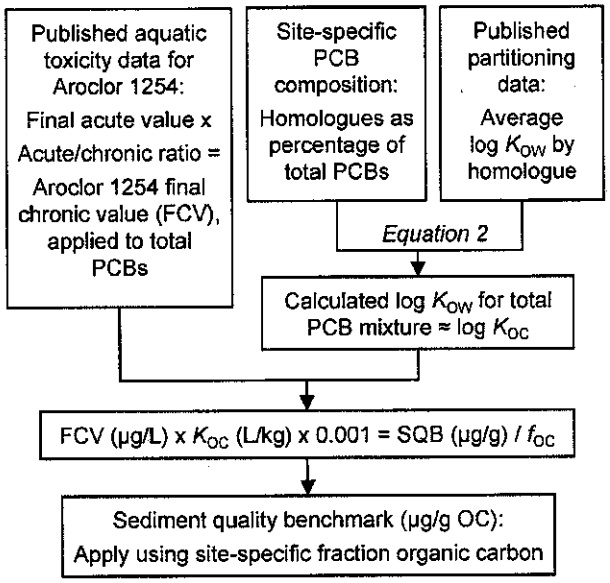


Fig. 1. Conceptual overview of equilibrium partitioning model for polychlorinated biphenyls (PCBs). Note that site-specific PCB composition is considered in estimating partitioning, but not toxicity, because of limitations in available toxicity data. $f_{oc}$ = fraction organic carbon; $K_{oc}$ = organic carbon–water partition coefficient; $K_{ow}$ = octanol–water partition coefficient; SQB = sediment-quality benchmark.

sediment PCB concentrations based on sample-specific organic carbon measurements is assumed by expressing sediment-quality benchmarks on a microgram per gram of organic carbon (µg/g OC) basis.

### *Water-quality benchmark derivation*

Because the available water-quality criteria for PCBs are based on piscivorous wildlife and human health endpoints, it was necessary to derive an alternative water-quality benchmark for this assessment based on aquatic toxicity data. A PCB water-quality benchmark was developed specifically for invertebrates based primarily on data obtained from the U.S. EPA online AQUIRE (Aquatic Toxicity Information Retrieval) database. Original studies were obtained and reviewed to the extent possible. The complete data set is available in the *Supplemental Information* (see Supplemental Tables 1 and 2 and supplemental references; SETAC Supplemental Data Archive, Items ETC-25-10-002 to ETC-25-10-004, http://etc.allenpress.com). Although the EqP approach typically uses aquatic toxicity data for both invertebrates and fish, the toxic mechanism of action exerted by PCBs may differ significantly between invertebrates and fish. Specifically, toxicity of planar PCB congeners to vertebrates is mediated by the aryl hydrocarbon (Ah) receptor and may be more potent (at least for some vertebrates) than the toxicity from other PCB congeners. Invertebrates generally lack the Ah receptor and are not susceptible to this mechanism of toxicity [26,27]. Although acute toxicity to both invertebrates and fish likely operates via narcosis, chronic toxicity in fish may be mediated by the Ah receptor, at least in some cases. Therefore, only invertebrate toxicity data are considered for the present evaluation.

Acute toxicity values, representing concentrations lethal to 50% of test organisms (LC50s), are available primarily for the commercial PCB mixtures Aroclor 1254 (21 species) and Aroclor 1242 (nine species). Although individual species exhibit

Table 1. Derivation of Aroclor 1254 final acute value for invertebrates

| Scientific name | Common name | Reference | Method[a] | Test duration (h) | 50% Lethal concn. (μg/L) | Genus mean acute value |
|---|---|---|---|---|---|---|
| *Palaemonetes pugio* (adult)[b] | Daggerblade | [75] | S, M | 96 | 41–86 | 6.9 |
| *Palaemonetes pugio* (juvenile) | Grass shrimp | | S, M | 96 | 6.1 | |
| *Palaemonetes pugio* (juvenile) | | | S, M | 96 | 7.8 | |
| *Corophium insidiosum* | Amphipod | [76][c] | NR, U | 96 | 9 | 9 |
| *Uca pugilator* | Fiddler crab | [77] | R, U | 96 | 10 | 10 |
| *Crangon septemspinosa* | Bay shrimp | [78] | S, M | 96 | 12 | 12 |
| *Crangon crangon* | Common shrimp | [29][c] | R, U | 48 | 3,000–10,000[d] | |
| *Ampelisca abdita* | Amphipod | [57] | S, M | 96 | 40 | 40 |
| *Elasmopus bampo* | Amphipod | [76][c] | NR, U | 96 | 40 | 40 |
| *Mysidopsis bahia* | Opossum shrimp | [57] | S, M | 96 | 57 | 57 |
| *Orconectes nais* | Crayfish | [79] | S, U | 96 | 100 | 100 |
| *Ischnura verticalis* | Damselfly | [80] | S, U | 96 | 200 | 200 |
| *Macromia* sp. | Dragonfly | [81][c] | F, U | 96 | 200 | 200 |
| *Procambarus* sp. | Crayfish | [79] | S, U | 96 | >550 | 550 |
| *Capitella capitata* | Polychaete worm | [82][c] | S, U | 96 | >1,000 | 1,000 |
| *Neanthes grubei* | Polychaete worm | [82][c] | S, U | 96 | >1,000 | 1,000 |
| *Nereis arenaceodentata* | Polychaete worm | [82][c] | S, U | 96 | >1,000 | 1,000 |
| *Ophryotrocha labronica* | Polychaete worm | [82][c] | S, U | 96 | >1,000 | 1,000 |
| *Litopenaeus vannamei* | White shrimp | [83] | S, U | 48 | 1,640 | 1,640 |
| *Gammarus fasciatus* | Amphipod | [84] | S, U | 96 | 2,400 | 2,400 |
| *Gammarus pseudolimnaeus* | Amphipod | [80] | S, NR | 96 | 2,400 | |
| *Cerastoderma edule* | Cockle | [29][c] | R, U | 48 | >10,000 | 10,000 |
| *Hydra oligactis* | Hydra | [85] | R, U | 72 | 20,000–22,000 | 21,000 |
| Final acute value[e] | | | | 6.96 μg/L | | |

[a] F = flow-through; M = measured; NR = not reported; S = static; R = renewal; U = unmeasured.
[b] Genus mean acute value for *Palaemonetes pugio* is calculated using the more sensitive lifestage (juvenile).
[c] As cited in U.S. Environmental Protection Agency online AQUIRE database (http://www.epa.gov/ecotox).
[d] The 50% lethal concentration for *Crangon crangon* is not used in the FAV calculation because of the wide concentration range, short test duration, and availability of data for another species in this genus.
[e] Calculated according to Stephan et al. [30].

differences in sensitivity among commercial PCB mixtures, Aroclors 1242 and 1254 exhibit similar toxicity based on comparisons of the most sensitive species and the overall distributions of toxicity values. The limited data available for commercial PCB mixtures containing lower (Aroclors 1221 and 1232) and higher (Aroclors 1260, 1262, and 1268) chlorine content suggest that these formulations are less toxic than Aroclors 1242 and 1254 [28,29]. Data for Aroclor 1254 were interpreted using the U.S. EPA [30] methodology for calculating final acute values (FAVs). The FAV is designed to estimate the fifth percentile of acute values for all possible genera based on the goal of protecting 95% of aquatic species from acute toxicity. The FAV for Aroclor 1254 is calculated as 6.96 μg/L (Table 1).

Because the available chronic toxicity data are more limited than the acute data, a final chronic value (FCV) was calculated from the FAV using an acute to chronic ratio (ACR). The ACR was developed using all studies from the LC50 data set for which an acute LC50 could be paired with a measure of test organism development or reproduction after chronic exposure. Three studies containing the requisite information were identified (Table 2). The acute value was divided by the chronic value to yield an ACR for each pair of tests. The geometric mean of these ACRs is 13. Dividing the FAV by this value results in a FCV of 0.54 μg/L. This FCV is approximately equal to the lowest chronic effect concentrations reported for mortality (LC50s for larvae and pupae of the midge *Tanytarsus dissimilis* exposed to Aroclor 1254 [28]) and is lower than most chronic values identified for reproduction, growth, or development of various invertebrate species [28,31–35]. The only exception is a measurable increase in larval abnormalities in the clam *Mercenaria mercenaria* after 48-h exposure to

Table 2. Invertebrate acute to chronic ratio (ACR) for polychlorinated biphenyls

| Chemical | Species | Method[a] | Chronic endpoint | Reference | Acute value[b] | Chronic value[b] | ACR |
|---|---|---|---|---|---|---|---|
| Aroclor 1242 | *Gammarus pseudolimnaeus* | F, M | Reproduction | [28] | 73 | 4.9 | 15 |
| Aroclor 1248 | *Gammarus pseudolimnaeus* | F, M | Reproduction | [28] | 29 | 3.3 | 8.7 |
| Aroclor 1254 | *Microarthridion littorale* | S, M | Reproduction (spiked sediment test) | [43] | 4,380 | 100 | 44 |
| Aroclor 1254 | *Uca pugilator* | R, U | Development and chronic mortality | [77] | 10 | 2.2 | 4.5 |
| Final ACR[c] | | | | | | | 13 |
| Final Chronic Value[d] (μg/L) | | | | | | | 0.54 |

[a] F = flow-through; M = measured; S = static; R = renewal; U = unmeasured.
[b] Geometric mean of available acute or chronic values. Units are μg/L except for *Microarthridion littorale* (μg/g organic carbon).
[c] Geometric mean of species mean ACR values.
[d] Final chronic value = Final acute value/ACR.

Table 3. Percentage homologue composition and chronic sediment-quality benchmarks for selected U.S. commercial and environmental polychlorinated biphenyl (PCB) mixtures[a]

| PCB homologue | Homologue log $K_{ow}$[b] | Commercial Aroclors | | | | New Jersey | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | 1242 | 1248 | 1254 | 1260 | Site A | Site B | Ohio site | Oregon site |
| Monochlorobiphenyls | 4.64 | 1 | | | | | 0.5 | 0.4 | |
| Dichlorobiphenyls | 5.12 | 13 | 1 | | | | 3 | 7 | |
| Trichlorobiphenyls | 5.62 | 45 | 21 | 1 | | | 14 | 31 | |
| Tetrachlorobiphenyls | 6.04 | 31 | 49 | 15 | | 25 | 30 | 34 | |
| Pentachlorobiphenyls | 6.49 | 10 | 27 | 53 | 12 | 44 | 24 | 24 | 1 |
| Hexachlorobiphenyls | 6.84 | | 2 | 26 | 42 | 29 | 16 | 3 | 34 |
| Heptachlorobiphenyls | 6.98 | | | 4 | 38 | 2 | 7 | 0.8 | 43 |
| Octachlorobiphenyls | 7.72 | | | | 7 | | 3 | 0.03 | 23 |
| Nonachlorobiphenyls | 8.24 | | | | 1 | | 2 | | 0.02 |
| Decachlorobiphenyls | 8.26 | | | | | | 1 | | |
| Total PCB log $K_{ow}$[c] | | 5.59 | 5.95 | 6.43 | 6.85 | 6.38 | 5.98 | 5.77 | 6.99 |
| Sediment-quality benchmark (μg/g organic carbon) | | 210 | 490 | 1,500 | 3,800 | 1,300 | 510 | 320 | 5,300 |

[a] Homologue concentrations in commercial Aroclors are from DeVoogt and Brinkman [86]. Environmental data are from P.C. Fuchsman (unpublished data).
[b] Average homologue-specific log octanol–water partition coefficient ($K_{ow}$) values are from Mackay et al. [38] and from Shiu and Mackay [37].
[c] Total PCB log $K_{ow}$ values are calculated using Equation 2.

0.01 μg/L of Aroclor 1254 [36], reflecting a sensitivity markedly greater than that in any other study we identified. Benthic survey results for PCB-contaminated sites, which are discussed further below, show generally good agreement with EqP predictions based on the FCV of 0.54 μg/L, indicating that the much lower *M. mercenaria* lowest-effect concentration [36] would not be a good predictor of ecologically significant effects, at least at the community level.

Because individual PCBs exhibit limited solubility in water, the toxicity data for Aroclor 1254 also were evaluated to assess whether the calculated FCV might be unduly influenced by nominal (unmeasured) toxicity values. Solubility of PCBs decreases with increasing chlorination; for instance, solubility limits have been reported as 1 to 100 μg/L for tetrachlorobiphenyls, 4 to 20 μg/L for pentachlorobiphenyls, and 0.4 to 1 μg/L for hexachlorobiphenyls [37]. Thus, at higher test concentrations, nominal exposures could overestimate true exposure concentrations, particularly for the hexachlorobiphenyl component of Aroclor 1254. As shown in Tables 1 and 2, the highest LC50s reported for Aroclor 1254 (≥100 μg/L) are all based on nominal concentrations. However, most of the data influencing the FCV calculation (i.e., sensitive LC50s and chronic data) are based on measured concentrations. Also, at these lower exposures (≤10 μg/L), solubility limitations should not be a major concern based on the homologue composition of Aroclor 1254 (Table 3) and the large number of individual congeners within each homologue (e.g., up to 42 hexachlorobiphenyls). Therefore, the FCV of 0.54 μg/L appears to be a reasonable estimate of a chronic toxicity threshold for sensitive invertebrate taxa.

*Organic carbon–water partitioning*

To apply this FCV to total PCBs in sediment, it is necessary to quantify how PCBs partition between sediment organic carbon and interstitial water. This task is complicated by the fact that PCB mixtures are comprised of multiple congeners, each of which exhibits different partitioning characteristics. The $K_{ow}$ for the PCB mixture can be calculated based on the distribution of PCB concentrations among different levels of chlorination (homologues) as follows:

$$K_{\text{ow-Total PCB}} = \frac{1}{\sum_{n=1}^{i} \frac{f_{\text{homologue } i}}{K_{\text{ow-homologue } i}}} \quad (2)$$

where $K_{\text{ow-Total PCB}}$ represents the overall $K_{ow}$ for the PCB mixture, $f_{\text{homologue } i}$ represents the fraction of PCB mixture consisting of homologue $i$, and $K_{\text{ow-homologue } i}$ is the $K_{ow}$ for homologue $i$. Homologue-specific log $K_{ow}$ values are identified as averages of available congener-specific values [37,38] (Table 3). Although $K_{ow}$ values differ somewhat among congeners within the same homologue, these differences should tend to cancel one another and are considered to be negligible relative to other uncertainties in the EqP model (e.g., relationship between $K_{ow}$ and $K_{oc}$).

The $K_{oc}$ parameter is estimated to equal the calculated $K_{ow}$ values based on data presented by Bucheli and Gustafsson [39]. Those authors compiled measured $K_{oc}$ values of PCB congeners in field-collected sediments (as opposed to laboratory-spiked sediments) and related them to $K_{ow}$ values. Excluding open ocean samples (in which much of the particulate organic carbon consists of living organisms), the assumption of equality of $K_{ow}$ and $K_{oc}$ represents a conservative (protective) estimate of the actual $K_{oc}$ (Fig. 2). Note that this data set tends to underestimate true $K_{oc}$ values for PCBs in sediment for two reasons. First, some studies included in the data set used measurement techniques that overestimate freely dissolved PCB concentrations (i.e., PCBs adsorbed to dissolved organic carbon were not distinguished from freely dissolved PCBs). This bias most strongly affects the higher-chlorinated PCBs, as is evident in Figure 2. Second, the data set includes partitioning data for suspended sediments, which typically contain diagenetically younger organic matter with somewhat lower sorptivity for hydrophobic organic compounds [39]. Even with these biases, it is evident from Figure 2 that $K_{oc}$ values for PCBs often can exceed the $K_{ow}$ of the compound. Strong sorption to soot or other hard carbon is one likely explanation [40]. The assumption of a one-to-one relationship between $K_{ow}$ and $K_{oc}$ is intentionally conservative; site-specific investigations of PCB partitioning, where feasible, could provide more accurate estimates of site-specific PCB bioavailability.

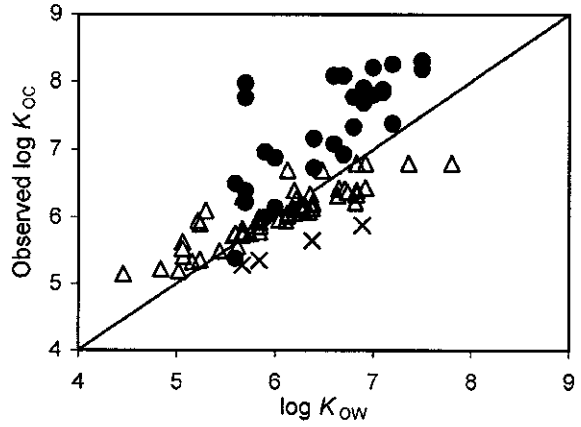


Fig. 2. Comparison of octanol–water partition coefficient ($K_{ow}$) and organic carbon/water partition coefficient ($K_{oc}$) values for polychlorinated biphenyls (PCBs) in different environmental compartments. Circles indicate bedded sediments (Ketelmeer), triangles indicate suspended sediments (Hudson, Detroit, and St. Clair rivers), and crosses indicate suspended sediments (Lake Superior). Data are from Bucheli and Gustafsson [39].

*EqP sediment-quality benchmarks*

The estimated log $K_{oc}$ values and sediment-quality benchmarks for various PCB mixtures, including technical Aroclors and site-specific example sediments, are shown in Table 3. All these sediment-quality benchmarks are calculated based on the FCV for Aroclor 1254, which introduces considerable uncertainty in the assessment, especially for sediments contaminated with less-chlorinated PCB mixtures. Because higher-chlorinated PCBs partition more strongly to sediment, the composition of freely dissolved (bioavailable) PCB mixtures in sediment is quite different than the PCB composition of the total PCB mixture in sediment (Fig. 3). Thus, sediment contaminated with Aroclor 1242 would exhibit a PCB profile in pore water more similar to that of Aroclor 1221 or 1232. Although invertebrate toxicity data are very limited for PCB mixtures with chlorine content lower than that of Aroclor 1242, *Daphnia magna* reproductive data [28] suggest that these less-chlorinated PCB mixtures are less toxic than the mixtures used here to represent the aquatic toxicity of PCBs. Despite potential overestimation of PCB bioavailability and toxicity in our calculations, the EqP approach provides sediment-quality benchmarks for a wide range of PCB homologue profiles that consistently exceed the consensus screening values of MacDonald et al. [16]. For comparison, if one assumes 1% organic carbon in the underlying data set of MacDonald et al. [16], then the consensus values are equivalent to 4 μg/g OC (threshold), 40 μg/g OC (midrange), and 170 μg/g OC (extreme), all of which are lower than the EqP benchmarks presented in Table 3.

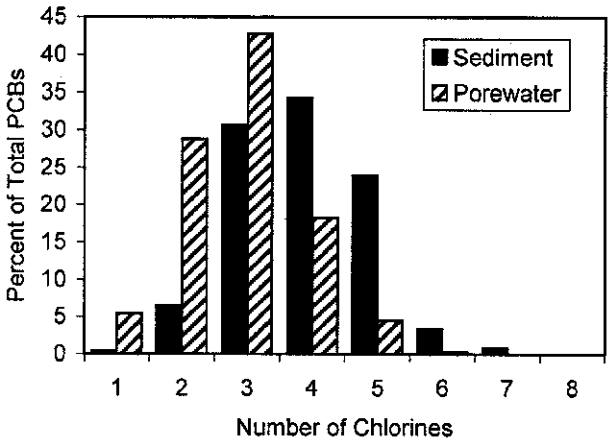


Fig. 3. Homologue composition of polychlorinated biphenyls (PCBs) sorbed to sediment (measured) and dissolved in pore water (predicted assuming equilibrium partitioning) at a site in Ohio, USA (P.C. Fuchsman, unpublished data). The composition of PCBs sorbed to sediment resembles that of Aroclor 1248, whereas the composition of PCBs dissolved in pore water resembles that of Aroclor 1232.

## SPIKED SEDIMENT TOXICITY TEST

Spiked sediment toxicity tests are controlled experiments that provide direct measurements of chemical toxicity in sediment. Such studies provide valuable cause–effect data, although chemical bioavailability may be unrealistically high if the study design does not incorporate a stabilization period to allow chemical partitioning to approach equilibrium [23]. Results of spiked sediment toxicity tests using PCBs are summarized in Table 4. The majority of tests showed no adverse effects at any test concentration (81–2,560 μg/g OC for PCB mixtures and up to 50,000 μg/g OC for individual PCB congeners). Toxicity was observed only in certain studies that did not incorporate a sediment equilibration period. For instance, Swartz et al. [41] observed a LC50 for the amphipod *Rhepoxinius abronius* of 2,900 μg/g OC, whereas Murdoch et al. [42] observed no significant mortality in the same species exposed to 2,560 μg/g OC after sediment equilibration. DiPinto et al. [43] also observed reduced copepod reproduction at 100 μg/g OC. However, this result was observed in only one of two duplicate experiments, and the study did not incorporate an equilibration period. By comparison, spiked sediment tests that incorporated an equilibration period and measured reproduction resulted in unbounded no-effect concentrations ranging from 2,560 μg/g OC for *Neanthes arenaceodentata* [44] to 10,000 μg/g OC for *Lumbriculus variegatus* [45] (although these species are not considered to be the most sensitive to hydrophobic organic compounds). The available results of spiked sediment toxicity tests with PCBs thus generally are consistent with the EqP benchmarks derived above, but additional testing would be useful to better define dose–response relationships and to compare the effects of differing PCB mixtures.

It should be noted that spiked sediment toxicity test results for individual PCB congeners or small numbers of congeners [45–48] may reflect a solubility cutoff (i.e., the Ferguson cutoff), at which the aqueous solubility of the compound is lower than the toxicity threshold. This effect also has been observed in high-molecular-weight PAHs, and it has been shown that even though the individual compounds cannot cause toxicity, mixtures of these compounds can [1]. The same phenomenon also would be expected for PCBs and should be taken into account in the design of future spiked sediment toxicity studies using specific PCB congeners.

In addition to the PCB-only spiking studies described above, two spiked sediment studies have evaluated the toxicity of chemical mixtures that included PCBs. Plesha et al. [49] observed toxicity from mixtures of PCBs, DDT, hexachlorobutadiene, and hexachlorobenzene. Because the reported DDT and hexachlorobutadiene levels were close to concentrations that could be toxic at equilibrium [50,51], and because the study did not incorporate an equilibration period, those results

Table 4. Spiked sediment toxicity test results for polychlorinated biphenyls (PCBs)[a]

| Species | Chemical | Equilibration time[b] | Exposure duration | Effect | Concentration (μg/g organic carbon) | Reference |
|---|---|---|---|---|---|---|
| Amphipod (*Pontoporeia hoyi*) | PCB mixture[c] | 1 d | 9 d | Survival unbounded NOEC | 420 | [46] |
| Amphipod (*Rhepoxynius abronius*) | Aroclor 1254 | NR | 10 d | LC50 | 2,900 | [41] |
| Amphipod (*Rhepoxynius abronius*) | Aroclor mixture[d] | 4 weeks | 10 d/20 d | Survival/growth unbounded NOEC | 2,560 | [42] |
| Shrimp (*Crangon septemspinosa*) | Aroclor 1242 | NR | 4 d | Survival unbounded NOEC | 280 | [78] |
| Shrimp (*Crangon septemspinosa*) | Aroclor 1254 | NR | 4 d | Survival unbounded NOEC | 1,200 | [78] |
| Copepod (*Microarthridion littorale*) | Aroclor 1254 | NR | 4 d | LC50 (females/males) | 6,400/3,000 | [43] |
| Copepod (*Microarthridion littorale*) | Aroclor 1254 | NR | 12 d | Reproduction unbounded LOEC[e] | 100 | [43] |
| Bent-nosed clam (*Macoma nasuta*) | PCB mixture[f] | Variable | 119 d | Survival/growth unbounded NOEC | 81 | [47] |
| Zebra mussel (*Dreissena polymorpha*) | 3,3′,4,4′-Tetrachlorobiphenyl | NR | 14 d | Survival unbounded NOEC | 50,000[g] | [48] |
| Oligochaete worm (*Lumbriculus variegatus*) | 3,3′,4,4′-Tetrachlorobiphenyl | 60 d | 10 d | Reproduction unbounded NOEC | 10,000 | [45] |
| Polychaete worm (*Neanthes arenaceodentata*) | Aroclor mixture[c] | 4 weeks | 120 d | Reproduction unbounded NOEC | 2,560 | [44] |

[a] LC50 = median lethal concentration; LOEC = lowest-effect concentration; NOEC = no-effect concentration; NR = not reported (presumably no equilibration period used).
[b] If spiked PCBs were not allowed to approach equilibrium partitioning before toxicity testing, bioavailability may have been exaggerated relative to field-contaminated sediment.
[c] Seven congeners.
[d] 22% Aroclor 1242, 66% Aroclor 1254, and 12% Aroclor 1260.
[e] Effect was not reproduced in duplicate experiment.
[f] Thirteen congeners.
[g] Sediment organic carbon not reported; concentration assumes 1% organic carbon.

provide little useful information specifically for PCBs. Swartz et al. [41] demonstrated that various mixtures of PCBs, metals, and PAHs could elicit toxicity when each component was present at half the individual-chemical LC50 (previously determined using comparable methods). These effects appeared to be subadditive. These results underscore the importance of considering mixture effects for sediments containing multiple constituents at concentrations approaching their individually derived cause–effect toxicity thresholds.

## BIOLOGICAL EFFECTS AT PCB-CONTAMINATED SITES

Toxicity testing and biological survey results were reviewed for seven major PCB-contaminated sites in the United States. Although other chemicals are present in sediment at these sites, PCBs are considered to be the primary chemicals of interest. Information from such sites does not, by itself, establish causality, but it can provide valuable confirmatory data. In addition, an eighth site in northern Canada is discussed, because this site contains PCBs but generally is remote from other sources of contamination. Toxicity test and benthic survey methods are summarized for the eight sites in Table 5. It is notable that few of the toxicity studies assessed chronic reproductive endpoints, because only recently have such test methods been developed. However, for chemicals other than PCBs, the difference in sensitivity between chronic reproduction and 10-d survival endpoints typically has been moderate [52] to essentially nonexistent [8,53–55], suggesting that any increased discriminatory power of chronic reproductive tests would be observed only for marginally toxic sediments. Ecologically significant chronic effects also should be observable in benthic community investigations (when appropriately designed), because such surveys directly measure the biological community targeted for protection.

### *New Bedford Harbor*

The 7,200-ha New Bedford Harbor Superfund site comprises an urban intertidal estuary and harbor in New Bedford, Massachusetts, USA. Historically, PCBs entered the site through waste discharges from electrical manufacturers. The homologue composition of PCBs at the site [56] results in EqP benchmarks of approximately 300 to 600 μg/g OC. Although PCB and metal concentrations covary at the site, two toxicity identification evaluation (TIE) studies indicated that metals were not important contributors to toxicity [57,58]. Specifically, a pore-water TIE identified PCBs as the most likely cause of shrimp and amphipod toxicity [57], and a whole-sediment TIE [58] confirmed hydrophobic organic compounds as the cause of toxicity to the same species. The latter study reported a sediment concentration of 304 mg/kg dry weight of total PCBs (~2,500 μg/g OC; organic carbon estimated from a site-specific correlation with total PCBs [59]). In a sediment dilution study, Zeng et al. [60] found that New Bedford Harbor sediment containing 15,000 μg/g OC of total PCBs inhibited sea urchin growth, whereas no effects were observed at 3,100 μg/g OC. Nelson et al. [59] integrated extensive biological monitoring data for the site, identifying three categories of sampling stations. Among stations exhibiting neither amphipod toxicity nor benthic degradation, average total PCB concentrations were 0.93 mg/kg dry weight (estimated as 90 μg/g OC). Stations exhibiting some benthic degradation without significant amphipod mortality were characterized by an average of 12 mg/kg dry weight of total PCBs (estimated as 150 μg/g OC); in these cases, it is unclear whether the benthic

Table 5. Summary of toxicity test and benthic survey methods employed at eight polychlorinated biphenyl–contaminated sites

| Site | Toxicity test methods | Benthic survey methods |
|---|---|---|
| New Bedford Harbor, USA | 10-d *Ampelisca abdita* survival [59]<br>Toxicity identification evaluation: Pore-water 48-h *A. abdita* and *Mysidopsis bahia* survival; pore-water 48-h *Arbacia punculata* fertilization; whole-sediment 96-h *A. abdita, M. bahia*, and *Leptocheirus plumulosus* survival [57]<br>Toxicity identification evaluation: 96-h *A. abdita* and *Americamysis bahia* survival [58]<br>35-d *Lytechinus pictus* survival, growth, and embryo health [60] | Young-modified van Veen grab (3 replicates/station): Biomass, species richness, Environmental Monitoring and Assessment Program (EMAP) benthic community condition index, community structure [59] |
| Fox River, USA | 10- and 37-d *Lumbriculus variegatus* survival; 10- and 30-d *Hexagenia limbata* survival; 10-d *Chironomus riparius* survival and growth; 10- and 30-d *Hyalella azteca* survival and growth [61]<br>Pore-water 7-d and 48-h *Ceriodaphnia dubia* survival and reproduction [62] | Ekman dredge (5 replicates/station): Abundance, taxa richness, taxa diversity, trophic condition index [61] |
| Hudson River, USA | 10-d *Chironomus tentans* and *H. azteca* survival and growth [63] | Ponar (3 replicates/station): Abundance, pollution-tolerant species, species diversity, species dominance, and community similarity [63]<br>Ekman grab (6 replicates/station): Abundance, biomass, taxa richness, species diversity, community similarity [64] |
| Housatonic River, USA | 28-d Survival and growth, 35- and 42-d survival and reproduction of *H. azteca*; 20-d survival and growth, 23- to 43-d survival and emergence of *C. tentans*; in situ 48-h and 10-d *Daphnia magna, C. tentans*, and *H. azteca* survival [65] | Grab (12 replicates/station): Abundance, taxa richness, taxa diversity, modified Hilsenhoff Biotic Index, Ephemeroptera/Plecoptera/Trichoptera (EPT) abundance, relative abundance of dipterans/oligochaetes/gastropods [65] |
| Brunswick Estuary, USA | Whole-sediment 10-d *H. azteca* survival and leaf consumption [67]<br>14-d *L. plumulosus* survival, behavioral abnormalities, and sediment avoidance [68]<br>28-d *L. plumulosus* survival and growth (T.H. DeWitt et al., 1997, report PNWD-2384, Battelle, Richland, WA, USA; unpublished data) | 3-cm Acetate corer (10 replicates/station): Abundance, density, community composition [68]<br>Dip-net: *Palaemonetes pugio* length, weight, brood size, brood weight [69] |
| Sheboygan River, USA | 10-d *H. azteca* and *C. riparius* survival and growth [70] | Petite ponar (5 replicates/station): Abundance and taxa richness; other metrics not useful because of ubiquitous oligochaete dominance [70] |
| Kalamazoo River, USA | None | Dip-net: Abundance, taxa richness, species diversity, community composition [72] |
| Irving Whale Salvage Site, Canada | 10-d *Amphiporeia virginiana* survival [74] | Ship-deployed Institutt for Kontinentalsokkelundersøkelser dredge: species abundance, richness, evenness, and diversity [74] |

degradation reflects a low-level toxic effect or covariance between PCBs and physical habitat gradients. Among stations exhibiting both amphipod toxicity (45% survival on average) and benthic degradation, the average total PCB concentration was 89 mg/kg dry weight (estimated as 900 μg/g OC). Overall, these results generally are consistent with EqP predictions and are not consistent with empirical guidelines.

*Fox River*

The Fox River/Green Bay Superfund site in Wisconsin, USA, includes a 63-km stretch of the Fox River as well as a portion of Green Bay. The source of PCBs in this system is related to past paper recycling (because of the historical use of PCBs in carbonless copy paper). Other potential stressors at the site include eutrophication, pesticides, PAHs, and metals. Call et al. [61] conducted benthic community surveys, toxicity testing, and chemical analyses of sediment and biota tissue at 12 locations within the site as well as at an upstream reference station. Total PCB concentrations in sediment ranged from 3.7 to 540 μg/g OC. By comparison, the homologue composition reported for the site results in an EqP benchmark of 270 μg/g OC. The benthic community throughout the study area was considered to be characteristic of a degraded, eutrophied system. In tests with several invertebrate species, toxicity was observed, but it was attributed to high ammonia levels. When ammonia was removed through vigorous aeration or zeolite treatment, no PCB-related toxicity was observed. Concentrations of PCBs in tissue from the toxicity test organisms and field-collected invertebrates were determined to be below potentially toxic levels [61]. Consistent with these findings, a pore-water TIE study demonstrated that ammonia was responsible for toxicity observed in exposures to pore water extracted from site sediments [62]. The biological data for the Fox River thus generally are consistent with EqP predictions and are inconsistent with empirical guidelines.

*Hudson River*

The Hudson River PCBs Superfund site includes 320 km of the Hudson River in New York, USA. The presence of PCBs at the site is related to past manufacture of electrical capacitors. The homologue composition of PCBs at the site (http://www.darp.noaa.gov/northeast/hudson/odf/hr577.pdf) results in EqP benchmarks of approximately 200 to 400 μg/g OC. The New York Department of Environmental Conservation ([63]; http://www.dec.state.ny.us/website/dow/bwam/sed.html) conducted amphipod and midge toxicity tests on six sediment samples from the site, containing 19 to 49 μg/g OC of total PCBs. No significant effects on survival or growth were observed. A benthic community survey was conducted concurrently, but it did not account for major physical habitat variables (e.g., the gradient from freshwater to estuarine habitat) and, therefore, was inconclusive. A separate benthic community survey ([64]; http://www.epa.gov/hudson/reports.htm#links2report) identified grain size characteristics as the likely explanation of differences among sample locations in the Upper Hudson River, where total PCB concentrations ranged from 300 to 550 μg/g OC. Survey results for the Lower Hudson River were considered to be potentially confounded by physical habitat differences ([64]; http://www.epa.gov/hudson/reports.htm#links2report). Overall, the Hudson River data show better agreement with EqP predictions than with empirical guidelines.

*Housatonic River*

The Housatonic River site, regulated under the Resource Conservation and Recovery Act, includes 200 km of the Housatonic River in Massachusetts and Connecticut (both in USA). The source of PCBs at the site is related to past manufacture and repair of electrical transformers. The site-specific homologue composition of PCBs results in an EqP benchmark of approximately 4,000 μg/g OC, which is within the range of sediment PCB concentrations reported for the site. An extensive investigation of the benthic invertebrate community was conducted at the site, and a variety of toxicity tests were performed for a subset of the benthic survey stations [65]. However, the relationship between sediment chemistry and biological effects cannot be interpreted with confidence because of extreme intrastation variability in reported PCB concentrations (one to two orders of magnitude), which prevents even a rank ordering of sediment PCB exposures among stations. In this case, invertebrate tissue concentrations should provide a more useful measure of exposure, and indeed, the paired tissue data for PCBs show better agreement with benthic survey results among coarse-grain locations than do the sediment data. The tissue data do not indicate any clear dose–response relationship between PCBs and toxicity test endpoints, however. Potential confounding factors include the presence of elevated PAH concentrations (not measured in tissue) and grain-size effects (fine-grain reference station excluded from toxicity testing). Although various putative sediment toxicity thresholds have been extracted from this data set [65], we do not consider the Housatonic River data to be a useful test of either EqP predictions or empirical guidelines because of the variability associated with the sediment PCB data.

*Brunswick Estuary*

The Linden Chemicals and Plastics Superfund site, located in Brunswick, Georgia, USA, is contaminated with PCBs and mercury because of past releases from a variety of industrial facilities. The site encompasses 220 ha, the majority of which are tidal marsh. Highly chlorinated Aroclor 1268 was the primary PCB mixture released to the site, and the site-specific homologue composition [66] results in an EqP benchmark of 12,000 μg/g OC. Winger et al. [67] observed no mortality of freshwater amphipods exposed to sediments containing 1,100 to 2,300 μg/g OC of total PCBs (as well as 18–25 mg/kg of mercury), although decreased leaf consumption was observed. Horne et al. [68] observed no significant mortality or behavioral effects on estuarine amphipods exposed to site sediments containing up to 19,000 μg/g OC of total PCBs (and 170 mg/kg of mercury), whereas with a longer test duration, the same species exhibited increased mortality when exposed to 27,000 μg/g OC of total PCBs (and 972 mg/kg of mercury) but not when exposed to 16,000 μg/g OC of total PCBs (and 570 mg/kg of mercury) (T.H. DeWitt et al., report PNWD-2384, Battelle, Richland, WA, USA, unpublished data). A benthic community investigation showed increased dominance by polychaetes and a shift toward surface feeders at PCB concentrations of 4,400 μg/g OC and higher, although covarying gradients in organic carbon and mercury concentrations, as well as unmeasured chemicals, complicate interpretation of this result [68]. Wall et al. [69] evaluated salt marsh function at lower trophic levels and found "only subtle indications" of impairment, the most notable being a disruption in the size-versus-fecundity relationship for grass shrimp. Methylmercury concentrations were more closely correlated to various mea-

sures of salt marsh function than were PCB levels. The Brunswick estuary findings show better agreement with EqP predictions than with empirical guidelines.

*Sheboygan River*

The Sheboygan River and Harbor Superfund site includes approximately 23 km of the Sheboygan River in Wisconsin, USA. The primary source of PCBs at the site was related to the use of hydraulic fluids in past die-casting operations. Toxicity testing and benthic community surveys were performed for 12 locations, of which two were selected to represent elevated PCB concentrations upstream of other major contaminant sources ([70]; http://response.restoration.noaa.gov/resource_resourcetopic.php?RECORD_KEY%28resourcetopics%29=resourcetopic_id&resourcetopic_id(resourcetopics)=13). One of these locations, containing 12,000 μg/g OC of total PCBs, was severely toxic to amphipods and midges and exhibited benthic degradation. This location was characterized by a predominance of di- and trichlorinated homologues (similar to Aroclor 1232), suggesting relatively high PCB bioavailability and an EqP benchmark of less than 200 μg/g OC. The remaining locations contained 4 to 33 μg/g OC of total PCBs, characterized as a combination of Aroclors 1242 and 1254 (suggesting an EqP benchmark between 200 and 1,500 μg/g OC). These locations did not appear to be adversely affected by PCBs, although two locations showed evidence of PAH-related toxicity. Because of the very large difference in PCB concentrations between affected and unaffected locations, the Sheboygan River findings cannot discriminate between EqP predictions and empirical guidelines.

*Kalamazoo River*

The Allied Paper/Portage Creek/Kalamazoo River Superfund site includes a 130-km stretch of the Kalamazoo River in Michigan, USA, as well as upland properties and a small tributary. Polychlorinated biphenyls were released into the site during the historical recycling of carbonless copy paper and have migrated downstream over time. The homologue composition of PCBs at the site ([71]; http://www.kzooriver.com/kalamazoo/filearchive/ri-report.pdf) results in an EqP benchmark of approximately 400 μg/g OC. Qualitative surveys indicated "excellent" benthic community quality in the upper portion of the site [72]; sediment chemistry was not evaluated in these surveys. However, sediment data for the same reaches indicate an average total PCB level of approximately 70 μg/g OC ([73]; http://www.michigan.gov/deq/0,1607,7-135-3311_4109_4217-85234-,00.html). Sediment toxicity tests have not been performed for this site. The limited data for the Kalamazoo River indicate agreement with EqP predictions and not with empirical guidelines.

*Irving Whale Salvage Site*

The Irving Whale Salvage Site is located in the Gulf of St. Lawrence, Canada. In 1970, the oil barge *Irving Whale* sank, during or after which PCBs were released from the barge heating system. Recovery and salvage operations were conducted in 1996. This site is of interest because it is remote from other sources of contamination, although it is reasonable to expect that PAHs also could have been released from the barge. Amphipod toxicity tests and benthic community data showed adverse effects only at total PCB concentrations of 9,800 μg/g OC or higher. No effects were observed at PCB concentrations up to 1,000 μg/g OC [74]. Although the homologue composition of PCBs at the site was not reported, these biological findings are consistent with the range of EqP benchmarks for less-chlorinated PCB mixtures and are not consistent with empirical guidelines for PCBs.

## DISCUSSION

The evident discrepancy between the cause–effect sediment-quality benchmarks developed here and the empirical consensus guidelines developed by MacDonald et al. [16] can be explained by two key factors. First, the empirical assessment methods underlying the consensus values necessarily identify screening values that are within the range of the data evaluated, regardless of whether a particular chemical actually contributed significantly to toxicity in the evaluated sediments. Second, covariance between multiple chemicals is the norm in contaminated sediments; that is, PCBs rarely are found in otherwise clean sediments. Thus, higher PCB concentrations likely are associated with a higher probability of toxicity from other chemicals, even when the PCB concentrations are not elevated enough to contribute significantly to toxicity. Thus, the consensus guidelines are viewed most appropriately as indicators of the overall levels of sediment contamination associated with different probabilities of toxicity. Assessment methods that integrate empirical sediment-quality guidelines for multiple chemicals can be useful for answering the question whether a given sediment will be toxic. Cause–effect benchmarks are more appropriate for answering questions such as what are the risks to invertebrates specifically from PCB exposures or, conversely, might PCBs be a contributing stressor in a sediment that exhibits toxicity to invertebrates.

The EqP benchmarks derived here provide an improved framework for understanding cause–effect relationships between PCBs in sediment and effects on benthic invertebrates. However, ample room exists for further research and refinement. For instance, additional information regarding ACRs (aquatic or sediment-based) would help to refine the toxicity analysis developed here. Additional studies comparing dose–response relationships for multiple, environmentally relevant PCB mixtures would help to clarify whether a single aquatic toxicity value is appropriate for broad application to total PCBs. The utility of homologue-specific toxicity values merits investigation, and sediment spiking studies also could be used to validate further the predicted differences in PCB bioavailability among mixtures of differing homologue composition. The EqP analysis provided above should help to guide the selection of exposure concentrations in such future studies so that effects thresholds can be bracketed more effectively.

Finally, it must be emphasized that the EqP benchmarks and analyses presented here are applicable only to benthic invertebrates. Generic bioaccumulation-based screening values likely are lower than these EqP benchmarks, because they are intended to be protective of piscivores in systems with high bioaccumulation and exposure potential. Assessments of risk to benthic invertebrates should be conducted in parallel with appropriate investigations of other relevant exposure pathways.

*Acknowledgement*—We thank Thomas Bucheli and Orjan Gustafsson for sharing their PCB partitioning data set. Chris Ingersoll, Don MacDonald, and three anonymous reviewers provided valuable comments on earlier drafts of this paper. Loren Yeager assisted with manuscript preparation.

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## Katz, Eric (OC)

**From:** Daniel Fuchs [Daniel.Fuchs@doj.ca.gov]
**Sent:** Friday, May 23, 2008 3:22 PM
**To:** Katz, Eric (OC)
**Cc:** Caryn Craig
**Subject:** SQOs
**Attachments:** e-mail chain with oal.pdf; Withdrawal_Ntc.pdf

Eric:

In response to your oral Public Records Act request for communications between the State Board and the Office of Administrative Law, and pursuant to our conversation of this afternoon, I attach a pdf of the e-mail chain between the State Board and OAL, as well as a scan of the notice of withdrawal sent by OAL to the State Board. The e-mail chain is in reverse chronological order -- i.e., it reads up.

These attachments constitute the entire set of documents responsive to your request. Please contact me with any questions.

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>>> "Dale P. Mentink" <dmentink@oal.ca.gov> 4/14/2008 12:42 PM >>>
Thanks, I'll process that today. Just so you're not searching through emails and opening/printing any that are not relevant, the three emails containing questions/issues for follow-up or supplemental information or amendment are: 4/2/08 at 10:05 a.m.; 4/4/08 at 2:34 p.m.; and 4/10/08 at 4:58 p.m. Please let me know if you have any other questions.

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-----Original Message-----
From: Chris Beegan [mailto:cbeegan@waterboards.ca.gov]
Sent: Monday, April 14, 2008 9:54 AM
To: Dale P. Mentink
Cc: Bruce Fujimoto; Dominic Gregorio; Sheila Vassey
Subject: OAL file no. SWRCB_2008-0229-07 sediment quality

Hi Dale: Thanks for the call this morning and your patience and diligence during the review of the record referenced above. As we discussed, I would like to withdraw the admininstrative record (OAL file no. SWRCB_2008-0229-07 sediment quality) with the intent of resubmitting a revised record ASAP after I have addressed those issues you identified in previous emails. Again thank you for your time and patience.

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>>> "Dale P. Mentink" <dmentink@oal.ca.gov> 4/11/2008 8:31 AM >>>
I promised to get back to you this morning re which comment that was that mentioned the CWC and CWA; it's number 499 on page 08657 of the record. Thank you.

Dale, OAL

323-6817

>>> "Dale P. Mentink" <dmentink@oal.ca.gov> 4/10/2008 4:57 PM >>>
Chris, I left a message for you this morning as to how things are going, but maybe you're out today.

A couple of other questions came up from our rulemaking file review committee. Two were just questions on the plan and one was a request for some additional information in response to one of the comments as long as I had asked about those other 14.

In terms of the plan, on page 08602 (page 8 of the plan), the committee wondered what the standard will be for approval of the "other methods" approved for use by the boards? In other words, if there are some standards already in mind, those should be stated in the plan, or the plan should simply explain why that approval process needs to be case by case.

On page 08614 (page 20 of the plan), what does the phrase "or other waters of significant national importance" refer to?

The other response to comment (and I'll find the page in the record in the morning) was just the one in which the use of the narrative as opposed to numeric object was challenged and being responded to and you mentioned that when numeric criteria are infeasible, the Water Code and Clean Water Act authorize the use of narratives. Could you provide the Water Code and Clean Water Act cites for that?

Thank you.

Please let me know how things are going. We have a due date of 4/14.

>>> "Dale P. Mentink" <dmentink@oal.ca.gov> 4/4/2008 2:33 PM >>>
Dear Chris:

I have finished my review of this file. In addition to the questions I raised in my email to you of April 2, I identified a few other things in the areas of the concise summary language for Title 23, the summary and response to comments, and the administrative record.

1. Concise Summary. Stressor Identification appears to be an important part of the plan. It's mentioned in the Summary of the plan at I.B. It's discussed for a number of pages in the Program of Implementation section. It answered some of the commenters regarding their objection to use of the empirical method rather than the mechanistic. It's discussed in several other contexts. Is there a reason why Stressor Identification was not mentioned specifically in the

Clear and Concise Summary which is going into Title 23?

2. Administrative Record. Because I spent a fair amount of time with the file and Index and noticed some anomalies, I made a note of them. I provide these to you only if they are of some use to you.

a. 00128 is not a blank page, it's the last page of the Wilson memo although on back side.

b. 00193-00208 is not a second copy of settlement agreement, it looks like a briefing paper.

c. 00351-00352 are not indexed; it is a Notice of Public Hearing.

d. 00427-00429 are not indexed; they are Beegan to Bay email.

e. 00450 missing.

f. 00525 says it's a Cassette Tape (SQO Work Plan Adoption), but there's no tape.

g. 05739-06191 is dated 10/09/07 in the Index but 11/09/07 on the document.

h. 00671-00690 not indexed and not in box.

i. 01577-01578 not indexed but are in box (sign in sheet).

j. 03062-03122 not indexed but are in box.

k. 06779-06780 index has Richardson letter as only 06780 (actually a two-page letter).

3. Summary and Response to Comments.

I found substantial compliance with the summary of public comments consistent with 40 CFR Sec. 25.8. A comment that does not appear to have been acknowledged is on page 06643: "Minimum sample size for multi-station assessment of a single water body and minimum geographical coverage extent for individual sampling events must be specified." If you are amending the summary and response to comments (see below), could you summarize and respond to that one please?

In terms of responses, I identified 12 where the response was only that Staff Disagrees, but the comment wasn't lacking in specifics or merely rhetorical. Could you flesh out the responses to those 12 somewhat please? They are on the following pages: 08717 (30), 08718 (228), 08719 (231), 08719 (557), 08727 (23), 08737 (257), 08739 (262), 08747 (20), 08777 (298), 08800 (366), 08805 (381), and 08805 (384).

On page 08725 (241), the comment contains a specific question [Does this mean that significant differences for any two lines of evidence could drive an impairment designation?] and a specific recommendation [We

recommend that benthic community data must be one of the two lines of evidence suggesting adverse effects before an impairment designation is assigned.]. The response is not really responsive. Could you flesh out that response?

On page 08803 (375), the comment is that monitoring may be as infrequent as once in five years and that such infrequent monitoring will allow degradation. The response is that the language describing maximum frequency has been deleted. There is nothing in the response explaining/justifying the minimum frequency monitoring which the commenter raised. Could you flesh out that response?

Please call or reply if you have any questions.

Dale Mentink, Office of Administrative Law, 323-6817.

>>> "Dale P. Mentink" <dmentink@oal.ca.gov> 4/2/2008 10:04 AM >>>
Chris:

Thank you for the DVD. It worked fine. I'm just trying to tick off some items on my file review checklist and forgive me if I'm overlooking the obvious or looking for something(s) not there. I suspect you don't have another copy of this file there, but if you have the index as a document on your computer, I was wondering if you could help me identify a couple of things.

1. In terms of a Water Code 13147, 13244, and/or 40 CFR 25.5(b) Notice document, prior to the 11/19/07 or 2/5/08 public hearings or the 11/30/07 comment deadline, what in the record would you refer me to?

2. The plan document presented to the public for public comment prior to the 11/30/07 comment deadline is where in the record?

3. Is there a document or documents evidencing consultation with Public Resources Code 21080.3, 14 CCR 15386 agencies, or a document evidencing that none of those agencies have jurisdiction?

4. Was there ever a vote of the Board approving the concise summary text that is going into title 23 and is there a document showing that?

Thank you for whatever assistance you can be.

Dale Mentink

Office of Administrative Law

323-6817

>>> "Dale P. Mentink" <dmentink@oal.ca.gov> 3/19/2008 9:31 AM >>>
Hello, my name is Dale Mentink. I'm with the Office of Administrative Law and am reviewing this file. I'm sorry if I'm unable to find something obvious in the boxes, but I'm trying to do a sampling of comments and responses review. I have the oral comment summary and responses at pages 08639 through 08656. Is there a tape, minutes, or DVD of that testimony that I can review to hear or see the comments? I have a DVD of the 2/19/08 meeting (p. 08899), but I believe the testimony was taken on 2/5/08.

Also with regard to the summary and response to written comments at pages 08657 through 08841, is there a way to determine which document in the file contains the comments of No. 1, or No. 102, or No. 499 (e.g., page 08657) and so forth?

Thank you very much for your assistance.

# State of California
# Office of Administrative Law

| | |
|---|---|
| **In re:**<br>**State Water Resources Control Board** | **NOTICE OF WITHDRAWAL** |
| **Regulatory Action:** | **Government Code Section 11349.3(c)** |
| **Title 23, California Code of Regulations** | **OAL File No. 2008-0229-07 S** |
| **Adopt sections:** 3006<br>**Amend sections:**<br>**Repeal sections:** | |

This notice confirms that your proposed regulatory action regarding Water Quality Control Plan For Enclosed Bays and Estuaries, Part 1 - Sediment Quality was withdrawn from OAL review pursuant to Government Code section 11349.3(c). We will retain the rulemaking record you submitted in the event that you resubmit this regulatory action.

Please contact me at (916)323-6817 or dmentink@oal.ca.gov, or the OAL Reference Attorney at (916)323-6815, if you have any questions about the resubmittal process. You may request the return of your rulemaking record by contacting the OAL Front Desk at (916)323-6225.

Date: 4/14/2008

Dale P. Mentink
Senior Staff Counsel

For: SUSAN LAPSLEY
Director

Original: Dorothy Rice
Copy: Chris Beegan

STATE OF CALIFORNIA ARNOLD SCHWARZENEGGER, Governor

**OFFICE OF ADMINISTRATIVE LAW**
300 Capitol Mall, Suite 1250
Sacramento, CA 95814
(916) 323-6225 FAX (916) 323-6826

**Susan Lapsley**
**Director**



Eric Katz
Latham & Watkins
600 West Broadway, Suite 1800
San Diego, CA 92101-3375

Re: Sediment Quality Objectives for Enclosed Bays and Estuaries
OAL file no. SWRCB_2008-0229-07S

Dear Mr. Katz:

Pursuant to your request, enclosed please find copies of all emails between this office and the SWRCB contact person, Chris Beegan, the Clear and Concise Summary of the Water Quality Control Plan for Enclosed Bays and Estuaries of California, Part 1 - Sediment Quality, and copies of any public documents generated by OAL in connection with this file. The latter consists of the Notice of Withdrawal.

Sincerely,

Dale Mentink
Senior Staff Counsel

Enclosures

## Mentink, Dale@OAL

**From:** Chris Beegan [cbeegan@waterboards.ca.gov]
**Sent:** Wednesday, March 19, 2008 2:46 PM
**To:** Dale P. Mentink
**Subject:** Re: Sediment Quality, Part 1, OAL file no. 2008-0229-07S
**Attachments:** Key to Comment Letters.xls; SQO index with Comment Abreviations.xls

Hi Dale: Thank you for discussing the record with me today. I have created a smaller spreadsheet (Key to comment letters) that encompasses just the comment letters and the affiliation that is referenced in the responses to comments document. I have also amended the index with the affiliations and highlighted those in yellow.

Chris Beegan
SWRCB
Division of Water Quality
Sediment Quality Objectives Development
1001 I Street, Sacramento Ca 95814
cbeegan@waterboards.ca.gov
Office (916) 341-5577
Cell (916) 955-9262

>>> "Dale P. Mentink" <dmentink@oal.ca.gov> 3/19/2008 9:31 AM >>>
Hello, my name is Dale Mentink. I'm with the Office of Administrative Law and am reviewing this file. I'm sorry if I'm unable to find something obvious in the boxes, but I'm trying to do a sampling of comments and responses review. I have the oral comment summary and responses at pages 08639 through 08656. Is there a tape, minutes, or DVD of that testimony that I can review to hear or see the comments? I have a DVD of the 2/19/08 meeting (p. 08899), but I believe the testimony was taken on 2/5/08.

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Thank you very much for your assistance.

Dale Mentink

Senior Staff Counsel

Office of Administrative Law

300 Capitol Mall, Suite 1250

Sacramento, CA 95814

(916) 323-6817

FAX (916) 323-6826

dmentink@oal.ca.gov

| | | | | |
|---|---|---|---|---|
| 2007/11/25 | Comment Letter - Weston | David W. Moore | BOX 2 | 06366-06403 |
| 2007/11/28 | Comment Letter - Sierra Club | Edward Kimura | BOX 2 | 06404-06409 |
| 2007/11/28 | Comment Letter - OCSD | Michael Moore | BOX 2 | 06410-06412 |
| 2007/11/29 | Comment Letter - SRCSD | Wendell H. Kido | BOX 2 | 06413-06417 |
| 2007/11/29 | Comment Letter - LACo | Donald L. Wolfe | BOX 2 | 06418-06421 |
| 2007/11/29 | Comment Letter - RB5 | Kenneth D. Landau | BOX 2 | 06422-06437 |
| 2007/11/29 | Comment Letter - OCRDMD | Chris Crompton | BOX 2 | 06438-06448 |
| 2007/11/29 | Comment Letter - CVCA | Debbie Webster | BOX 2 | 06449-06454 |
| 2007/11/29 | Comment Letter - LACSD | Stephen R. Magun | BOX 2 | 06455-06470 |
| 2007/11/29 | Comment Letter - JLB | James L. Byard | BOX 2 | 06471-06482 |
| 2007/11/30 | Comment Letter - Caltrans | G. Scott McGowen | BOX 2 | 06483-06488 |
| 2007/11/30 | Comment Letter - WPHA | Nasser Dean | BOX 2 | 06489-06499 |
| 2007/11/30 | Comment Letter - Tri-TAC/CASA | Jim Colston | BOX 2 | 06500-06507 |
| 2007/11/30 | Comment Letter -SCV | Adam W. Olivieri | BOX 2 | 06508-06509 |
| 2007/11/30 | Comment Letter - HTB | Kirsten James, Mark Gold | BOX 2 | 06510-06514 |
| 2007/11/30 | Comment Letter - BACWA | Michelle Pla | BOX 2 | 06515-06521 |
| 2007/11/30 | Comment Letter - IEA | Patti Krebs | BOX 2 | 06522-06533 |
| 2007/11/30 | Comment Letter - FLI | Susan C. Paulsen | BOX 2 | 06534-06551 |
| 2007/11/30 | Comment Letter - CASQA | Bill Busath | BOX 2 | 06552-06563 |
| 2007/11/30 | Comment Letter - SFBK | Jen Kovecses | BOX 2 | 06564-06587 |
| 2007/11/30 | Comment Letter - WSPA | Kevin Buchan | BOX 2 | 06588-06627 |
| 2007/11/30 | Comment Letter - CCOC | Valerie Nera | BOX 2 | 06628-06778 |
| 2007/11/30 | Comment Letter - LW | Kelly E. Richardson | BOX 3 | 06780 |
| 2007/11/30 | Comment Letter - SDCK | Gabriel Solmer | BOX 3 | 08310-08319 |
| 2007/11/30 | Comment Letter - CLTNS | Tim Johnson, Parry Klassen, L. Ryan Broddrick | BOX 3 | 08320-08321 |
| 2007/11/30 | Comment Letter - LADWP | Katherine Rubin | BOX 3 | 08322-08326 |

ADMINISTRATIVE RECORD
FOR SUBMITTAL OF REGULATORY PROVISIONS OF PART 1 OF THE
WATER QUALITY CONTROL PLAN FOR BAYS ESTUARIES

| DATE | TITLE | AUTHOR | BOX | PAGE |
|---|---|---|---|---|
| 1994/01/04 | Guidance on consideration of economics in the adoption of water quality objectives | SWRCB | BOX 1 | 00001-00006 |
| 1995 | Incidence of Adverse Biolobical Effects Within Ranges of Chemical Concentrations in marine and Estuarine Sediments | | BOX 1 | 00007-00008 |
| 1996/10/02 | Resolution No. 92-49 Policy for invest. And cleanup and abatement under WC 13304 | | BOX 1 | 00009-00027 |
| 1998/10/28 | Use of Sediment Quality Guidelines (SQGs) in Dredged Material Management Decision-Making | CECW-OD | BOX 1 | 00028-00030 |
| 1999/03/01 | Do TMDLs have to include implementation Plans? | William R. Attwater | BOX 1 | 00031-00038 |
| 1999/01/07 | TMDL questions | Sheila K. Vassey | BOX 1 | 00039-00040 |
| 1999/10/27 | economic considerations in TMDL development and basin planning | Sheila K. Vassey | BOX 1 | 00041-00046 |
| 1999/12/30 | Hearing: re: proposed judgment Superior Court of the State of California County of Sacramento | | BOX 1 | 00047-00063 |
| 2000/09/20 | Use of washington state sed cleanup STDs | | BOX 1 | 00064-00073 |
| 2000/12/21 | Guidance regarding section 303(d) list for the 2002 submission | Mchael J. Levy | BOX 1 | 00074-00077 |
| 2001/01/26 | Guidance Regarding the Extent to Which Effluent Limitations Set Forth in NPDES Permits Can Be Relaxed in Conjunction with a TMDL | Stefan Lorenzato | BOX 1 | 00078-00081 |
| 2001/08/02 | Regulatory and statutory time limits implicated in developing california's 303(d) listing and delisting policy | Michael J. Levy | BOX 1 | 00082-00085 |
| 2001/09/18 | Respondents' Proposed Judgment | Bruce Reeves for Bill Lockyer | BOX 1 | 00086-00110 |
| 2001/10/11 | Certificate of Service by Mailing | | BOX 1 | 00111-00012 |
| 2001/10/11 | Premptory writ of mandate | | BOX 1 | 00113-00114 |
| 2001/10/11 | Complaint for Declaratory Relief; Petition for writ of Mandate entry of Judgment grantin gperemptory writ of mandate | | BOX 1 | 00115 |
| 2001/10/16 | Declaration of Service | | BOX 1 | 00116 |
| 2001/10/16 | Legal authority for offsets, pollutant trading, and market programs to supplement water quality regulation in california's impaired waters | Craig M. Wilson | BOX 1 | 00117-00127 |
| | Blank Page | | BOX 1 | 00128 |
| 2001/12/04 | Sediment Quality Objective Board Resolution for $900,000 from Clean Water Programs | Mike Harper | BOX 1 | 00129-00133 |
| 2002/01/23 | CAA funding item resolution | | BOX 1 | 00134-00136 |
| 2002/01/28 | The extent to which TMDLs are subject to the Alaska rule | Michael J. Levy | BOX 1 | 00137-00142 |
| 2002/02/22 | Applicability of state board resolution 92-49 in setting sediment cleanup levels | Craig M. Wilson | BOX 1 | 00143-00152 |
| 2002/02/25 | Post Hearing Documents | Bruce Reeves/Shelladie | BOX 1 | 00153-00177 |

ADMINISTRATIVE RECORD
FOR SUBMITTAL OF REGULATORY PROVISIONS OF PART 1 OF THE
WATER QUALITY CONTROL PLAN FOR BAYS ESTUARIES

| | | | | |
|---|---|---|---|---|
| 2002/03/21 | Re: Cleanup and abatement account fund contracts (toxic hotspots/sediment quality objectives) | Mike Harper | BOX 1 | 00178-00179 |
| 2002/03/21 | cleanup and abatement account fund contracts (toxic hotspots/sediment quality objectives) | Chis Beegan | BOX 1 | 00180 |
| 2002/03/25 | Post Hearing documents | Bruce F. Reeves | BOX 1 | 00181-00186 |
| 2002/03/29 | Sediment objectives/effluent limits | Chris Beegan | BOX 1 | 00187-00192 |
| 2002/04/15 | Settlement agreement and stipulated order regarding compliance schedule | | BOX 1 | 00193-00198 |
| 2002/04/12 | Settlement agreement and stipulated order regarding compliance schedule | | BOX 1 | 00199-00208 |
| 2002/05/14 | Post Hearing Documents | Bruce Reeves | BOX 1 | 00209-00232 |
| 2002/06/12 | The Distinction Between a TMDL's Numeric Targets and Water Quality Standards | Michael J. Levy | BOX 1 | 00233-00242 |
| 2002/06/21 | TMDLS For condition-based impairments | Michael J. Levy | BOX 1 | 00243-00244 |
| 2002/06/28 | annual costs sediment quality objectives | Chris Beegan | BOX 1 | 00245 |
| 2002/07/12 | Biotic ligand model: a historical overview | | BOX 1 | 00246-00274 |
| 2002/07/17 | I have been asked to survey organizations who have received Cleanup and Abatement Account (CAA) fund | Chris Beegan | BOX 1 | 00275 |
| 2002/10 | SF Bay Draft PCB Food Web model report | Frank Gobas, John Wilcockson | BOX 1 | 00276-00328 |
| 2003 | Request for sole source contract with SAIC to support adoption and implementation of sediment Quality Objectives | Chris Beegan | BOX 1 | 00329-00331 |
| 2003/01/17,31 | Agency Coordination Meeting Minutes, Development of SQOs for California Bays and Estuaries | | BOX 1 | 00332-00338 |
| 2003/01/29 | CAA project expenditure update | Mike Harper | BOX 1 | 00339 |
| 2003/01/30 | effluent limits to protect sediment quality | Bret Betts | BOX 1 | 00340-00341 |
| 2003/01/30 | effluent limits to protect sediemtn quality | Chris Beegan | BOX 1 | 00342-00343 |
| 2003/02 | Summary of SWRCB Program to Develop sediment quality objectives | | BOX 1 | 00344-00349 |
| 2003/02/19 | CAA # 02-148-250-0- Sediment quality objectives | Janice Clemons | BOX 1 | 00350 |
| 2003/02/10 | revised draft workplan | Chris Beegan | BOX 1 | 00353-00362 |
| 2003/03/21 | Revised Workplan to develop sediment quality objectives for enclosed bays and estuaries of California | Chris Beegan | BOX 1 | 00363-00378 |
| 2003/03/21 | draft Revised Workplan to Develop Sediment Quality Objectives for Enclosed Bays and Estuaries | Chris Beegan | BOX 1 | 00379-00380 |
| 2003/04/22 | Faxed Comment Letters | | BOX 1 | 00381-00426 |
| 2003/04/04 | Workplan Comments | Lee, G. Fred | BOX 1 | 00430-00436 |
| 2003/04/22 | Comment Letter on Workplan | Steven Arita | BOX 1 | 00437-00446 |
| 2003/04/22 | SQO Workplan Comments LA County Sanitation Districts | LA County Sanitation Districts | BOX 1 | 00447-00448 |

ADMINISTRATIVE RECORD
FOR SUBMITTAL OF REGULATORY PROVISIONS OF PART 1 OF THE
WATER QUALITY CONTROL PLAN FOR BAYS ESTUARIES

| | | | | |
|---|---|---|---|---|
| 2003/04/22 | SQO Workplan Comments, TriTAC | Tri-TAC, technical advisory group | BOX 1 | 00449-00452 |
| 2003/04/23 | Revised Workplan for Development of SQOs for Enclosed Bays and Estuaries of California Presentation | Chris Beegan | BOX 1 | 00453-00458 |
| 2003/04/01 | Notes: 303(d) Listing | | BOX 1 | 00459-00461 |
| 2003/04/24 | California Sediment Objectives project | mesl@island.net | BOX 1 | 00462-00463 |
| 2003/05/01 | Review of "Revised Work Plan for the Development of Sediment Quality Objectives for Enclosed Bays and Estuaries of California" dated 19 march, 2003 | | BOX 1 | 00464-00473 |
| 2003/05/06 | NorCAL SETAC Annual Meeting Program | | BOX 1 | 00474-00506 |
| 2003/05/06 | Notice of public hearing revised workplan | | BOX 1 | 00507-00508 |
| 2003/05/06 | SQO Workplan Adoption Hearing List | | BOX 1 | 00509-00524 |
| 2003/05/06 | SQO Workplan Adoption Cassette tape of hearing item #1 Side #1 | | BOX 1 | 00525 |
| 2003/05/09 | Draft Workplan | | BOX 1 | 00526-00547 |
| 2003/05/20 | Response to comments SQO Draft Workplan | | BOX 1 | 00548-00583 |
| 2003/05/21 | Response to Department of Toxic Substances Control's (DTSC's) Comments on the State Water Resources Control Board's (SWRCB) Revised Workplan for the Development of Sediment Quality Objectives (SQOs) for California's Enclosed Bays and Estuaries. | Chris Beegan | BOX 1 | 00584-00595 |
| 2003/05/21 | State Water Resources Control Board Proposal for the Development of Sediment Quality Objectives for California enclosed Bays and Estuaries. | James Polisini | BOX 1 | 00596-00608 |
| 2003/05/21 | Consideration of a Resolution Adopting a revised workplan for the development of sediment quality objectives (SQOs) for enclosed Bays and Estuaries of California. | | BOX 1 | 00609-00649 |
| 2003/05/21 | SWRCB meeting and hearing agenda | | BOX 1 | 00650-00669 |
| 2003/06/20 | Sediment Quality Objectives Update | Chris Beegan | BOX 1 | 00670 |
| 2003/06/24 | Organizational meeting for the formation of the Sediment Quality Advisory committee in accordance with California water code section 13394.6 | Celeste Cantu | BOX 1 | 00691-00722 |
| 2003/07/29 | Meeting: Agenda-CASQO Advisory Committee planning meeting | | BOX 1 | 00723-00801 |
| 2003/09/25-2003/10/07 | Agency Coordination Meeting | | BOX 1 | 00802-00807 |
| 2003/09/19 | comments on My Role in the Seiment Quality Advisory Committee | Fredd Lee | BOX 1 | 00808-00820 |
| 2003/09/23 | Participation on Advisory Committee--California Water Code Chapter 5.6. Bay Protection and toxic cleanup | Valerie Nera | BOX 1 | 00821-00822 |
| 2003/09/30 | Invoice, Development SQOs for Enclosed Bays and Estuaries of CA | Southern California Coastal Water Research Project | BOX 1 | 00823 |
| 2003/10/06 | Advisory Committee meeting | | BOX 1 | 00824-00861 |

ADMINISTRATIVE RECORD
FOR SUBMITTAL OF REGULATORY PROVISIONS OF PART 1 OF THE
WATER QUALITY CONTROL PLAN FOR BAYS ESTUARIES

| | | | | |
|---|---|---|---|---|
| 2004/01/05 | Sediment Quality Advisory Committee | Chris Beegan | BOX 1 | 00862-00865 |
| 2004/01/30 | Info Only meeting CaSQO Advisory Committee | | BOX 1 | 00866 |
| 2004/02/12 | Ca SQO Advisory Committee | | BOX 1 | 00867-00904 |
| 2004/02/20 | cleanup and abatement fund | Janice Clemons | BOX 1 | 00905 |
| 2004/02/28 | California Sedmient Quality Objectives Development SF Bay Database Inventory | Chris Beegan | BOX 1 | 00906-00950 |
| 2004/03/03 | Agency Coordination Meeting | | BOX 1 | 00951-00993 |
| 2004/03-2004/04 | Comments on Data inventory | | BOX 1 | 00994-00995 |
| 2004/04/16 | C/A 200 | David Jones | BOX 1 | 00996-00999 |
| 2004/04/22 | Weisberg/Beegan presentation SARWQCB | | BOX 1 | 01000-01013 |
| 2004/04/23 | Sediment Quality Advisory Committee Meeting | Chris Beegan | BOX 1 | 01014-01047 |
| 2004/04/28 | Agnecy Coord. Committee BCDC SF | | BOX 1 | 01048-01052 |
| 2004/04/28 | C/A 200 - Sediment quality objectives | Janice Clemons | BOX 1 | 01053 |
| 2004/04/29 | Agency Coordination Committee San Diego Regional Board | | BOX 1 | 01054 |
| 2004/06/21 | SSC meeting agenda | Steve Bay | BOX 1 | 01055-01080 |
| 2004/05/21 | Bioaccumulation Scope | | BOX 1 | 01081-01098 |
| 2004/05/26 | Preliminary Analysis Phase 1 chronic Sediment test evaluation | | BOX 1 | 01099-01117 |
| 2004/06/02 | SQO science team planning meetin | | BOX 1 | 01118-01122 |
| 2004/06/04 | Sediment Quality Meeting-agenda & directions | Brock Bernstein | BOX 1 | 01123-01133 |
| 2004/06/07 | Request for Executive director approbal of the California sediment quality advisory committee membership | Stan Martinson | BOX 1 | 01134-01136 |
| 2004/06/09 | SQO Progress report and attachments | Steve Bay | BOX 1 | 01137-01162 |
| 2004/06/10 | SQO progress report and attachments | Steve Bay | BOX 1 | 01163-00164 |
| 2004/06/14 | Advisory Committee Meeting | Chris Beegan | BOX 1 | 01165-01200 |
| 2004/06/14 | SQO technical updates and quarterly report | Chris Beegan | BOX 1 | 01201-01225 |
| 2004/06/15 | attendees for the SQAC meeting | | BOX 1 | 01226-01278 |
| 2004/06/15 | cleanup levels resolution | Chris Beegan | BOX 1 | 01279 |
| 2004/06/24 | SQO meeting presentations | Chris Beegan | BOX 1 | 01280-01341 |
| 2004/06/21 | SQO agency coordination committee meetings June 22 Carlsbad, June23 Oakland | Chris Beegan | BOX 1 | 01342-01344 |
| 2004/09/03-2004/09/04 | Sediment Advisory Group Meeting | | BOX 1 | 01345-01362 |
| 2004/07/07 | Workshop Annual Status Report | | BOX 1 | 01363-01369 |
| 2004/07/24 | Chem indicators workplan | | BOX 1 | 01370-01377 |
| 2004/07/26 | Region 9 SD Stormwater permit reissuance | | BOX 1 | 01378-01400 |
| 2004/08/03 | Sediment Quality Agency Coordination Committee Meeting | | BOX 1 | 01401-01522 |
| 2004/08/13 | SSC comments on workplans | Steve Bay | BOX 1 | 01523-01538 |
| 2004/08/10 | SQO Advisory committee meeting | Brock Bernstein | BOX 1 | 01539-01570 |

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WATER QUALITY CONTROL PLAN FOR BAYS ESTUARIES

| | | | | |
|---|---|---|---|---|
| | Blank Page | | BOX 1 | 01571-01572 |
| 2004/05/20 | Dear SAG Members.doc | Doris Vidal | BOX 1 | 01573-01576 |
| 2004/09/29 | Wight of Evidence meeting agenda | Steve Bay | BOX 1 | 01579-01592 |
| 2004/10 | Development of SQOs for CA bays and estuaries | | BOX 1 | 01593-01610 |
| 2004/10/06 | cleanup and abatement fund contract for sediment quality objectives | Linda Puliz | BOX 1 | 01611 |
| 2004/10/18 | Work Plan for: Using Biota Sediment Accumulation Models to Develop Sediment Chemsitry Indicators | | BOX 1 | 01612-01633 |
| 2004/10/19 | Workplan for: Development of Benthic Community Condition Indicators | | BOX 1 | 01634-01651 |
| 2004/10/23 | SQO Development Task Workplans | Chris Beegan | BOX 1 | 01652 |
| 2004/10/27 | SQO Advisory committee meeting | | BOX 1 | 01653-01654 |
| 2004/10/26 | SQO Advisory Meeting Teleconference Number | Chris Beegan | BOX 1 | 01655-01690 |
| 2004/11/01 | CASQO Database draft | Dors Vidal | BOX 1 | 01691 |
| 2004/11/03 | November 4 Bioaccumulation Meeting Presenations | Chris Beegan | BOX 1 | 01692-01759 |
| 2004/12/15 | ecological risk-based screening levels for contaminants in sediments of San Diego Bay, Technical memorandum | Jim A. Bargel | BOX 1 | 01760-01860 |
| 2004/12/13 | revised presentation materials | Chris Beegan | BOX 1 | 01861-01944 |
| 2005 | Text from R. Board Basin Plans Narrative objectives- sed, sed. Tox, toxicity | | BOX 1 | 01945-01948 |
| 2005/01/10 | Report and draft objectives | Steve Bay | BOX 1 | 01949-01953 |
| 2005/01/13 | SQO Committee meeting summary | Brock Bernstein | BOX 1 | 01954-01965 |
| 2005/01/18 | Sediment assessment at the mouths of the Chollas Creek and Paleta Creek, power point | | BOX 1 | 01966-02000 |
| 2005/01/20 | Agency coordination committee meeting roster | | BOX 1 | 02001 |
| 2005/02/01 | CA SWO toxicity workplan for review and project update | Bob Van Dolah | BOX 1 | 02002-02030 |
| 2005/02/03 | work plan comments | Steve Bay | BOX 1 | 02031-02050 |
| 2005/02/08 | Database QA screening | Steve Bay | BOX 1 | 02051-02072 |
| 2005/02/15 | RWQCB meeting R9 Staff presentation | | BOX 1 | 02073-02082 |
| 2005/02/16 | approaches linking whole body fish tissue residues of mercury or DDT | | BOX 1 | 02083-02094 |
| 2005/02/25 | SQO tech team meeting | | BOX 1 | 02095-02148 |
| 2005/02/25 | Comments on Task workplans Tom Gries | Thomas Gries | BOX 1 | 02149-02158 |
| 2005/03 | Enviornmental health coalition survey of fishers on piers in SD bay results and conclusions | | BOX 1 | 02159-02167 |
| 2005/03/18 | CEARS annual meeting | | BOX 1 | 02168-0234 |
| 2005/03/22 | Sarah Newkirk, resignation letter | Sarah G. Newkirk | BOX 1 | 02235 |
| 2005/03/24 | Meeting w/USEPA R IX staff | | BOX 1 | 02236-02248 |
| 2005/03/24 | Revised MLOE workplan | Steve Bay | BOX 1 | 02249-02264 |
| 2005/03/28 | Advisory Committeee meeting call in number and schedule update | Steve Bay | BOX 1 | 02265 |
| 2005/04/01 | SQO meeting summary | Brock Bernstein | BOX 1 | 02266-02345 |

ADMINISTRATIVE RECORD
FOR SUBMITTAL OF REGULATORY PROVISIONS OF PART 1 OF THE
WATER QUALITY CONTROL PLAN FOR BAYS ESTUARIES

| | | | | |
|---|---|---|---|---|
| 2005/03/28 | SQAC meeting, Proposed guidelines for use of sediment quality objectives in the regulation of dredging and disposal operations | | BOX 1 | 02364-02466 |
| 2005/03/30 | Complete EHC pier fishers survey | Laura Hunter | BOX 1 | 02467-02468 |
| 2005/04/01 | SSC meeting agenda and call in number | Steve Bay | BOX 1 | 02469-02628 |
| 2005/04/06 | Env. Meeting caucus summary | | BOX 1 | 02629-02636 |
| 2005/04/20 | Distribution and Persistence of Pyrethroids in Runoff Sediments | | BOX 1 | 02637-02642 |
| 2005/04/26 | Status of SQO-NPDES subcommittee progress | Dave Montagne | BOX 1 | 02643-02647 |
| 2005/04/26 | SQO committee meeting, May 2 | Brock Bernstein | BOX 1 | 02648-02649 |
| 2005/04/28 | FW: SQO database summary information | Brock Bernstein | BOX 1 | 02650-02690 |
| 2005/04/29 | Tenative CAO Contaminated Marine Sediment SD Bay | | BOX 1 | 02691-02724 |
| 2005/05/02 | SQO Advisory Comm. | | BOX 1 | 02725-02746 |
| 2005/05/09 | SFEI Tech Team Bioaccumulation | | BOX 1 | 02747-02759 |
| 2005/05/18 | More Details | Ananda Ranasinghe | BOX 1 | 02760-02786 |
| 2005/06/15 | Data Summary, CA SQO Project | | BOX 1 | 02787-02797 |
| 2005/06/29 | Survey of Applied Taxonomists in SCB | Dave Montagne | BOX 1 | 02798-02804 |
| 2005/06/24 | Public Notice Sediment Quality Workshop Information Only Item July 6 | lyris@swrcb18.waterboards.ca.gov | BOX 1 | 02805-02810 |
| 2005/07/15 | SQAC July 15th meeting agenda | | BOX 1 | 02811-02826 |
| 2005/07/26-28 | SSC meeting agenda | | BOX 1 | 02827-02964 |
| 2005/08/01 | August 10 RB MEETING AGENDA MATERIAL | Mike McCann | BOX 1 | 02965 |
| 2005/08/03 | SQO Completion Proposal | Chris Beegan | BOX 1 | 02966-02968 |
| 2005/08/03 | SQO summary and C&A | Dominic Gregorio | BOX 1 | 02969-02992 |
| 2005/08/03 | Re: August 10 RB meeting presentations | Mike McCann | BOX 1 | 02993 |
| 2005/08/05 | Preliminary Summary of SQOs and General Approach for Implementation | Chris Beegan | BOX 1 | 02994-03006 |
| 2005/08/08 | final proposal | Chris Beegan | BOX 1 | 03007-03009 |
| 2005/08/10 | Region 9 Executive Officer Report | Michael P. McCann, P.E. | BOX 1 | 03010-03027 |
| 2005/08/11 | SD Tribune R9 cleanup article | Chris Beegan | BOX 1 | 03028-03029 |
| 2005/08/22 | Letter to Bruce Reeves from Baykeepers Attorneys | Bruce Reeves | BOX 1 | 03030-03033 |
| 2005/08/26 | Notice of Workshopv SQOs | lyris@swrcb18.waterboards.ca.gov | BOX 1 | 03034-03036 |
| 2005/08/30 | Re: Update on SQGs from the July Meetings | Chris Beegan | BOX 1 | 03037 |
| 2005/09/06 | Response from Bill White to Bruce Reeves | Bill White | BOX 1 | 03038-03041 |

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| | | | | |
|---|---|---|---|---|
| 2005/09/07 | fuding from the State Water Pollution CAA for contract support services for the development of SQOs and authorizing the executive director to negotiate, execute, and amend contracts for the development of SQOs | | BOX 1 | 03042-03044 |
| 2005/09/07 | Closed Session info to board members | | BOX 1 | 03045-03047 |
| 2005/11/07 | Re: dregding and SQOs | Michael Lyons | BOX 1 | 03048 |
| 2005/11/17 | Nest SSC meetign scheduled | Steve Bay | BOX 1 | 03049 |
| 2005/11/22 | SQO meeting agenda | Brock Bernstein | BOX 1 | 03050-03051 |
| 2005/11/30 | SQO meeting materials | Steve Bay | BOX 1 | 03052-03060 |
| 2005/11/30 | SQO meeting call in number | Brock Bernstein | BOX 1 | 03061 |
| 2005/12/07 | Sediment Quailty Objectives Agency Coordination Committee meeting, Agenda | | BOX 1 | 03123-03124 |
| 2006/01/05 | Newport Bay Case Study | Ben Greenfield | BOX 1 | 03125-03126 |
| 2006/01/10 | SQO issues and alternatives | Brock Bernstein | BOX 1 | 03127-03137 |
| 2006/01/18 | Potential Tissue Thresholds For CA SQOs | | BOX 1 | 03138-03150 |
| 2006/01/19 | Validation Framework, Validation of SQO MLOE Assesment Framework | | BOX 1 | 03151-03187 |
| 2006/01/26 | SQAC meeting 13th meeting AGENDA | | BOX 1 | 03188-03234 |
| 2006/01/31 | Sediment quality assessment study | Steve Bay | BOX 1 | 03235-03281 |
| 2006/02/06 | Draft Indirect effects SQOs text partial | | BOX 1 | 03282-03373 |
| 2006/02/08 | Baykeeper complaint, Paul Singerella | Chris Beegan | BOX 2 | 03374-03375 |
| 2006/02/28 | SSC meeting Benthic Indicator Development PPT presentation | | BOX 2 | 03376-03557 |
| 2006/03/20 | Delta Stakeholder Meeting Notice on Sediment Quality Objectives | lyris@swrcb18.waterboards.ca.gov | BOX 2 | 03558-03561 |
| 2006/04/13 | Delta Stakeholder Kickoff Meeting | | BOX 2 | 03562-03584 |
| 2006/04/14 | Slides from April 13, 2006 SQO meeting | Chris Beegan | BOX 2 | 03585-03630 |
| 2006/04/14 | Re: Copies of Slides from Stakeholder Meeting on 4/1/06 | Chris Beegan | BOX 2 | 03631-03676 |
| 2006/04/14 | Monday and Tuesday Sediment Quality Agency Coordination Committee | Chris Beegan | BOX 2 | 03677-03678 |
| 2006/04/14 | Fwd: Final SQO SSC comments | Chris Beegan | BOX 2 | 03679-03686 |
| 2006/04/17-18 | State Water Boards SQO development ACC meeting | | BOX 2 | 03687-03700 |
| 2006/05/12 | SQO Advisory committee meeting date | Brock Bernstein | BOX 2 | 03701 |
| 2006/05/23 | RMP Benthic Assessment Workshop Agenda | | BOX 2 | 03702 |
| 2006/05/24 | SQO meeting w/WSPA on May 25: Agenda | Kevin Buchan | BOX 2 | 03703 |
| 2006/05/25 | Western States Petroleum Association meeting agenda | | BOX 2 | 03704 |
| 2006/05/30 | Sediment Quality Advisory Committee June meeting Schedule | lyris@swrcb18.waterboards.ca.gov | BOX 2 | 03705 |
| 2006/05/30 | Returned mail: permanent problems with the remote server | Gabriel Solmer | BOX 2 | 03706-03714 |
| 2006/06/05 | SQO meeting agenda/ call-in number | Brock Bernstein | BOX 2 | 03715-03717 |
| 2006/06/05 | meeting reminder- sediment Quality objectives | Brock Bernstein | BOX 2 | 03718 |

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|---|---|---|---|---|
| 2006/06/12 | Documents for Fridays SSC conference call | Brock Bernstein | BOX 2 | 03719-03784 |
| 2006/06/17 | Final SQO meeting summary | Brock Bernstein | BOX 2 | 03785-03795 |
| 2006/06/19 | SQO policy development issues | Brock Bernstein | BOX 2 | 03796-03799 |
| 2006/06/19 | SQO meeting agenda | Brock Bernstein | BOX 2 | 03800-03801 |
| 2006/06/19 | Documents for Fridays SSC chemical indicators conference call | Brock Bernstein | BOX 2 | 03802-03825 |
| 2006/06/20 | Delta Advisory meeting roster | | BOX 2 | 03826-03829 |
| 2006/06/22 | PPT for tomorrow's SSC conference call | Brock Bernstein | BOX 2 | 03830-03871 |
| 2006/06/23 | Draft notes from today's conference call | Steve Bay | BOX 2 | 03872-03879 |
| 2006/06/29 | SQO materials to review | Brock Bernstein | BOX 2 | 03880-03900 |
| 2006/07/08 | SQO meeting agenda and materials | Brock Bernstein | BOX 2 | 03901-03942 |
| 2006/07/08 | SQO committee meeting reminder | Brock Bernstein | BOX 2 | 03943-03954 |
| 2006/07/13 | Delta Advisory meetin, SQO Delta | | BOX 2 | 03955 |
| 2006/07/19 | Corrected SQO Delta Committee list | Brock Bernstein | BOX 2 | 03956-03958 |
| 2006/07/24 | SQO flow charts | Lisa Haney | BOX 2 | 03959-03965 |
| 2006/07/25 | Type I errors for Options 1-4, 6, 7 | Steve Saiz | BOX 2 | 03966-03969 |
| 2006/07/25 | Type I errors for Options 1-4, 6, 7 | Kerry Ritter | BOX 2 | 03970-03971 |
| 2006/07/26 | SQO committee meeting summary | Brock Bernstein | BOX 2 | 03972-03980 |
| 2006/07/28 | SSC Conference call comments | Steve Bay | BOX 2 | 03981-03995 |
| 2006/08/17 | State Water Board Staff Scoping Document- Development of Sediment Quality Objectives | Brock Bernstein | BOX 2 | 03996-04000 |
| 2006/08/17 | SQO CEQA Scoping documents | Chris Beegan | BOX 2 | 04001 |
| 2006/08/17 | SQO Scoping Document | | BOX 2 | 04002-04072 |
| 2006/08/18 | Scoping Document for Sedimetn Quality Objectives | lyris@swrcb18.waterboards.ca.gov | BOX 2 | 04073 |
| 2006/08/28 | Water Quality NewsFlash | | BOX 2 | 04074 |
| 2006/08/29 | your 'review' command | Lyris ListManager | BOX 2 | 04075-04086 |
| 2006/09/22 | Sediment Quality Objectives Notice of Status Report Workshop and Scoping Meeting | Chris Beegan | BOX 2 | 04087-04094 |
| 2006/09/28 | SQO advisory committee meeting info | Brock Bernstein | BOX 2 | 04095-04169 |
| 2006/10/01 | Reminder SQO meeting | Brock Bernstein | BOX 2 | 04170 |
| 2006/10/03 | SQO scoping document (correct file) | Brock Berstein | BOX 2 | 04171-04203 |
| 2006/10/05 | SQO call in number | Brock Bernstein | BOX 2 | 04204 |
| 2006/10/05 | SWAMP Roundtable Talk | | BOX 2 | 04205-04212 |
| 2006/10/23 | Scoping Meeting SD regional Board offices | | BOX 2 | 04213-04228 |
| 2006/10/30 | Delta Protection Commision Scoping Comments | Linda Fiack | BOX 2 | 04229-04231 |

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FOR SUBMITTAL OF REGULATORY PROVISIONS OF PART 1 OF THE
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| | | | | |
|---|---|---|---|---|
| 2006/10/30 | Comments on SQO Scoping Document | Lori Clamurro, loridpc@citilink.net | BOX 2 | 04232-04236 |
| 2006/11/08 | Scoping meeting RB2 agenda | | BOX 2 | 04237-04283 |
| 2006/11/16 | Sierra Club Scoping Comments | Ed Kimura | BOX 2 | 04284-04289 |
| 2006/11/21 | City of LA Scoping comments | Rita L. Robinson | BOX 2 | 04290-04308 |
| 2006/11/27 | City of Morro Bay Scoping Comments | Bruce Ambo, Rick Algert | BOX 2 | 04309-04317 |
| 2006/11/27 | USACE SF District Scoping comments | Thomas R. Kendall | BOX 2 | 04318-04319 |
| 2006/11/28 | Scoping Meeting RB5 | | BOX 2 | 04320-04365 |
| 2006/11/28 | CASQA comments on CEQA Scoping Meeting Informational Document--Development of Sediment Quality Objectives for Enclosed Bays and Estuaries | Bill Busath | BOX 2 | 04366-04372 |
| 2006/11/28 | Comment letter- Sediment Quality Objectives | Wendell H. Kido | BOX 2 | 04373-04381 |
| 2006/11/28 | Comment Letter--Sediment Quality Objectives--CEQA Scoping comments of Western States Petroleum Association | Kevin Buchan | BOX 2 | 04382-04414 |
| 2006/11/28 | Comment Letter--Sediment Quality Objectives | Mark Grey | BOX 2 | 04415-04418 |
| 2006/11/28 | Ventura Scoping Comments | Dan Pfeifer | BOX 2 | 04419-04424 |
| 2006/11/28 | Port of Oakland Comment letter-SQOs | Gerald M. Serventi | BOX 2 | 04425-04428 |
| 2006/11/28 | County Sanitation Districts of LA Comments on SQOs | James F. Stahl | BOX 2 | 04429-04433 |
| 2006/11/28 | Nossaman, Guthner, Knox, and Elliott, LLP Comment letter--SQOs | Mary Lynn Coffee, for Nossaman, Guthner, Knox, and Elliott LLP | BOX 2 | 04434-04449 |
| 2006/11/28 | Central Valley Water Board comments on SQOs | Kenneth Landau | BOX 2 | 04450-04455 |
| 2006/11/28 | Bay Keeper coast keepers, heal the bay coastkeeper bay foundation of morrow bay scoping comments | Ed Kimura, Garbriel Solmer, Laura Hunter | BOX 2 | 04456-04475 |
| 2006/11/28 | Comment Letter- SQOs, URS Corp | Usha Vedagiri | BOX 2 | 04476 |
| 2006/11/28 | BACWA scoping comments for SQOs | Michele M Pla | BOX 2 | 04477-04479 |
| 2006/11/28 | Tri-TAC Scoping comments | Chuck Weir | BOX 2 | 04480-04485 |
| 2006/11/28 | Central Valley Clean Water Association comment letter on SQOs | Warren Tellefson | BOX 2 | 04486-04492 |
| 2006/11/28 | California Chamber of Commerce Scoping comments | Valerie Nera | BOX 2 | 04493-04560 |
| 2006/11/28 | LADWP scoping comments | Katherine Rubin | BOX 2 | 04561-04564 |
| 2006/11/28 | City of San Diego Scoping Comments | Chris Zirkle | BOX 2 | 04565-04567 |

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| | | | | |
|---|---|---|---|---|
| 2006/11/29 | USACE- LA district scoping comments | Ruth Bajza Billalobos | BOX 2 | 04568-04572 |
| 2006/12/01 | Sediment Quality Objectives Comment Letters | lyris@swrcb18.waterboards.ca.gov | BOX 2 | 04573-04574 |
| 2006/12/06 | Approval of subcontractors for development of sediment quality objectives | Darrin Polhemus | BOX 2 | 04575-04576 |
| 2006/12/15 | Jan 12_2007 CASQA General Meeting Agenda_Dec 7 06 | Karen Ashby KarenA@LWA.com | BOX 2 | 04577-04579 |
| 2006/12/18 | summary of statutes of sediment quality objectives project | Dominic Gregorio | BOX 2 | 04580-04586 |
| 2007 | Indicator Development and Framework for Assessing Indirect Effects of Sediment Contaminants | | BOX 2 | 04587-04846 |
| 2007/01/17 | SCCWRP 01-274-250-1 contract approval | Wendy Johnson | BOX 2 | 04847-04861 |
| 2007/02/26 | Re: Regulatory baseline | Michael Lyons | BOX 2 | 04862-04867 |
| 2007/03/05 | SFEI DRAFT Indirect Effects Tech Report | | BOX 2 | 04868-05097 |
| 2007/03/06 | Corrected SQO meeting summary | Brock Bernstein | BOX 2 | 05098-05104 |
| 2007/03/06 | SQO Summary and issues discussion document | Chris Beegan | BOX 2 | 05105-05108 |
| 2007/03/08 | Regulatory Baseline | Michael Lyons | BOX 2 | 05109-05110 |
| 2007/03/09 | SQO technical report | lyris@swrcb18.waterboards.ca.gov | BOX 2 | 05111 |
| 2007/03/19 | Supplemental Agreement | | BOX 2 | 05112-05122 |
| 2007/03/28 | SQO materials | Brock Bernstein | BOX 2 | 05123-05227 |
| 2007/03/28 | SQO materials | Brock Bernstein | BOX 2 | 05228-05236 |
| 2007/03/30 | SQO Survey | Naomi Feger | BOX 2 | 05237-05240 |
| 2007/04 | Public Scoping Meeting Proposed Wetland and Riparian Area Protection | | BOX 2 | 05241-05260 |
| 2007/04/02 | Delta Sediment Monitoring | Daniel McClure | BOX 2 | 05261 |
| 2007/04/04 | SQO Survey | Naomi Feger | BOX 2 | 05262-05265 |
| 2007/04/09 | Upcomiong SQO meetings | lyris@swrcb18.waterboards.ca.gov | BOX 2 | 05266 |
| 2007/04/06 | Regulatory Baseline | Jerry Burns | BOX 2 | 05267-05270 |
| 2007/04/12 | Draft Work Plan | Eloise T Castillo, | BOX 2 | 05271-05274 |
| 2007/04/13 | Agenda for July SSC meeting | Steve Bay | BOX 2 | 05275-05276 |
| 2007/04/16 | SQO meeting materials | Brock Bernstein | BOX 2 | 05277-05291 |
| 2007/04/16 | SQO meeting | Brock Bernstein | BOX 2 | 05292-05307 |
| 2007/04/19 | Delta LTMS: Permitting TWG agenda for 4/27/07 | Jessica L Burton Evans | BOX 2 | 05308-05313 |
| 2007/04/19 | Index comparison paper | Ana Ranasinghe | BOX 2 | 05314-05375 |
| 2007/04/20 | Public Record Request SQOs | Darrel Ramsey | BOX 2 | 05376-05378 |

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| | | | | |
|---|---|---|---|---|
| 2007/04/23 | SQO list of chemicals | Brock Bernstein | BOX 2 | 05379-05380 |
| 2007/04/25 | SQO committee meeting summary | Brock Bernstein | BOX 2 | 05381-05400 |
| 2007/04/25 | SQO Advisory (stakeholder) committee meetings summary | | BOX 2 | 05401 |
| 2007/05/01 | sediment quality reg Baseline | Craig Carlisle | BOX 2 | 05402-05427 |
| 2007/05/07 | cleanup comment | Eloise T Castillo | BOX 2 | 05428-05431 |
| 2007/05/08 | FYI: Exec sighned "consolidated Statewide Toxic Hot Spots Cleanup Plan" letters | Jan Hisao | BOX 2 | 05432-05444 |
| 2007/05/10 | baykeeper agreement | Chris Beegan | BOX 2 | 05445-05456 |
| 2007/05/15 | econ workplan comments | Chris Beegan | BOX 2 | 05457-05460 |
| 2007/05/15 | Fwd: Re: Sediment Quality Objectives status | Chris Beegan | BOX 2 | 05461-05490 |
| 2007/05/17 | WPHA Submission of Comments Concerning Phase-II | Dean Nasser | BOX 2 | 05491-05504 |
| 2007/05/30 | scoping document comments | Chris Beegan | BOX 2 | 05505 |
| 2007/06/05 | SQO Scientific Steering Committee Meeting | Chris Beegan | BOX 2 | 05506 |
| 2007/06/06 | SQO Benthic Indicators Reports for SSC review | Steve Bay | BOX 2 | 05507-05547 |
| 2007/06/12 | update | Eloise T Castillo | BOX 2 | 05548-05571 |
| 2007/06/12 | Benthic Reports | Peter Landrum | BOX 2 | 05572-05587 |
| 2007/06/20 | SQO dredging language comments | Brock Bernstein | BOX 2 | 05588-05594 |
| 2007/06/20 | Meeting | Mike Connor | BOX 2 | 05595 |
| 2007/06/20 | Re: sediment quality objectives in the delta | Dominic Gregorio | BOX 2 | 05596-05597 |
| 2007/06/20 | SQO dredging language suggestion | Brock Bernstein | BOX 2 | 05598-05604 |
| 2007/06/20 | SQO dredging language comments | Brock Bernstein | BOX 2 | 05605-05611 |
| 2007/06/22 | Re: sediment quality objectives in the Delta | Dominic Gregorio | BOX 2 | 05612-05613 |
| 2007/06/25 | Update of Sediment Quality Objectives | Bruce Fujimoto | BOX 2 | 05614-05615 |
| 2007/06/29 | SQO Scientific Steering Committee Meeting Agenda | lyris@swrcb18.waterboards.ca.gov | BOX 2 | 05616-05617 |
| 2007/07/02 | SQO SSC mtng agenda | Brock Bernstein | BOX 2 | 05618-05620 |
| 2007/07/02 | SQO SSC mtng Indirect Effects Report | Brock Bernstein | BOX 2 | 05621-05622 |
| 2007/07/02 | SQO SSC mtng Indirect Effects Report | Brock Bernstein | BOX 2 | 05623-05624 |
| 2007/07/02 | SQO SSC mtng agenda | Brock Bernstein | BOX 2 | 05625-05627 |
| 2007/07/03 | SSC meeting logistics: Hotel reservations and final agenda | Steve Bay | BOX 2 | 05628-05630 |
| 2007/07/10 | SSC Meeting agenda presentations & attendance sheet | | BOX 2 | 05631-05690 |
| 2007/08/01 | RE: FW: Invitation t o Organize a Session for the 2008 Monterey Conference | Bart Chadwick | BOX 2 | 05691-05694 |
| 2007/08/08 | Final Meeting notes | Peter Landrum, Peter.Landrum@noaa.gov | BOX 2 | 05695-05701 |
| 2007/08/08 | New SQO SSC members and next SSC meeting | Steve Bay, steveb@sccwrp.org | BOX 2 | 05702-05704 |

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| | | | | |
|---|---|---|---|---|
| 2007/07/30 | WPHA meeting on 8/15 | Nasser Dean, nasserd@healthyplants.org | BOX 2 | 05705-05716 |
| 2007/08/31 | Agenda for September 18 Board meeting | lyris@swrcb18.waterboards.ca.gov | BOX 2 | 05717-05719 |
| 2007/09/13 | Corrections to thresholds in draft policy | Steve Bay, steveb@sccwrp.org | BOX 2 | 05720-05721 |
| 2007/09/18 | SQO Status Report Workshop agenda & ppt | | BOX 2 | 05722-05738 |
| 2007/10/09 | Peer Review Request Memo and Attachments | Dominic Gregorio | BOX 2 | 05739-06191 |
| 2007/10/10 | Web posting/lyris announcement: SCCWRP Sediment Quality Technical Reports and Database | lyris@swrcb18.waterboards.ca.gov | BOX 2 | 06192 |
| 2007/10/15 | Request for SSC Review Framework for Assessing MLOE Sed Quality Triad Data | Steve Bay | BOX 2 | 06193-06233 |
| 2007/10/18 | Sediment Quality Related Technical Reports | lyris@swrcb18.waterboards.ca.gov | BOX 2 | 06234 |
| 2007/11/07 | Comment letter: Proposed Water Quality Control Plan for Enclosed Bays and Estuaries of California, Sediment Quality Objectives | Valerie Nera | BOX 2 | 06235-06238 |
| 2007/11/13 | Comment Letter | Dee Ann Staats | BOX 2 | 06239-06269 |
| 2007/11/15 | Request for time extension on comment period for draft Sediment Quality Plan | Dorothy Rice | BOX 2 | 06270-06272 |
| 2007/11/19 | Public Hearing for the Draft Water Quality Contro Plan for Enclosed Bays and Estuaries Part 1 Sediment Quality | Jacqueline Toliver | BOX 2 | 06273-06339 |
| 2007/11/19 | SWRCB SQO Hearing Sign-in Sheets | | BOX 2 | 06340-06352 |
| 2007/11/19 | SWRCB SQO Hearing Speaker Cards | | BOX 2 | 06353-06365 |
| 2007/11/25 | Comment Letter - Weston | David W. Moore | BOX 2 | 06366-06403 |
| 2007/11/28 | Comment Letter - Sierra Club | Edward Kimura | BOX 2 | 06404-06409 |
| 2007/11/28 | Comment Letter - OCSD | Michael Moore | BOX 2 | 06410-06412 |
| 2007/11/29 | Comment Letter - SRCSD | Wendell H. Kido | BOX 2 | 06413-06417 |
| 2007/11/29 | Comment Letter - LACo | Donald L. Wolfe | BOX 2 | 06418-06421 |
| 2007/11/29 | Comment Letter - RB5 | Kenneth D. Landau | BOX 2 | 06422-06437 |
| 2007/11/29 | Comment Letter - OCRDMD | Chris Crompton | BOX 2 | 06438-06448 |
| 2007/11/29 | Comment Letter - CVCA | Debbie Webster | BOX 2 | 06449-06454 |
| 2007/11/29 | Comment Letter - LACSD | Stephen R. Magun | BOX 2 | 06455-06470 |
| 2007/11/29 | Comment Letter - JLB | James L. Byard | BOX 2 | 06471-06482 |
| 2007/11/30 | Comment Letter - Caltrans | G. Scott McGowen | BOX 2 | 06483-06488 |

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|---|---|---|---|---|
| 2007/11/30 | Comment Letter - WPHA | Nasser Dean | BOX 2 | 06489-06499 |
| 2007/11/30 | Comment Letter - Tri-TAC/CASA | Jim Colston | BOX 2 | 06500-06507 |
| 2007/11/30 | Comment Letter -SCV | Adam W. Olivieri | BOX 2 | 06508-06509 |
| 2007/11/30 | Comment Letter - HTB | Kirsten James, Mark Gold | BOX 2 | 06510-06514 |
| 2007/11/30 | Comment Letter - BACWA | Michelle Pla | BOX 2 | 06515-06521 |
| 2007/11/30 | Comment Letter - IEA | Patti Krebs | BOX 2 | 06522-06533 |
| 2007/11/30 | Comment Letter - FLI | Susan C. Paulsen | BOX 2 | 06534-06551 |
| 2007/11/30 | Comment Letter - CASQA | Bill Busath | BOX 2 | 06552-06563 |
| 2007/11/30 | Comment Letter - SFBK | Jen Kovecses | BOX 2 | 06564-06587 |
| 2007/11/30 | Comment Letter - WSPA | Kevin Buchan | BOX 2 | 06588-06627 |
| 2007/11/30 | Comment Letter - CCOC | Valerie Nera | BOX 2 | 06628-06778 |
| 2007/11/30 | Comment Letter - LW | Kelly E. Richardson | BOX 3 | 06780 |
| 2007/11/30 | Comment Letter - SDCK | Gabriel Solmer | BOX 3 | 08310-08319 |
| 2007/11/30 | Comment Letter - CLTNS | Tim Johnson, Parry Klassen, L. Ryan Broddrick | BOX 3 | 08320-08321 |
| 2007/11/30 | Comment Letter - LADWP | Katherine Rubin | BOX 3 | 08322-08326 |
| 2007/12/02 | Peer Review Comments David Sedlak | David L. Sedlak | BOX 3 | 08327-08331 |
| 2007/12/13 | SQO review by Schaffner | Linda C. Schaffner, linda@vims.edu | BOX 3 | 08332-08338 |
| 2007/12/14 | Review of Sediment Quality Objectives, John Knexovich | John Knezovich, knexovich1@linl.gov | BOX 3 | 08339-08344 |
| 2007/12/26 | Peer Review, Sediment Quality Objectives | Dominic Gregorio | BOX 3 | 08345 |
| 2008/01/10 | February 5 Board Meeting | lyris@swrcb18.waterboards.ca.gov | BOX 3 | 08346 |
| 2008/01/24 | SQO Agenda February 5 Board Meeting | lyris@swrcb18.waterboards.ca.gov | BOX 3 | 08347-08349 |
| 2008/01/25 | Revised Draft Part 1SQOs | lyris@swrcb18.waterboards.ca.gov | BOX 3 | 08350-08385 |
| 2008/01/30 | SQOs draft final Staff Report | lyris@swrcb18.waterboards.ca.gov | BOX 3 | 08386-08593 |

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|---|---|---|---|---|
| 2008/02/01 | Revised Draft Part I SQOs | lyris@swrcb18.waterboards.ca.gov | BOX 3 | 08594-08629 |
| 2008/02/04 | SQO Item 7 February 2 Agenda Item | lyris@swrcb18.waterboards.ca.gov | BOX 3 | 08630 |
| 2008/02/04 | Clarification SQO Item 7 on the February 5 Agenda | lyris@swrcb18.waterboards.ca.gov | BOX 3 | 08631-08635 |
| 2008/02/05 | Board Agenda, February 5 | | BOX 3 | 08636-08637 |
| 2008/02/14 | SQOs Responses to Comments Draft Document | lyris@swrcb18.waterboards.ca.gov | BOX 3 | 08638-08841 |
| 2008/02/15 | Fwd: If chemistry is missing | Chris Beegan | BOX 3 | 08842-08856 |
| 2008/02/17 | Revised Draft Part 1 | lyris@swrcb18.waterboards.ca.gov | BOX 3 | 08857-08898 |
| 2008/02/19 | SWRCB Board Meeting SQO Adoption Video DVD 1 & 2 | Tam Doduc | BOX 3 | 08899 |
| **REFERENCE MATERIAL** | | | | |
| 1946/06 | On the theory of scales and measurement | | BOX 3 | 08900-08904 |
| 1971/07 | the nonconcept of species | | BOX 3 | 08905-08920 |
| 1975 | Scales and Statistics | | BOX 3 | 08921-08935 |
| 1985 | Guidelines for deriving numerical water quality criteria for aquatic organisms | | BOX 3 | 08936-09040 |
| 1990 | The Sediment Quality Triad Approach to Determining Pollution-Induced Degradation | | BOX 3 | 09041-09051 |
| 1990 | Comparative Evaluation of Five Toxicity Tests with Sediments from San Francisco Bay and Tomales Bay | Long, Edward M. Buchman, Steve M.Bay, Ronald Breteler, R.Scott Carr, Peter Chapan, et al | BOX 3 | 09052-09073 |
| 1990 | The Potential for Biological Effects of Sediment Sorbed Contaminants Tested in thwe National Status and Trends Program | Long, E.R. and L.R. Morgan | BOX 3 | 09074-09305 |
| 1990 | The Sediment Quality TRIAD Approach for Determining Pollution-Induced Degradation | Chapman, Peter | BOX 3 | 09306-09316 |
| 1990/12 | Contaminated Sediments, Relevant statutes and EPA program activities | | BOX 3 | 09317-09467 |
| 1991/02 | Evaluation of dredged material proposed for Ocean Disposal | | BOX 3 | 09468-09680 |
| 1991/06 | SQO Workplan | | BOX 3 | 09681-09716 |
| 1992/09 | Sediment Classification Methods compendium | | BOX 3 | 09817-10044 |
| 1991/04 | Technical Support Document for Water Quality Based Toxics control | | BOX 4 | 10045-10379 |
| 1994 | Water Quality Control Plan For the San Diego Basin | | BOX 4 | 10380-10448 |
| 1994/06 | Methods for Assessing the toxicity of sediment-associated contaminants with estuarine and marine amphipods | | BOX 4 | 10449-10605 |

ADMINISTRATIVE RECORD
FOR SUBMITTAL OF REGULATORY PROVISIONS OF PART 1 OF THE
WATER QUALITY CONTROL PLAN FOR BAYS ESTUARIES

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|---|---|---|---|---|
| 1994/07 | Toxic Cleanup Program Quality Assurance Project Plan | | BOX 4 | 10606-10762 |
| 1995 | Incidence of Adverse Biological Effects within Ranges of Chemical Concetrations in Marine and Estuarine Sediments | Long, Edward, Don Macdonald, Sherri Smith, Fred Calder | BOX 4 | 10763-10779 |
| 1995/03 | EPA Water Quality Guidance for the Great Lakes System: SID | | BOX 4 | 10780-10828 |
| 1995/03 | EPA Great Lakes Water Quality Initiative Criteria | | BOX 4 | 10829-10877 |
| 1995/07 | Recommended guidelines for conducting laboratory bioassays on Puget Sound sediments | | BOX 4 | 10878-10966 |
| 1995/11/16 | Water Quality Control policy for the enclosed bays and estuaries of CA | | BOX 4 | 10967-10983 |
| 1995/12/29 | Washington Dept. of Ecology, Chapter 173-204 Wash. Admin. Code, Sed. Management Standards | | BOX 4 | 10984-11011 |
| 1996 | Assessment of Sediment toxicity at the sediment toxicity at the sed-water interface | | BOX 4 | 11012-11027 |
| 1996 | Sediment quality in Florida coastal waters | | BOX 4 | 11028-11033 |
| 1996/06 | EMAP/NS&T Pilot studies in the Carolinian Province: Indicator Testing and Evaluation in the Southeastern Estuaries | | BOX 4 | 11034-11158 |
| 1996/09 | Chem, toxicity and benthic community conditions in sediments of the San Diego Bay region | | BOX 4 | 11159-11280 |
| 1997/03 | An Estuarine Benthic Index of Biotic Integrity for Chesapeake Bay Estuaries Vol 20. No 1 p 149-158 | | BOX 4 | 11281-11300 |
| 1997/05 | Chem. Toxicity, and benthic community Conditions in sediments of selected So. CA Bays and Estuaries | | BOX 4 | 11301-11371 |
| 1998 | Predicting Toxicity in Marine Sediments with Numerical Sediment Quality guidelines | | BOX 4 | 11372-11385 |
| 1998/12 | So. CA Bight 1994 pilot project VI. Sed. Tox | | BOX 4 | 11386-11450 |
| 2001/04 | Total Maximum Daily Loads (TMDL) Question and Answers | Staff | BOX 4 | 11451-11452 |
| 1998 | Comparrison of marine sediment toxicity test protocols for... | | BOX 4 | 11453-11460 |
| 1998/02 | So. CA Bight 94 pilot project Volume IV. Benthic Infauna | | BOX 4 | 11461-11499 |
| 1998/02 | EPA/USACE Evaluation of Dredged Material Prposed for Discharge in Waters of the U.S. (Inland Testing Manual) | EPA/USACE | BOX 4 | 11500-11667 |
| 1998/04 | EPA's Contaminated Sediment Management Strategy | | BOX 4 | 11668-11796 |
| 1998/08 | Sediment Quality and Biological effects in SF bay final tech report | | BOX 4 | 11797-11898 |
| 1998/08 | Sediment Chemistry toxicity and Benthic community conditions in selected Water Bodies of thwe Los Angeles Region, final report | | BOX 4 | 11899-11998 |
| 1998/11/30 | EPA federal register, hazardous remediation waste management | | BOX 4 | 11999-12023 |

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|---|---|---|---|---|
| 1998/07/05 | Swartz Consensus SQGs Guidelines for Polycyclic Aromatic Hydrocarbon Mixtures | | BOX 4 | 12024-12031 |
| 1998/09/11 | SETAC Press evaluating Sed Chem and Tox data using logistic regression | | BOX 4 | 12032-12037 |
| 1999/02 | Marine Env. Research benthic index of biological integrity | | BOX 4 | 12038-12052 |
| 1999/02 | assessing sediment contamination in estuaries | | BOX 4 | 12053-12072 |
| 1999/06 | Prevalence of Selected Target Chemical Contaminants in Sport Fish from Two CA Lakes | | BOX 4 | 12073-12100 |
| 1999/08 | Toxicity reduction evaluation guidance for municipal wastewater treatment plants | EPA | BOX 4 | 12101-12197 |
| 1999/09 | development and evaluation of consensus-based sediment effect concentrations for PCB | | BOX 4 | 12198-12208 |
| 2000 | A large-scale categorization of sites in San Francisco Bay, USA, Based on the Sediment Quality Triad, Toxicity indentification evaluations, and gradient studies. | | BOX 4 | 12209-12222 |
| 2000 | Development and Eval of consensus-based sed. Effect concentrations for Polychlorinated Biphenyls | | BOX 4 | 12223-12233 |
| 2000 | Classifying probabilities of acute tox in marine sed with empirically derived sed qual guidelines | | BOX 4 | 12234-12237 |
| 2000 | Assessment of benthic infaunal condition on the mainland shelf of Southern California | | BOX 4 | 12238-12251 |
| 2000/02 | Bioaccumulation Testing and Interpretation for the purpose of sediment quality assessment | | BOX 4 | 12252-12386 |
| 2000/02 | Estuaries, relationships between benthi community condition, water quality, sediment quality etc. | | BOX 4 | 12387-12403 |
| 2000/05 | EPA evaluation guidelines for ecological indicators | | BOX 4 | 12404-12492 |
| 2000/11 | Guidance for assesing chemical contaminant data for use in fish tissue advisories Vol 1 Fish Sampling | | BOX 4 | 12493-12922 |
| 2000/12 | Estuarine and Coastal Marine Waters: Bioassessment and Biocriteria Technical Guidance | | BOX 4 | 12923-13018 |
| 2000/12 | Sediment quality targets and sediment contamination in the St. Lousi River Area of Concern | | BOX 4 | 13019-13142 |
| 2001 | A large-scale categorization of sites in SF Bay, USA, based on the sed. Quality triad, tox. Identification evals and gradient studies | | BOX 4 | 13143-13156 |
| 2001 | Relationships between acute sediment toxicity in laboratory tests and abundance and diversity of benthic infauna in marine sediments: A review | | BOX 4 | 13157-13185 |
| 2001 | Relationshi[p Between Acute Sediment Toxicity in Laboratory Tests and Abundance and Diversity of Benthic Infauna in Marine Sediments; A Review | Edward Long, Carolyn Hong, Corinne Severn | BOX 4 | 13185-13157 |

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|---|---|---|---|---|
| 2001/02/18 | An Evaluation of Methods for Calculating mean sediment quality guideline quotients as indicators of contamination and acute toxicity to amphipods by chemical mixtures | | BOX 4 | 13186-13196 |
| 2001/03 | Method for Assessing the Chronic Toxicity of Marine and Esturaine Sediment-associated Contaminants with the Amphipod Leptocheirus plumulosus 1st ed. | | BOX 4 | 13197-13310 |
| 2001/06 | National Coastal Assessment, Field Operations Manual | | BOX 5 | 13311-13387 |
| 2001/08 | Benthic Response Index for Assessing Infaunal Communities on the Southern California Mainland Shelf | | BOX 5 | 13388-13403 |
| 2002 | Use of Sediment Qaulity Guidelines and Related Tools Executive Summary of the SETAC Pellston Workshop | | BOX 5 | 13404-13447 |
| 2002 | Weight of Evidence Approaches for Assessing Ecosystem Impairment | | BOX 5 | 13448-13464 |
| 2002 | A Weight-of-Evidence Framework for Assessing Sediment (Or Other) Contamination: Improving Certainty in the Decision-Making Process | | BOX 5 | 13465-13486 |
| 2002 | overview of toxicant identification in sediments and dredged materials | | BOX 5 | 13487-13494 |
| 2002/03 | Model monitoring program for large ocean discharges in southern CA | | BOX 5 | 13495-13599 |
| 2002/03/03 | Predicting Amphipod Toxicity From Sediment Chemistry Using Logistic Regression Models | | BOX 5 | 13600-13612 |
| 2002/09 | Aquatic Stressors, framework and implementation plan for effects research | | BOX 5 | 13613-13809 |
| 2002/12 | EPA guidance manual to support the assessment of contaiminated sediments in freshwater ecosystems vol. I | | BOX 5 | 13810-13909 |
| 2002/12 | EPA guidance manual to support the assessment of contaiminated sediments in freshwater ecosystems vol. III | | BOX 5 | 13910-14012 |
| 2003/01 | Development and Evaluation of Numerical Sediment Quailty Assessment Guidelines for Florida Inland Waters | | BOX 5 | 14013-14162 |
| 2003/03 | Benthic Macrofauna | Ranasinghe, Montagne et al. | BOX 5 | 14163-14261 |
| 2003/08 | EPA Procedures for the Derivation of Equilibrium Partitioning Sediment Benchmarks (ESBs) Dieldrin | | BOX 5 | 14262-14336 |
| 2003/11 | EPA Procedures for the Derivation of Equilibrium Partitioning Sedimetn Benchmarks, PAH mixtures | | BOX 5 | 14337-14511 |
| 2004 | Assessment of Macrobenthos response to sediment contamination in the San Francisco Estuary, CA USA | | BOX 5 | 14512-14521 |
| 2004 | EPA Procedures for the Derivation of ESBs for the Protection of Benthic Organisms: Metal Mixtures | | BOX 5 | 14522-14642 |
| 2004 | Marine quality assessment by use of benthic species-abundance distributions: a proposed new protocol within the European Union Water Framework Directive | Rosenberg et al | BOX 5 | 14643-14666 |
| 2004/01/22 | Amended Consolidated Hotspots Cleanup Plan Functional Equivalent Document | Tam Dudoc | BOX 5 | 14667-15149 |

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|---|---|---|---|---|
| 2004/04 | Evaluation of Benthic Assessment Methodology in Southern California Bays and San Francisco Bay | | BOX 5 | 15150-15227 |
| 2004/09 | Final Functional Equivalent Document Water Quality Control Policy for Developing Californias Clean Water Act Section 303(d) List | SWRCB | BOX 5 | 15228-15354 |
| 2004/09/30 | Water Quality Control Policy 303(d) | | BOX 5 | 15355-15388 |
| 2004/11 | The incidence and severity of sed contamination in surface waters of the United States... | | BOX 5 | 15389-15668 |
| 2006/07 | Washington State Department of Ecology Water Quality Program Permit Writers Manual | Gary Bailey | BOX 5 | 15669-15726 |
| 2005 | Policy for Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries of CA | | BOX 5 | 15727-15771 |
| 2005 | A Decision-Making Framework for Sed. Contamination | | BOX 5 | 15772-15782 |
| 2005 | Water Quality Control Plan for Ocean Waters of California California Ocean Plan | Tam Doduc | BOX 5 | 15783-15837 |
| 2005 | Contaminated Sediments Research Plan, multi-year plan | | BOX 5 | 15838-15939 |
| 2005/01/06 | Draft SQ guidelines for Contaminated sediments in LA area | | BOX 5 | 15940-15956 |
| 2005/05 | Sediment Assessment Study for the Mouths of Chollas and Paleta Creek, San Diego | SSCWRP and SPAWAR | BOX 5 | 15957-16172 |
| 2006 | Selecting Discriminant Function Models for Predicting the Expected Richness of Aquatic Macroinvertebrates | Van Sickle, John, David D. Huff, Charles Hawkins | BOX 5 | 16173-16186 |
| 2006 | Calculation and Uses of Mean Sediment Quality Guidilne Quotients: A Critical Review | Long, Edward, Christopher Ingersoll, Donald D. Macdonald | BOX 5 | 16187-16197 |
| 2006/04 | The regional monitoring program: science in support of managing water quality in the SF estuary | | BOX 5 | 16198-16245 |
| 2006/05/31 | Contaminated Sediments Breakout Session | | BOX 5 | 16246-16295 |
| 2006/08/01 | Mercury in SF Bay, proposed basin plan amendment and Staff report | | BOX 5 | 16296-16411 |
| 2006/08/17 | CEQA Scoping Meeting Informational Document Development of Sediment Quality Objectives for Enclosed Bays and Estuaries | chris beegan | BOX 5 | 16412-16485 |
| 2007 | Evaluating Consistency of Best Professional Judgment in the Application of a MLOE Sed. Quality Triad | | BOX 6 | 16486-16492 |
| 2007/01/18 | San Francisco Bay Basin (Region 2) Water Quality Control Plan (Basin Plan) | SFRWQCB | BOX 6 | 16493-16561 |
| 2007/01 | Water Quality Control Plan For the North Coast Region | | BOX 6 | 16562-16597 |
| 2007/03 | Evaluation of methods for measuring sediment toxicity in CA bays and Estuaries | | BOX 6 | 16598-16789 |

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|---|---|---|---|---|
| 2007/03/22 | Marina Del Ray Harbor Toxic Pollutant | Technical Committee County of Los Angeles Chair | BOX 6 | 16790-17010 |
| 2007/05 | So. CA Bight 2003 Regional Monitoring program III. Benthic Macrofauna | | BOX 6 | 17011-17076 |
| 2007/06/22 | Total Maximum Daily Load for PCBs in San Francisco Bay | SFRWQCB | BOX 6 | 17077-17126 |
| 2007/09/18 | Economic Consideration of Proposed Sediment Quality Plan for Enclosed Bays in California | | BOX 6 | 17127-17276 |
| 2007/10/15 | A Framework for Interpreting Sediment Quality Triad Data | | BOX 6 | 17277-17299 |
| 2007/10/24 | Comparison of National and regional sediment quality guideilnes for predicting ... | | BOX 6 | 17300-17322 |
| 2007/11/14 | Development and Evulation of Sediment Quality Guidelines based on Benthic Macrofauna Responses | | BOX 6 | 17323-17348 |
| 2008/02/29 | Submittal of Regulatory Provisions | Liz Haven | BOX 6 | 17349-17354 |
| 2008/02/19 | Water Quality Control Plan for Enclosed Bays nad Estuaries Part 1 Sediment Quality | Tam Doduc | BOX 6 | 17355-17389 |

**Mentink, Dale@OAL**

**From:** Dale P. Mentink
**Sent:** Wednesday, March 19, 2008 2:58 PM
**To:** 'Chris Beegan'
**Subject:** RE: Sediment Quality, Part 1, OAL file no. 2008-0229-07S

Thank you very much. I never asked you this on the phone, but if minutes or a tape or DVD of the 2/5 hearing is something we need, was a recording of some kind made, or is there no recording of it? Thanks, Dale

-----Original Message-----
From: Chris Beegan [mailto:cbeegan@waterboards.ca.gov]
Sent: Wednesday, March 19, 2008 2:46 PM
To: Dale P. Mentink
Subject: Re: Sediment Quality, Part 1, OAL file no. 2008-0229-07S

Hi Dale: Thank you for discussing the record with me today. I have created a smaller spreadsheet (Key to comment letters) that encompasses just the comment letters and the affiliation that is referenced in the responses to comments document. I have also amended the index with the affiliations and highlighted those in yellow.

Chris Beegan
SWRCB
Division of Water Quality
Sediment Quality Objectives Development
1001 I Street, Sacramento Ca 95814
cbeegan@waterboards.ca.gov
Office (916) 341-5577
Cell (916) 955-9262

>>> "Dale P. Mentink" <dmentink@oal.ca.gov> 3/19/2008 9:31 AM >>>
Hello, my name is Dale Mentink. I'm with the Office of Administrative Law and am reviewing this file. I'm sorry if I'm unable to find something obvious in the boxes, but I'm trying to do a sampling of comments and responses review. I have the oral comment summary and responses at pages 08639 through 08656. Is there a tape, minutes, or DVD of that testimony that I can review to hear or see the comments? I have a DVD of the 2/19/08 meeting (p. 08899), but I believe the testimony was taken on 2/5/08.

Also with regard to the summary and response to written comments at pages 08657 through 08841, is there a way to determine which document in the file contains the comments of No. 1, or No. 102, or No. 499 (e.g., page 08657) and so forth?

Thank you very much for your assistance.

Dale Mentink

Senior Staff Counsel

Office of Administrative Law

300 Capitol Mall, Suite 1250

Sacramento, CA 95814

(916) 323-6817

FAX (916) 323-6826

dmentink@oal.ca.gov

**Mentink, Dale@OAL**

**From:** Chris Beegan [cbeegan@waterboards.ca.gov]
**Sent:** Thursday, March 20, 2008 9:05 AM
**To:** Dale P. Mentink
**Subject:** RE: Sediment Quality, Part 1, OAL file no. 2008-0229-07S

Good morning Dale: I had the clerk of the Board make a DVD of the 2/5/08 Workshop. I don't know how you want to handle this, but if you want to view it, I can drop it by after my 10am meeting is over.

>>> "Dale P. Mentink" <dmentink@oal.ca.gov> 3/19/2008 2:57 PM >>>
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Senior Staff Counsel

Office of Administrative Law

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Sacramento, CA 95814

(916) 323-6817

FAX (916) 323-6826

dmentink@oal.ca.gov

**Mentink, Dale@OAL**

**From:** Dale P. Mentink
**Sent:** Thursday, March 20, 2008 9:50 AM
**To:** 'Chris Beegan'
**Subject:** RE: Sediment Quality, Part 1, OAL file no. 2008-0229-07S

Or just mail it; it's not something that has to happen today or even this week. Thank you.
Dale Mentink, OAL

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From: Chris Beegan [mailto:cbeegan@waterboards.ca.gov]
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Dale Mentink

Senior Staff Counsel

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Sacramento, CA 95814

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FAX (916) 323-6826

dmentink@oal.ca.gov

**Mentink, Dale@OAL**

**From:** Dale P. Mentink
**Sent:** Wednesday, April 02, 2008 10:05 AM
**To:** 'Chris Beegan'
**Cc:** Dale P. Mentink
**Subject:** OAL file no. SWRCB_2008-0229-07S sediment quality objectives

Chris:

Thank you for the DVD. It worked fine. I'm just trying to tick off some items on my file review checklist and forgive me if I'm overlooking the obvious or looking for something(s) not there. I suspect you don't have another copy of this file there, but if you have the index as a document on your computer, I was wondering if you could help me identify a couple of things.

1. In terms of a Water Code 13147, 13244, and/or 40 CFR 25.5(b) Notice document, prior to the 11/19/07 or 2/5/08 public hearings or the 11/30/07 comment deadline, what in the record would you refer me to?
2. The plan document presented to the public for public comment prior to the 11/30/07 comment deadline is where in the record?
3. Is there a document or documents evidencing consultation with Public Resources Code 21080.3, 14 CCR 15386 agencies, or a document evidencing that none of those agencies have jurisdiction?
4. Was there ever a vote of the Board approving the concise summary text that is going into title 23 and is there a document showing that?

Thank you for whatever assistance you can be.

Dale Mentink
Office of Administrative Law
323-6817

**Mentink, Dale@OAL**

**From:** Dale P. Mentink
**Sent:** Friday, April 04, 2008 2:34 PM
**To:** 'Chris Beegan'
**Cc:** Dale P. Mentink
**Subject:** OAL file no. SWRCB_2008-0229-07 sediment quality objectives

Dear Chris:

I have finished my review of this file. In addition to the questions I raised in my email to you of April 2, I identified a few other things in the areas of the concise summary language for Title 23, the summary and response to comments, and the administrative record.

1. Concise Summary. Stressor Identification appears to be an important part of the plan. It's mentioned in the Summary of the plan at I.B. It's discussed for a number of pages in the Program of Implementation section. It answered some of the commenters regarding their objection to use of the empirical method rather than the mechanistic. It's discussed in several other contexts. Is there a reason why Stressor Identification was not mentioned specifically in the Clear and Concise Summary which is going into Title 23?

2. Administrative Record. Because I spent a fair amount of time with the file and Index and noticed some anomalies, I made a note of them. I provide these to you only if they are of some use to you.

a. 00128 is not a blank page, it's the last page of the Wilson memo although on back side.

b. 00193-00208 is not a second copy of settlement agreement, it looks like a briefing paper.

c. 00351-00352 are not indexed; it is a Notice of Public Hearing.

d. 00427-00429 are not indexed; they are Beegan to Bay email.

e. 00450 missing.

f. 00525 says it's a Cassette Tape (SQO Work Plan Adoption), but there's no tape.

g. 05739-06191 is dated 10/09/07 in the Index but 11/09/07 on the document.

h. 00671-00690 not indexed and not in box.

i. 01577-01578 not indexed but are in box (sign in sheet).

j. 03062-03122 not indexed but are in box.

k. 06779-06780 index has Richardson letter as only 06780 (actually a two-page letter).

3. Summary and Response to Comments.

I found substantial compliance with the summary of public comments consistent with 40 CFR Sec. 25.8. A comment that does not appear to have been acknowledged is on page 06643: "Minimum sample size for multi-station assessment of a single water body and minimum geographical coverage extent for individual sampling events must be specified." If you are amending the summary and response to comments (see below), could you summarize and respond to that one please?

In terms of responses, I identified 12 where the response was only that Staff Disagrees, but the comment wasn't lacking in specifics or merely rhetorical. Could you flesh out the responses to those 12 somewhat please? They are on the following pages: 08717 (30), 08718 (228), 08719 (231), 08719 (557), 08727 (23), 08737 (257), 08739 (262), 08747 (20), 08777 (298), 08800 (366), 08805 (381), and 08805 (384).

On page 08725 (241), the comment contains a specific question [Does this mean that significant differences for any two lines of evidence could drive an impairment designation?] and a specific recommendation [We recommend that benthic community data must be one of the two lines of evidence suggesting adverse effects before an impairment designation is assigned.]. The response is not really responsive. Could you flesh out that response?

On page 08803 (375), the comment is that monitoring may be as infrequent as once in five years and that such infrequent monitoring will allow degradation. The response is that the language describing maximum frequency has been deleted. There is nothing in the response explaining/justifying the minimum frequency monitoring which the commenter raised. Could you flesh out that response?

Please call or reply if you have any questions.

Dale Mentink, Office of Administrative Law, 323-6817.

**Mentink, Dale@OAL**

**From:** Dale P. Mentink
**Sent:** Thursday, April 10, 2008 4:58 PM
**To:** 'Chris Beegan'
**Cc:** Dale P. Mentink
**Subject:** OAL file no. SWRCB_2008-0229-07 sediment quality

Chris, I left a message for you this morning as to how things are going, but maybe you're out today.

A couple of other questions came up from our rulemaking file review committee. Two were just questions on the plan and one was a request for some additional information in response to one of the comments as long as I had asked about those other 14.

In terms of the plan, on page 08602 (page 8 of the plan), the committee wondered what the standard will be for approval of the "other methods" approved for use by the boards? In other words, if there are some standards already in mind, those should be stated in the plan, or the plan should simply explain why that approval process needs to be case by case.

On page 08614 (page 20 of the plan), what does the phrase "or other waters of significant national importance" refer to?

The other response to comment (and I'll find the page in the record in the morning) was just the one in which the use of the narrative as opposed to numeric object was challenged and being responded to and you mentioned that when numeric criteria are infeasible, the Water Code and Clean Water Act authorize the use of narratives. Could you provide the Water Code and Clean Water Act cites for that?

Thank you.

Please let me know how things are going. We have a due date of 4/14.
Dale Mentink
OAL, 323-6817

**Mentink, Dale@OAL**

**From:** Dale P. Mentink
**Sent:** Friday, April 11, 2008 8:32 AM
**To:** 'Chris Beegan'
**Cc:** Dale P. Mentink
**Subject:** OAL file no. SWRCB_2008-0229-07 sediment quality

I promised to get back to you this morning re which comment that was that mentioned the CWC and CWA; it's number 499 on page 08657 of the record. Thank you.

Dale, OAL
323-6817

**Mentink, Dale@OAL**

**From:** Chris Beegan [cbeegan@waterboards.ca.gov]
**Sent:** Monday, April 14, 2008 9:54 AM
**To:** Dale P. Mentink
**Cc:** Bruce Fujimoto; Dominic Gregorio; Sheila Vassey
**Subject:** OAL file no. SWRCB_2008-0229-07 sediment quality

Hi Dale: Thanks for the call this morning and your patience and diligence during the review of the record referenced above. As we discussed, I would like to withdraw the admininstrative record (OAL file no. SWRCB_2008-0229-07 sediment quality) with the intent of resubmitting a revised record ASAP after I have addressed those issues you identified in previous emails. Again thank you for your time and patience.

Chris Beegan
SWRCB
Division of Water Quality
Sediment Quality Objectives Development
1001 I Street, Sacramento Ca 95814
cbeegan@waterboards.ca.gov
Office (916) 341-5577
Cell (916) 955-9262

**Mentink, Dale@OAL**

**From:** Dale P. Mentink
**Sent:** Monday, April 14, 2008 12:42 PM
**To:** 'Chris Beegan'
**Subject:** RE: OAL file no. SWRCB_2008-0229-07 sediment quality

Thanks, I'll process that today. Just so you're not searching through emails and opening/printing any that are not relevant, the three emails containing questions/issues for follow-up or supplemental information or amendment are: 4/2/08 at 10:05 a.m.; 4/4/08 at 2:34 p.m.; and 4/10/08 at 4:58 p.m. Please let me know if you have any other questions.

Dale Mentink
Senior Staff Counsel
Office of Administrative Law
300 Capitol Mall, Suite 1250
Sacramento, CA 95814
(916) 323-6817
FAX (916) 323-6826
dmentink@oal.ca.gov

-----Original Message-----
From: Chris Beegan [mailto:cbeegan@waterboards.ca.gov]
Sent: Monday, April 14, 2008 9:54 AM
To: Dale P. Mentink
Cc: Bruce Fujimoto; Dominic Gregorio; Sheila Vassey
Subject: OAL file no. SWRCB_2008-0229-07 sediment quality

Hi Dale: Thanks for the call this morning and your patience and diligence during the review of the record referenced above. As we discussed, I would like to withdraw the admininstrative record (OAL file no. SWRCB_2008-0229-07 sediment quality) with the intent of resubmitting a revised record ASAP after I have addressed those issues you identified in previous emails. Again thank you for your time and patience.

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**Clear and Concise Summary**
**Water Quality Control Plan for Enclosed Bays and Estuaries of California,**
**Part 1- Sediment Quality,**
**Adopted by the State Water Resources Control Board on February 19, 2008**

**Title 23, CCR Section 3006**
On February 19, 2008, the State Water Resources Control Board (State Water Board) adopted Resolution No. 2008-0014 adopting Part 1 of the Water Quality Control Plan for Enclosed Bays and Estuaries of California (Part 1). Part 1 includes two narrative sediment quality objectives and a policy to implement the narrative objectives. The narrative objectives protect:

- benthic communities from direct exposure to toxic pollutants in sediment, and
- human health from pollutants in sediments that bioaccumulate in aquatic life to levels that are harmful.

Part 1 includes an implementation program that describes how to interpret and to apply the narratives in regulatory activities. The implementation program addresses:

- Determining exceedance of the sediment quality objectives
- Monitoring ambient sediment quality to determine whether beneficial uses are protected or are at risk
- Use of the sediment quality objectives as receiving water limits in Waste Discharge Requirements/National Pollutant Discharge Elimination System Permits
- Violations of receiving water limits
- The application of the sediment quality objectives for dredging activities, and
- The application of State Water Board Resolution No. 92-49 for cleanup actions and cleanup levels.

# State of California
# Office of Administrative Law

| | |
|---|---|
| **In re:**<br>**State Water Resources Control Board** | **NOTICE OF WITHDRAWAL** |
| **Regulatory Action:** | **Government Code Section 11349.3(c)** |
| **Title 23, California Code of Regulations**<br><br>**Adopt sections: 3006**<br>**Amend sections:**<br>**Repeal sections:** | **OAL File No. 2008-0229-07 S** |

This notice confirms that your proposed regulatory action regarding Water Quality Control Plan For Enclosed Bays and Estuaries, Part 1 - Sediment Quality was withdrawn from OAL review pursuant to Government Code section 11349.3(c). We will retain the rulemaking record you submitted in the event that you resubmit this regulatory action.

Please contact me at (916)323-6817 or dmentink@oal.ca.gov, or the OAL Reference Attorney at (916)323-6815, if you have any questions about the resubmittal process. You may request the return of your rulemaking record by contacting the OAL Front Desk at (916)323-6225.

Date: 4/14/2008

Dale P. Mentink
Senior Staff Counsel

For: SUSAN LAPSLEY
Director

Original: Dorothy Rice
Copy: Chris Beegan

## DECLARATION OF MAILING

**OAL File Number 2008-0229-07 S**

I transmitted Notice of Withdrawn to Dorothy Rice, Executive Director of the State Water Resources Control Board, at 1001 I Street , Sacramento, California 95814, on 4/14/2008 on behalf of the Office of Administrative Law, 300 Capitol Mall, Suite 1250, Sacramento, California 95814 by depositing the documents in the United States mail at Sacramento, California, enclosed in a sealed envelope(s) with postage fully prepaid.

I declare under penalty of perjury under the laws of the State of California, that the foregoing is true and correct and this declaration was executed on 4/14/2008 at Sacramento, California.

Lori Welton

(Signature of the Declarant)
Lori Welton