A Tiered Assessment Framework to Evaluate Human Health Risk of Contaminated Sediment

Article *in* Integrated Environmental Assessment and Management · December 2014

DOI: 10.1002/ieam.1610

3 authors:

Ben K Greenfield
Southern Illinois University Edwardsville
59 PUBLICATIONS **737** CITATIONS
SEE PROFILE

Aroon R Melwani
Applied Marine Sciences
25 PUBLICATIONS **116** CITATIONS
SEE PROFILE

Steven M Bay
Southern California Coastal Water Research …
112 PUBLICATIONS **1,752** CITATIONS
SEE PROFILE

Integrated Environmental Assessment and Management — Volume 11, Number 3—pp. 459–473
© 2015 SETAC

A Tiered Assessment Framework to Evaluate Human Health Risk of Contaminated Sediment

Ben K Greenfield,†‡ Aroon R Melwani,†§ and Steven M Bay||*

†San Francisco Estuary Institute, Richmond, California, USA
‡University of California, Berkeley, California, USA
§Macquarie University, Sydney, Australia
||Southern California Coastal Water Research Project, Costa Mesa, California, USA

(Submitted 19 March 2014; Returned for Revision 13 May 2014; Accepted 5 December 2014)

ABSTRACT

For sediment contaminated with bioaccumulative pollutants (e.g., PCBs and organochorine pesticides), human consumption of seafood that contain bioaccumulated sediment-derived contaminants is a well-established exposure pathway. Historically, regulation and management of this bioaccumulation pathway has focused on site-specific risk assessment. The state of California (United States) is supporting the development of a consistent and quantitative sediment assessment framework to aid in interpreting a narrative objective to protect human health. The conceptual basis of this framework focuses on 2 key questions: 1) do observed pollutant concentrations in seafood from a given site pose unacceptable health risks to human consumers? and 2) is sediment contamination at a site a significant contributor to seafood contamination? The first question is evaluated by interpreting seafood tissue concentrations at the site, based on health risk calculations. The second question is evaluated by interpreting site-specific sediment chemistry data using a food web bioaccumulation model. The assessment framework includes 3 tiers (screening assessment, site assessment, and refined site assessment), which enables the assessment to match variations in data availability, site complexity, and study objectives. The second and third tiers use a stochastic simulation approach, incorporating information on variability and uncertainty of key parameters, such as seafood contaminant concentration and consumption rate by humans. The framework incorporates site-specific values for sensitive parameters and statewide values for difficult to obtain or less sensitive parameters. The proposed approach advances risk assessment policy by incorporating local data into a consistent region-wide problem formulation, applying best available science in a streamlined fashion. *Integr Environ Assess Manag* 2015;11:459–473. ©2015 SETAC

Keywords: California Human health risk assessment Organochlorine Seafood Sediment quality objectives

INTRODUCTION

The health of fish, wildlife, and humans is often more adversely impacted by indirect exposure to contaminated sediment (bioaccumulation and food web trophic transfer), rather than by direct contact with sediment. It is well established that sediment-associated bioaccumulative compounds, such as DDT, polychlorinated biphenyls (PCBs), and methylmercury, biomagnify and can cause deleterious effects to wildlife and potential health risks to humans at environmentally relevant concentrations (Anderson et al. 1975; Hesslein et al. 1991; Kidd et al. 1995; Beyer et al. 1996; Huang et al. 2006; Schaeffer et al. 2006; Wiener and Suchanek 2008; Alava et al. 2012). Significant relationships between sediment contamination and fish contamination demonstrate instances where legacy polluted sediment is the source of risk to wildlife and humans (Wong et al. 2001; Zeng and Tran 2002; Melwani et al. 2009; Gehrke et al. 2011; Greenfield and Allen 2013), and conceptual and mechanistic models of contaminant bioaccumulation indicate sediment to be an important exposure pathway (Connolly 1991; Thomann et al. 1992; Arnot and Gobas 2004; Gobas and Arnot 2010; Parkerton and Connolly 2013).

Despite the importance of sediment as a contaminant reservoir, the relationship between sediment contamination and risk to humans and wildlife exhibits significant variability and uncertainty, affecting assessment and cleanup strategies and cost (Linkov et al. 2002; 2005; Gobas and Arnot 2010). In the US, federal Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) risk assessments managed by the Superfund program can cost tens of millions of dollars and take a decade or more to perform (Hamilton and Viscusi 1999; Gustavson et al. 2007). Although such complex and expensive assessments are warranted when costly cleanup efforts may result, there is also a need for more rapid screening-level assessments to evaluate overall ecosystem health and to prioritize among multiple sites. For sediment evaluation, tiered assessment frameworks make good use of limited resources by scaling the effort level to the magnitude and potential cost of the problem (USEPA 1991, 1998a; Contaminated Sites Management Working Group 1999; Bridges et al. 2005; Chapman and Anderson 2005; Hope 2009; Saloranta et al. 2011).

All Supplemental Data may be found in the online version of this article.

* Address correspondence to: greenfieldben1@gmail.com

DOI: 10.1002/ieam.1610

This article describes a conceptual framework and methodology for evaluating whether sediments meet a narrative objective for protection of human consumers of seafood. This tiered framework was developed under the oversight and guidance of committees comprising scientists and interested parties, in technical support of the California Sediment Quality Objectives Program (Beegan and Bay 2012). The framework is intended to address the need for a consistent and scalable approach to evaluate effects of contaminated sediments to seafood consumers. It selectively draws from methods established in existing tiered risk assessment practices (USEPA 1991, 1998a; Contaminated Sites Management Working Group 1999; Bridges et al. 2005; Chapman and Anderson 2005; Health Canada 2010a, 2010b), incorporating those elements relevant to human health risk evaluation of in-place sediment as part of meeting narrative sediment quality objectives. The article contains 5 sections: 1) current status of government sediment quality assessment for human seafood consumers, 2) scope of the framework, including the management intent within California, 3) conceptual overview of the framework, including the main features (tiered assessment, treatment of variability and uncertainty, and use of ecological feeding guilds to quantify seafood exposure), 4) detailed description of the framework's 3 tiers, and 5) discussion of advances and limitations.

INDIRECT EFFECTS SEDIMENT ASSESSMENT IN GOVERNMENT PROGRAMS

In principle, sediment quality assessment should integrate both direct effects to benthic organisms and indirect effects to humans and wildlife via the bioaccumulation pathway (Chapman and Anderson 2005). Perhaps a result of the perceived complexity of evaluation, US state and national sediment quality assessment programs often do not include standardized approaches for evaluating contaminant bioaccumulation and the resulting risk to wildlife and humans. Sediment quality guidelines in place in Canada and multiple US states (including New Jersey, Texas, Georgia, Minnesota, South Carolina, and Florida) presently focus on direct effects to benthic macrofauna but do not consider bioavailability nor bioaccumulation to higher trophic level organisms (Canadian Council of Ministers of the Environment 1999; Wenning et al. 2005; Maruya et al. 2012). The US Environmental Protection Agency/US Army Corps of Engineers (USEPA/USACE) 4-tiered approach for dredged materials evaluation incorporates indirect effects to a limited extent, in that bioaccumulation testing is used in Tiers 3 and 4, with results compared to US Food and Drug Administration (USFDA) Action Levels for human consumers of seafood using best professional judgment (USEPA 1991, 1998a). This standardized tiered approach has since been supplanted by more sophisticated approaches for evaluation of dredged material disposal alternatives (von Stackelberg et al. 2003; von Stackelberg and Burmistrova 2003; Bridges et al. 2005; USEPA 2005).

The US states of Washington and California are, to our knowledge, the only states with statewide sediment quality standards and the quantitative tools to implement these standards. These tools focus on measures of chemistry, benthic invertebrate toxicity tests, and community composition (State of Washington 1995; Beegan and Bay 2012). In contrast to direct effects to benthic communities, assessments of indirect effects to human health and wildlife are commonly performed at the individual site or water body scale, applying general risk assessment principles (USEPA 1998b; Cura et al. 1999), while using a diversity of individual approaches. For example, although Washington's recently revised Sediment Management Standards (Washington Department of Ecology 2013) include "a mechanism for setting standards to protect human health [pg. 4]" of seafood consumers, the policy indicates that compliance monitoring methods "shall be developed on a site specific basis. [pg. 93]" However, technical tools are readily available for applying standardized and consistent problem formulation and exposure assessment methods to evaluate indirect effects at state or regional scales (Health Canada 2010a, 2010b).

Perhaps due to increasing awareness and acceptance of valid technical approaches, some countries have recently developed recommended approaches for indirect effects sediment assessment. In Norway, for example, a 3-tiered methodology has been developed to evaluate human and ecological risk of sediment contamination at regional scales, with increasing complexity and effort as needed for higher tiers (Saloranta et al. 2011). Australia's guidance for sediment quality assessment includes bioaccumulation testing, in addition to the traditional benthic quality triad of sediment chemistry, toxicity, and community bioassessment, implicitly acknowledging the importance of ecosystem effects resulting from food web trophic transfer (Batley and Simpson 2008). A Netherlands guidance document recommends a simple method to categorize whether sediments cause food standards for fish consumption to be exceeded. The method compares numeric standards to fish concentrations, both measured in the field and modeled from sediment (Hin et al. 2010). Finally, although not specifically focused on sediment, Canada has recently provided detailed guidance on site risk assessment for human health, incorporating options of both preliminary quantitative screening assessment (Health Canada 2010a) and detailed quantitative risk assessment (Health Canada 2010b).

The recent development of standard methodologies to categorize sediments based on indirect effects parallels prior approaches for standardized assessment of sediment direct effects to benthic communities, such as interpretation of the sediment quality triad (Long and Chapman 1985; Chapman et al. 1997; Bay and Weisberg 2010; Beegan and Bay 2012), and complements existing integrated and ecosystem-based assessment frameworks (Chapman and Anderson 2005). The framework detailed below is intended to provide a standardized and consistent evaluation of the indirect effects of sediment for use in regulatory programs.

FRAMEWORK SCOPE

Environmental quality assessment should follow from explicitly stated environmental and management goals (Kwok et al. 2014). To that end, the framework we describe is intended to determine whether estuarine and marine embayment sediments meet California's narrative Sediment Quality Objective (SQO) for human health. The narrative states: "Pollutants shall not be present in sediments at levels that will bioaccumulate in aquatic life to levels that are harmful to human health (SWRCB 2009)." The framework determines whether sediment contamination at a site poses an unacceptable human health risk due to human consumption of contaminated fish and shellfish (i.e., seafood), a major exposure mechanism for bioaccumulative pollutants in sediment (USEPA 2000a; Huang et al. 2006; Schaeffer et al. 2006). This is a separate evaluation from that used for the aquatic life SQO, which evaluates whether sediments are able to support healthy communities of benthic macrofauna (Bay and Weisberg 2010; Beegan and Bay 2012). Other human exposure pathways (e.g., dermal contact) are not considered here. The assessment is performed at a "site" scale, with a site defined simply as an area of interest within a water body. Depending on the specific programmatic assessment goal and study design, a site may be equivalent to an entire bay or estuary, or a portion thereof.

This assessment framework is intended to provide a consistent method for interpreting monitoring data and implementing the human health SQO. The results of the assessment are expected to be used for multiple purposes where the SQO applies. These include 1) identifying water bodies or segments on the 303(d) impaired list for the US Total Maximum Daily Load regulatory program (Birkeland 2001), 2) assessing compliance with permit conditions, and 3) provisionally prioritizing sites for management action (Figure 1). Implementation of the SQO throughout California requires a standardized assessment framework that uses comparable methods and thresholds among water bodies, so management resources can be focused on priority areas.

The framework focuses on associating risk with sediment contamination, rather than evaluating overall risk of seafood consumption. As such, it would not be appropriate for development of seafood consumption advisories, fishing restrictions, or other fishing-related risk management activities. In California, such activities are performed by the Office of Environmental Health Hazard Assessment and may include water body scale risk assessment based on local fishing and consumption practices, species-specific contamination, and target consumer population (USEPA 2000a, 2000b; Klasing and Brodberg 2008).

This assessment framework also does not provide all the information needed to plan a management response. If a need for management action has been determined, a more comprehensive and specific risk assessment may be needed to delineate the spatial extent of the impact, identify sources, and determine cleanup goals (Apitz et al. 2005; Bridges et al. 2005). Additional data and analysis would likely be needed to determine an appropriate management response (e.g., no action, monitored natural recovery, dredging, capping, or confined disposal) (Contaminated Sites Management Working Group 1999; USEPA 2005).

The technical tools developed for the framework are applicable to legacy organochlorine compounds: PCBs, DDTs, chlordanes, and dieldrin. These compounds were chosen due to well established and validated empirical and mechanistic approaches for characterizing bioaccumulation and human exposure from sediment sources (Thomann et al. 1992; Arnot and Gobas 2004; Gobas and Arnot 2010), and management concern for human exposure to these pollutants in California (SWRCB 2006; Connor et al. 2007; Davis et al. 2007, 2011). The overall assessment approach (e.g., indicators, data integration strategy, site classification criteria) should be applicable to other contaminants (e.g., Hg, contaminants of emerging concern), after further development of the necessary tools and parameters.

CONCEPTUAL APPROACH: 2 QUESTIONS AND 3 TIERS

Rather than simply providing a hazard or risk characterization, the assessment evaluates whether a state objective is met, with potential implications for site classification and management. Therefore, the conceptual approach must be

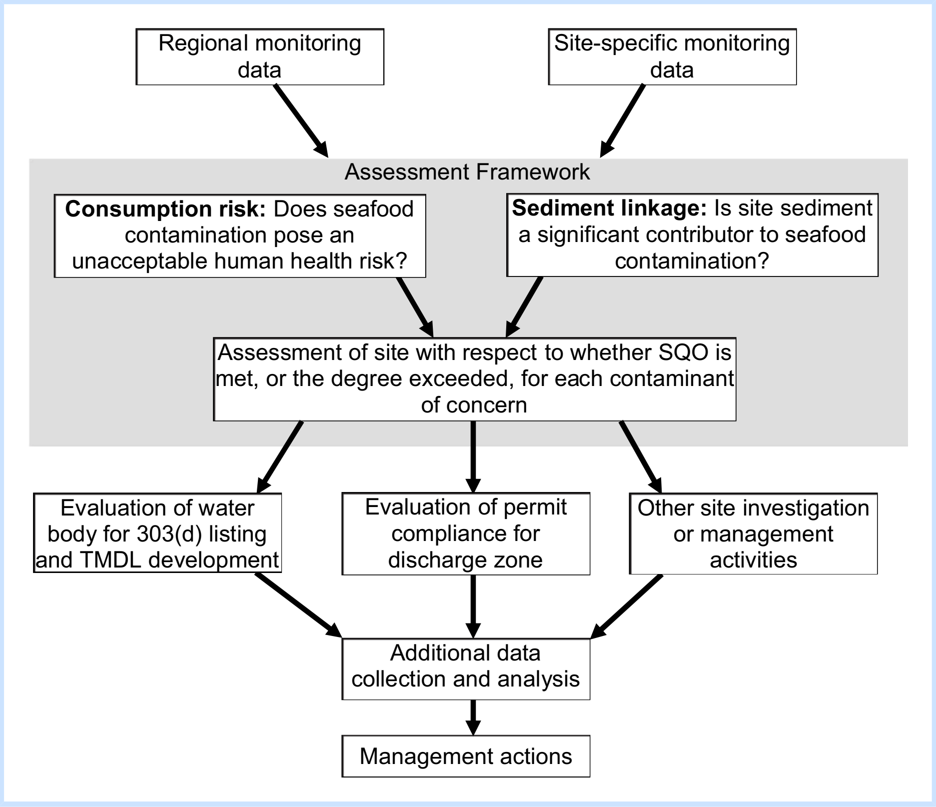

Figure 1. Role of the human health SQO data assessment framework (shaded region) in site evaluation and management actions.

carefully and precisely defined. Two specific questions are addressed in the framework (Figure 1): 1) to what extent do pollutant concentrations in seafood pose unacceptable health risks to human consumers (hereafter referred to as "consumption risk")?; and 2) to what degree is sediment contamination at a site significantly linked to seafood contamination (hereafter, "sediment linkage")? Consumption risk is calculated as human health risk from consuming seafood associated with the site. Sediment linkage describes the strength of the association between sediment contamination within the site and health risk. The degree of sediment linkage indicates the bioaccumulation due to sediment contamination from the site, estimated using bioaccumulation models. The presence of a strong linkage with site sediment is a critical element in determining whether the SQO is attained for the site because it indicates whether health risks are likely due to site conditions (relevant to the SQO) or due to off-site factors (such as fish movement) that are the focus of other regulatory and/or monitoring programs.

Integration of the results (Figure 2) produces a categorical site assessment, which summarizes the likelihood and magnitude of health risk associated with sediment contamination within the site. Finally, the site assessment results are compared to criteria to be adopted by the California State Water Resources Control Board (hereafter, State Water Board) to determine whether the human health SQO is met at the site. As is typical in environmental assessment (Kwok et al. 2014), the entire approach incorporates a combination of technical information (e.g., estimated exposure concentrations based on field measurements) and policy judgments (e.g., what risk levels to what portion of the human population are deemed acceptable for meeting the SQO). The technical elements of the assessment framework are the focus of this article, as the policy questions are still under consideration by the State Water Board. We have included policy decisions in this article as examples to demonstrate the complete assessment process; these examples may differ from those adopted by the state in the future.

Application of the assessment framework is organized into 3 tiers (Figure 3). Following a general tiered assessment methodology, each tier represents an increasing level of effort and complexity in order to enable the assessment to match variations in data availability, site complexity, and study objectives (USEPA 1991, 1998a; Contaminated Sites Management Working Group 1999; Bridges et al. 2005; Chapman and Anderson 2005; Hope 2009).

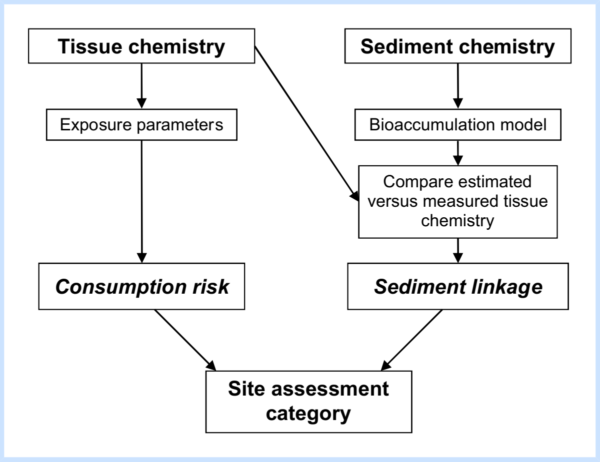

Figure 2. Schematic of conceptual approach to Tier 2 site assessment for the human health SQO.

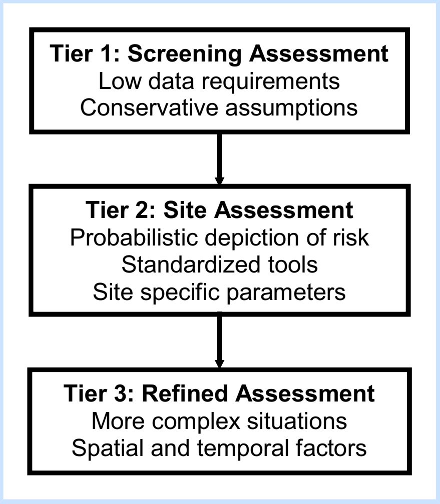

Figure 3. Tiered assessment framework.

Tier 1 consists of a screening assessment of either tissue data or sediment data to determine whether there is sufficient potential concern for human health impacts to warrant a complete site assessment. The purpose of Tier 1 is to provide an option for initial site evaluation with relatively low data requirements, thereby enabling rapid identification of areas of low concern (USEPA 1991, 1998a; Hope 2009). Sediment or tissue chemical concentration data are interpreted using standardized conservative assumptions to evaluate concern for human consumers of seafood. If Tier 1 indicates potential concern, the analysis proceeds to Tier 2; otherwise, the site is determined to meet the SQO without a requirement for further assessment.

In Tier 2, both tissue and sediment data are evaluated with additional site-specific information to determine human health risk. Tier 2 differs from Tier 1 in 3 important respects. First, in Tier 2, some default assumptions and parameters are replaced with site-specific assumptions and parameters, such as seafood forage area, and habitat characteristics. Second, in Tier 2, the resulting estimates of consumption risk (from tissue data) and sediment linkage (from sediment data) are compared to classify the site condition. Finally, the Tier 2 analysis produces probabilistic output, including cumulative distribution functions of 2 indicators: consumption risk and sediment linkage (further described below). If Tier 2 results indicate an acceptable condition, the sediment is classified as meeting the human health SQO. If Tier 2 results indicate an unacceptable risk to human health due to site sediments, there are 2 alternative outcomes: 1) determine that the SQO is not met, or 2) determine that site-specific conditions warrant a more refined Tier 3 analysis.

The Tier 3 assessment may be used when the Tier 2 results are deemed unreliable due to site-specific conditions such as other sources of contamination, temporal variability, or substantial uncertainty in exposure parameters (e.g., seafood exposure to site sediments, human consumption rate, or bioavailability of contaminants in site sediments). The specifics of the Tier 3 assessment method are determined on a site-specific basis and might include the collection of additional data and use of alternative data analysis methods. However, the data would be assessed following the concepts described for Tier 2 (Figure 2).

Treatment of variability and uncertainty

The approach to variability (true heterogeneity) and uncertainty (lack of knowledge) follows standard tiered assessment practice in using conservative assumptions in earlier tiers (Tier 1) and probabilistic descriptions of outcome range in later tiers (Tiers 2 and 3) (Thompson and Graham 1996; Hope 2009; Health Canada 2010a, 2010b). A deterministic (i.e., based on calculations that do not include a random element) approach is used for Tier 1 to provide simplicity and ease of application.

The Tier 1 approach addresses variability and uncertainty by using conservative point estimates of key input parameter values. The Tier 2 and Tier 3 assessments use a stochastic (Monte Carlo) simulation model to incorporate aspects of variability and uncertainty into the assessment, generating a probability distribution of outcome results for both consumption risk and sediment linkage at a site (McKone and Bogen 1991). This follows the accepted practice of using probabilistic approaches to evaluate human health risk associated with contaminated sediment and seafood (Thompson and Graham 1996; Wilson et al. 2001; Linkov et al. 2002; Gobas and Arnot 2010). The probabilistic approach enables an estimate of how the risk from consuming contaminated seafood varies for different portions of the human population.

The Tier 2 stochastic analysis is standardized and limited in scope to focus on the variability and uncertainty of a subset of general parameters that will be locally available and are expected to have the greatest influence the assessment outcome. If a Tier 3 analysis is used, the specific modifications to the approach will be determined by the management and information needs for the site in question.

Use of bioaccumulation model and ecological feeding guilds

An ecological feeding guild approach is used to evaluate health risk and model exposure from sediments. For this approach, finfish species in California estuaries and marine embayments are categorized into 1 of 8 feeding guilds, based on their diets (detailed in Supplemental Data and Tables S1, S2, and S3). This is based on the general finding that trophic position and consumption of benthic prey influences contaminant exposure (Connolly 1991; Thomann et al. 1992; Kidd et al. 1995). In particular, species that consume similar prey types will have similar food web exposure to sediment-associated contaminants.

The feeding guild approach is used in 2 ways. First, information regarding guild membership aids in selecting local seafood species to assess risk to seafood consumers. The Supplemental Data contains a tabulation of guild membership of appropriate species, which can be used to inform target species selection (Table S1). Selection and analysis of species across multiple guilds provides additional information regarding the variability in seafood exposure associated with different diets. Second, guild-based diet attributes are used to calculate the linkage of site sediments to local seafood exposure, based on the predicted biota-sediment accumulation factor (BSAF) for the target fish species. In this approach and throughout the article, BSAF represents (wet weight concentration in tissue)/(dry weight concentration in sediment).

The model used to calculate the BSAF is the Arnot and Gobas (2004) food web model, modified by Gobas and Arnot (2010). This model was selected because it is a mechanistic bioaccumulation model that has publicly available and open source spreadsheets and documentation, limited complexity to increase ease of application, validated depiction of the primary bioaccumulation processes (Burkhard 1998; Arnot and Gobas 2004; Gustavson et al. 2011), as well as multiple examples of successful prior use (Linkov et al. 2002; von Stackelberg et al. 2002, 2003; Anchor Environmental 2005; Gewurtz et al. 2006; Saloranta et al. 2006; Gobas and Arnot 2010; Gustavson et al. 2011; Alava et al. 2012). BSAFs obtained from this model have been previously corroborated with empirical field data in multiple water bodies, including finfish from multiple feeding guilds within marine embayments. These data are appropriate for conservative estimates of site sediment linkage to seafood tissue contamination (Arnot and Gobas 2004; Condon and Gobas 2007; Gobas and Arnot 2010).

For the indirect effects SQO framework, model parameter development was augmented by development of dietary estimates for indicator species representative of each feeding guild (Tables S2 and S3). These dietary estimates are incorporated into the calculation of sediment linkage. Because of the strong influence of organic carbon on contaminant partitioning, variability in BSAF was more sensitive to site sediment total organic carbon than feeding guild. BSAF increased approximately 10-fold as sediment TOC decreased from 4% to 0.1% but varied approximately 2-fold among feeding guilds (Tables S4, S5, S6, and S7).

Developing a conceptual site model

Conceptual site model development is key to planning the assessment and formulating management decisions (Contaminated Sites Management Working Group 1999; Cura et al. 1999; Bridges et al. 2005). Using local information and expertise, the conceptual site model should include information needed to determine 1) site boundaries and site size, 2) seafood consumer population characteristics (e.g., consumption rate) (USEPA 2000a), 3) seafood species to be monitored (USEPA 2000b), 4) site-specific modification to other parameters, as needed, such as seafood movement range or diet (Connolly 1991; Linkov et al. 2002; Gehrke et al. 2011; Chen et al. 2014), 5) benthic community structure and diet (Saloranta et al. 2006; Gobas and Arnot 2010; Alava et al. 2012), 6) sources and sinks of allochthonous and autochthonous organic C (Jassby et al. 1993), and 7) the biologically active zone in the sediment bed (Bridges et al. 2005).

DESCRIPTION OF TIERS

Tier 1

Tier 1 is an optional tier that uses standardized conservative assumptions, a simplified analysis approach, and existing available data. Tier 1 evaluates whether the site poses sufficient concern for seafood consumers to warrant further investigation. A Tier 1 assessment may be performed using either seafood tissue or sediment data (or both), depending on data availability. Sediment and tissue data must meet 5 minimum data applicability requirements: 1) obtained within site boundaries, 2) obtained within the 6 years before the assessment, 3) sediment has matching total organic carbon content data, 4) contaminant measurements must include a required list of constituent compounds (i.e., specific PCB congeners, DDT isomers and metabolites, and chlordane compounds), and 5) tissue contaminant data are from the list of acceptable seafood species (Table S1).

The Tier 1 tissue-based consumption risk evaluation is performed by comparing measured tissue contaminant concentration to screening thresholds. The measured tissue concentration is conservatively estimated based on 95% upper confidence limits of the mean ($\mu_{95\%\mathrm{UCL}}$) tissue concentration for each individual species

$$C_{B,T1} = [\Sigma(\mu_{95\%\mathrm{UCL}})]/n, \tag{1}$$

where $C_{B,T1}$ is the Tier 1 seafood (biota) tissue concentration (ng/g wet weight), Σ is the sum across all species, and n is the number of species included. If the sample size is less than 3 samples for a given seafood species, the maximum concentration is used in lieu of $\mu_{95\%\mathrm{UCL}}$ for that estimate. Consumption risk is determined by comparing $C_{B,T1}$ to the biota tissue screening thresholds (T_B, ng/g wet weight) in Table 1.

Sediment-based evaluation is similarly determined by comparing measured contaminant concentrations in sediment (C_S, ng/g dry weight) to sediment screening thresholds (T_S). The Tier 1 sediment contaminant concentration ($C_{S,T1}$) is simply $\mu_{95\%\mathrm{UCL}}$ of all available sediment data (or the maximum concentration if sample size is <3). The T_S is calculated as the tissue threshold (T_B, Table 1) divided by the BSAF [(ng/g wet biota tissue)/(ng/g dry sediment)]

$$T_S = T_B/\mathrm{BSAF}, \tag{2}$$

where BSAF is obtained as a function of contaminant class, seafood guild, and sediment TOC, using contaminant-specific lookup tables (Tables S4, S5, S6, and S7). The BSAF lookup tables were developed using a bioaccumulation model (Arnot and Gobas 2004; Gobas and Arnot 2010), run with several standard food webs based on guild-specific dietary composition (Tables S2 and S3). From the feeding guilds evaluated at the site, the guild with the highest BSAF is used to set the BSAF value in Equation 2.

If both sediment and tissue are available, and either measurement ($C_{B,T1}$ or $C_{S,T1}$) exceeds the appropriate screening threshold (T_B or T_S, respectively), a Tier 2 assessment is required. If the $C_{B,T1}$ and $C_{S,T1}$ values are below the screening thresholds, the assessment is complete, with the site sediment determined to be unimpacted, based on low consumption risk for seafood consumers.

Tier 2

Both seafood and sediment contamination data are used for the Tier 2 assessment. The separate evaluation of consumption risk (based on seafood data) and sediment linkage (based on sediment and seafood data) is a key distinguishing feature of the Tier 2 approach (Figure 2). This may require new collection of sediment or tissue data to obtain a complete data set.

Consumption risk is evaluated by inputting observed seafood contamination data into a probabilistic exposure model for human consumers (Figure 4). Percentiles along the resulting cumulative distribution functions are compared to risk thresholds, to categorize the consumption risk indicator. Thus seafood tissue data, rather than sediment data, forms the basis for assessment of risk to human consumers.

Sediment linkage is evaluated by entering the measured sediment contamination data into a bioaccumulation model

Table 1. Tier 1 biota tissue screening thresholds (ng/g wet tissue)

Threshold [a]	DDT	PCB	Chlordane	Dieldrin
Cancer risk [b]	214	36	56	4.6
Noncancer hazard	1563	63	103	156

[a] Risk parameters used for the calculation of these thresholds include consumption rate = 32 g/d, cooking reduction = 0.7, exposure duration = 30 y, averaging time = 70 y (SFEI 2000; USEPA 2004; Klasing and Brodberg 2008).
[b] Based on allowable cancer risk of 10^{-5}.

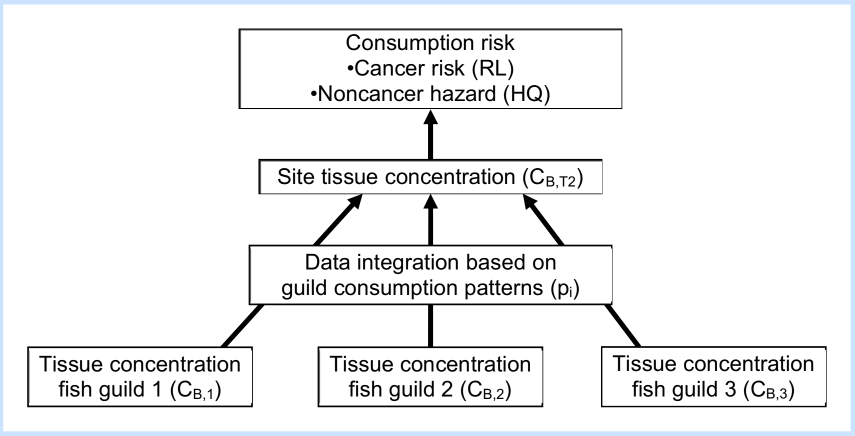

Figure 4. Strategy for determining the consumption risk to seafood consumers. The number of guilds included in the analysis depends on the conceptual site model.

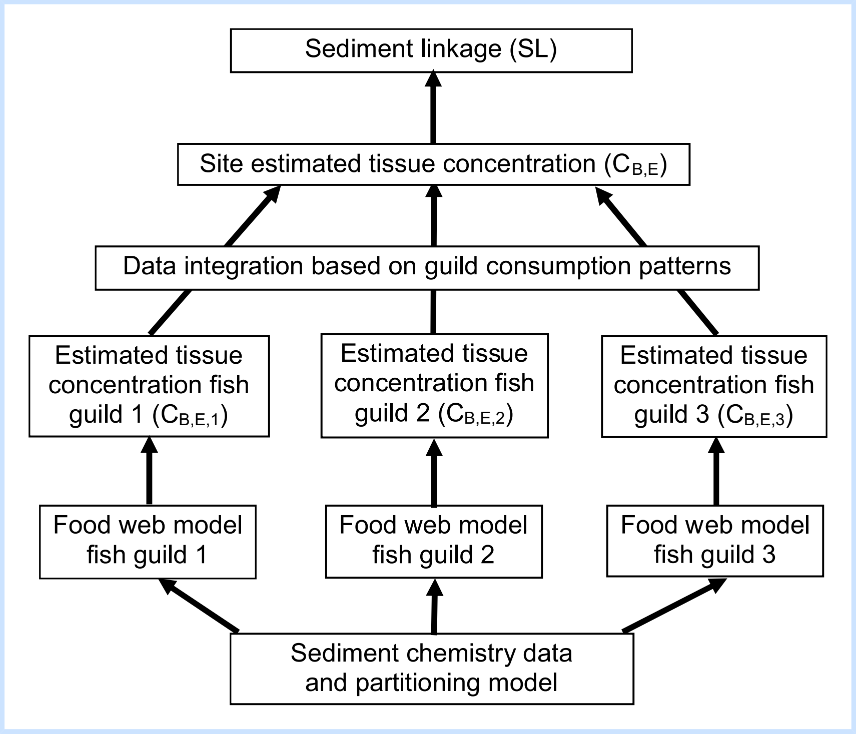

Figure 5. Strategy for determining the sediment linkage to fish bioaccumulation. The number of guilds included in the analysis depends on the conceptual site model.

that calculates the estimated seafood contamination due to contaminated sediment at the site. This distribution of expected tissue contaminant concentrations is then compared to the average measured seafood tissue contaminant concentration at the site, and a distribution of this ratio is calculated (Figure 5). The quotient of the estimated tissue contaminant concentration divided by the measured seafood tissue contamination is defined as the sediment linkage indicator. Thus, in the case where a very low proportion of the seafood tissue contamination is explained by site sediment contamination, there is a weak association to the sediments (i.e., a low sediment linkage), and the tissue exposure is likely due to other sources (e.g., off-site sediments or direct discharges to the water column).

All Tier 2 calculations for seafood consumption risk and sediment linkage are performed in a Decision Support Tool spreadsheet model, developed in Microsoft Excel. The user parameterizes the Decision Support Tool by defining the monitored species, which fall into 1 of 8 dietary guilds, and quantifying their relative importance in the human diet. Other site-specific attributes, such as sediment and tissue contaminant concentrations, are also entered. The Monte Carlo simulations generate probabilistic distributions of 3 results: cancer risk and noncancer hazard due to the overall seafood diet of the human consumers (Figure 6), and sediment linkage to the seafood contamination (Figure 7).

Consumption risk calculation. Consumption risk is based on calculations of cancer risk and noncancer hazard, that are in turn based on measured tissue contaminant concentrations from multiple species monitored at the site (Figure 4). Appropriate species are selected based on development of a conceptual site model, including consideration of the different dietary guilds present at the site. For statewide comparability, the risk calculations are the same as those used by California's Office of Environmental Health Hazard Assessment (OEHHA) to develop seafood consumption guidelines and advisories (Klasing and Brodberg 2008), which are also consistent with USEPA guidance for evaluating seafood consumption risk (USEPA 2000a, 2000b).

Cancer risk

$$\mathrm{RL} = \underline{\boldsymbol{C_{B,T2}}} \times \underline{\mathbf{CR}} \times \mathrm{CSF} \times (\mathrm{ED}/\mathrm{AT}) \times \mathrm{CRF}/\mathrm{BW}, \qquad (3)$$

Noncancer hazard

$$\mathrm{HQ} = \underline{\boldsymbol{C_{B,T2}}} \times \underline{\mathbf{CR}} \times \mathrm{CRF}/(\mathrm{RfD} \times \mathrm{BW}), \qquad (4)$$

where $C_{B,T2}$ = Tier 2 tissue contaminant concentration for seafood species measured within the site (ng/g), AT, averaging time (y); BW, human body weight (kg); CR, consumption rate (kg/d); CRF, cooking reduction factor (unitless); CSF, cancer slope factor $(\mathrm{ng/g/d})^{-1}$; ED, exposure duration (y); HQ, hazard quotient for noncancer effects (unitless); RfD, reference dose (ng/g/d); and RL, cancer risk level (unitless). In Equations 3 and 4, the underlined and boldface parameters (tissue concentration and consumption rate) are varied stochastically in the Tier 2 Monte Carlo simulation.

Tissue concentration ($C_{B,T2}$) is the weighted average contaminant concentration for a given compound class (e.g., sum of PCBs, sum of DDTs, sum of chlordanes, or dieldrin). The tissue concentration is based on measurements of appropriate seafood species collected within the site. Exposure concentrations are average concentrations weighted by human diet

$$C_{B,T2} = \sum_{i=1}^{n} C_{B,i} \times p_i, \qquad (5)$$

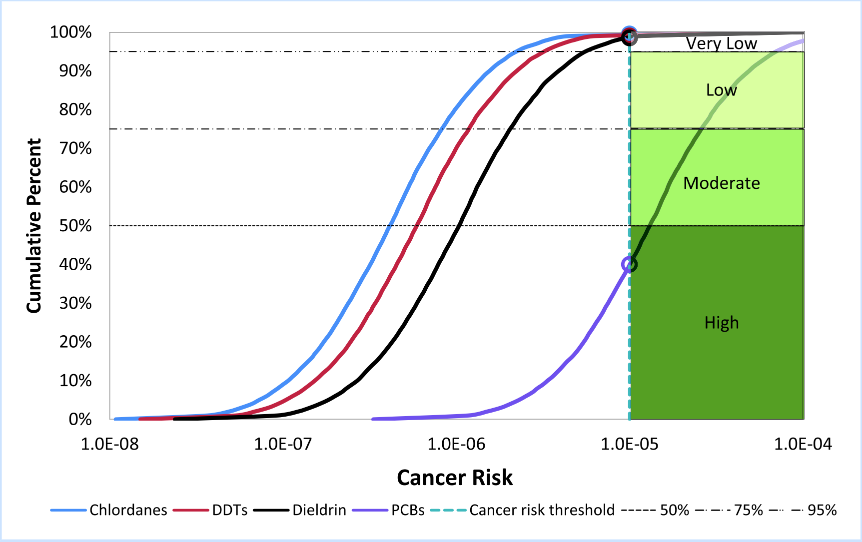

Figure 6. An example of the consumption risk output for cancer risk. For each contaminant the cumulative frequency distribution of cancer risk is compared to the threshold (here set at 10^{-5}), with the portion of the population below the threshold indicating the categorical outcome.

where i (1, 2, … n) are the individual seafood species monitored, $C_{B,i}$ is the average tissue concentration for species i, and p_i is the proportion of the human consumer seafood diet represented by species i. The relative proportion of each guild in the diet, determined as part of a conceptual site model, is based on local catch or consumption data for each fish species. Although not a component of the Tier 2 assessment, the end user can also examine and compare tissue concentrations across species and guilds. This can help to identify the factors that most influence seafood exposure (e.g., diet vs tissue lipid content), refine the conceptual site model, and identify priority species for future monitoring.

Consumption rate (CR) and tissue concentration ($C_{B,T2}$) were chosen for probabilistic sampling based on higher contribution to variability in consumption risk outcome than other empirically available parameters (CRF, ED, and BW), and greater local data availability than cancer risk and noncancer hazard parameters (CSF, RfD) (Table 2). Explicitly considering variability versus uncertainty in contribution to outcome (Bogen and Spear 1987; von Stackelberg et al. 2002), it is further assumed that CR and $C_{B,T2}$ dispersion are dominated by variability rather than uncertainty. Consumption rate exhibits wide variability across the seafood consumer population and thus the population central tendency and dispersion are described (e.g., average and standard deviation). Seafood contamination exhibits measurement uncertainty and sample size limitations but is time-integrated by human consumers; as such, sample central tendency and dispersion (e.g., average and standard error) are used (Table 2). Tier 3 (below) provides an option to introduce dispersion in other parameters, if this is determined to be a high priority for assessment.

Sediment linkage calculation. Sediment linkage is calculated by comparing estimated versus observed biota tissue contaminant concentration ($C_{B,E}$ vs $C_{B,T2}$). Estimated tissue contaminant concentration is calculated for each species monitored ($C_{B,E,i}$, ng/g), where i (1, 2, … n) refers to the individual species

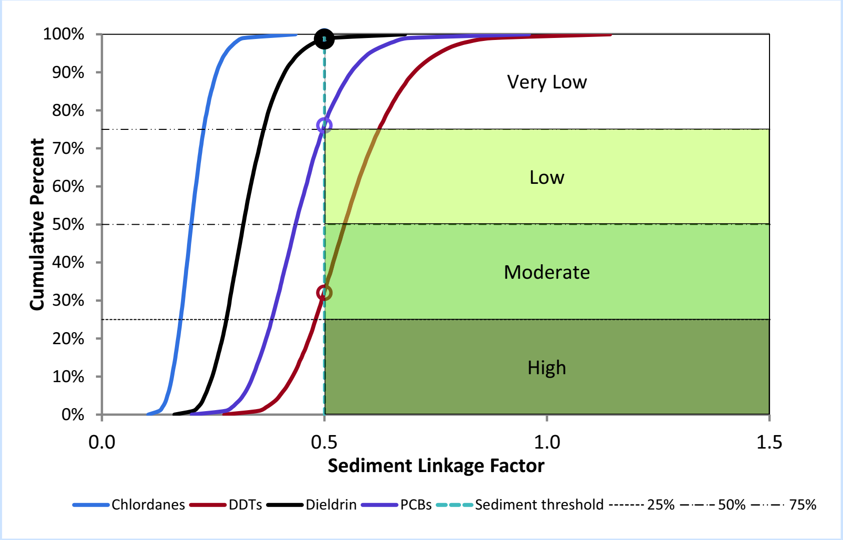

Figure 7. An example of the sediment linkage output. For each contaminant the cumulative frequency distribution of the sediment linkage is compared to the threshold (here set at 0.5), with the portion of the population below the threshold indicating the categorical outcome.

Table 2. Treatment of stochastic parameters in the Tier 2 assessment

Parameter	Data source	Focus of dispersion measurement [a]	Notes
CR	Statewide estimate or local site data (when available)	Entire statistical population (i.e., SD)	Need to assess potential impact to full population that consumes local seafood
$C_{B,T2}$	Local data from site	Average (i.e., SE)	Site-specific data needed on contaminant exposure; exposure of human consumers will tend to average across sediment and tissue population over time
$C_{S,T2}$	Local data from site	Average (i.e., SE)	Site-specific data needed on contaminant exposure; exposure of human consumers will tend to average across sediment and tissue population over time
BSAF	Calculate using bioaccumulation model with combination of local and general parameters	Entire statistical population (i.e., SD)	Bioaccumulation model incorporates key site-specific parameters; variability and uncertainty in bioaccumulation within and across populations is characterized using a standard parameter (Burkhard et al. 2010; Gobas and Arnot 2010), obtained in a statewide analysis
HR	Statewide estimate	Entire statistical population (i.e., SD)	Local data will generally be unavailable; movement range relatively uncertain

BSAF = biota-sediment accumulation factor; $C_{B,T2}$ = tissue contaminant concentration; CR = seafood consumption rate; $C_{S,T2}$ = sediment contaminant concentration; HR = seafood home range; SD = standard deviation; SE = standard error.

For some parameters, statewide estimates are provided, based on analysis of California data or best available information. For other parameters, local estimates are obtained for the site being assessed.

[a] Estimates of variability across the statistical population; examples include SD, standard error of the mean, or coefficient of variation.

monitored (Figure 5). $C_{B,E,i}$ is calculated based on measured sediment chemistry concentrations from the site, combined with a bioaccumulation model

$$C_{B,E,i} = \underline{C_{S,T2}} \times \underline{\mathrm{BSAF_i}} \times \mathrm{SA}/\underline{\mathrm{HR_i}}, \tag{6}$$

where $C_{S,T2}$ is the Tier 2 average measured contaminant concentration in sediment from the site (ng/g); $\mathrm{BSAF_i}$, biota-sediment accumulation factor for species i [(ng/g wet biota tissue)/(ng/g dry sediment)]; SA, site area ($\mathrm{km^2}$) or length across the site (km); and $\mathrm{HR_i}$, seafood home range ($\mathrm{km^2}$) or linear movement distance (km) for species i.

Three parameters in Equation 6 are varied probabilistically: sediment sum contaminant concentration ($C_{S,T2}$), home range ($\mathrm{HR_i}$), and biota-sediment accumulation factor ($\mathrm{BSAF_i}$), again based on a combination of sensitivity of the outcome to their variation and ability to obtain local or statewide estimates of variability or uncertainty (Table 2). Equation 6 uses a site-use factor (i.e., $\mathrm{SA/HR_i}$) to account for the movement of fish beyond the site boundaries (Hope 1995; Wickwire et al. 2011). The site-use factor is set to equal to 1 when the home range is smaller than the site (i.e., HR < SA).

Similar to measured tissue concentration (Eqn. 5), average tissue concentration estimated from sediment ($C_{B,E}$, ng/g) is calculated based on the combined results for all seafood species included in the assessment

$$C_{B,E} = \sum_{i=1}^{n} C_{B,E,i} \times \mathrm{p_i}, \tag{7}$$

where i (1, 2, … n) are the individual species modeled from sediment, $C_{B,E,i}$ is the modeled tissue concentration for seafood species i (Eqn. 6), and $\mathrm{p_i}$ is the proportion of the human seafood consumer diet represented by species i. In summary, Equation 6 estimates the biota tissue concentration that would result from exposure to site sediment for each seafood species monitored at the site. These results are combined in Equation 7 to determine the overall estimated concentration (Figure 5). Finally, sediment linkage (SL, unitless) is calculated as

$$\mathrm{SL} = C_{B,E}/C_{B,T2}, \tag{8}$$

the ratio of estimated tissue contaminant concentration ($C_{B,E}$) versus observed concentration resulting from exposure to site sediment contaminants ($C_{B,T2}$).

Tier 2 site assessment steps, categorization thresholds, and integration of results. Tier 2 is organized into 7 site assessment steps

Step 1: Develop conceptual site model

Step 2: Input data for site-specific parameters

Step 3: Run the bioaccumulation model to calculate BSAFs for use in sediment linkage calculations

Step 4: Perform Monte Carlo simulations to generate cumulative probability distributions of consumption risk and sediment linkage results

Step 5: Plot and evaluate results of the simulations

Step 6: Categorize results for the consumption risk and sediment linkage indicators

Step 7: Make a site assessment based on the indicator categories

Tier 2 indicator categorization is achieved by comparing the probabilistic output from the Decision Support Tool to a consumption risk threshold and a sediment linkage threshold. This process is performed for each contaminant group separately, with the final site assessment based on the highest level of risk from site contamination obtained for any compound.

We illustrate Tier 2 indicator categorization with a case study example. The State Water Board will consider and ultimately decide the threshold values for SQO assessment, following a formal public process. For this example, we have used thresholds of 10^{-5} for acceptable cancer risk and 1.0 for noncancer hazard index. In the assessment, the cumulative frequency distribution is plotted versus the threshold, and the point of overlap (indicating the percent of the statistical population that is below the acceptable cancer risk or noncancer hazard) determines the outcome category (Figure 6).

Both the acceptable risk level and the proportion of the human population at that risk level are integrated to make the assessment. The threshold is evaluated at 3 points on the consumption risk cumulative frequency distribution: 95%, 75%, and 50%, resulting in 1 of 4 categorical outcomes (Table S8). The outcome category is determined based on the percent of seafood consumers that is below the threshold. In the illustrated example (Figure 6), less than 50% of the population has cancer risk from PCBs below the 10^{-5} risk threshold, resulting in a category of high consumption risk due to cancer effects of PCBs. In contrast, at least 95% of the population has cancer risk from DDTs, chlordanes, and dieldrin below the threshold, indicating very low consumption risk.

For sediment linkage, a distribution of results (Eqn. 8) is calculated to account for variability and uncertainty in model parameters (Eqn. 6). Similar to the consumption risk evaluation, the sediment linkage evaluation is performed by overlaying this cumulative distribution of estimated versus observed tissue quotients versus a single threshold (for this example, set at 0.5). The point of overlap determines the categorical assessment (Figure 7), and is based on the proportion of modeled bioaccumulation results that would account for at least half of observed seafood contamination. The evaluation is performed at 3 points on the sediment linkage cumulative frequency distribution: 75%, 50%, and 25% (Table S9). In the illustrative example for PCBs, chlordanes, and dieldrin, less than 25% of the distribution is above the 0.5 threshold for sediment linkage. This low percentage indicates a very low linkage of sediment to tissue contamination for these contaminants. As a result, the interpretation in this example is that the PCB, chlordane, and dieldrin tissue contamination that exists is not largely attributable to site sediments. In contrast, for DDTs, 67% of the probabilistic simulation results are above the 0.5 sediment linkage threshold (i.e., 33% of results are below the threshold; Figure 7). This result falls into the category indicating moderate linkage to tissue contamination, and suggesting higher linkage for DDTs than the other examined contaminants (Figure 7).

Table 3 describes the proposed approach for integrating the consumption risk and sediment linkage categories into an assessment decision for each contaminant. The third column of Table 3 describes in narrative form the interpretation of each possible combination of the 2 indicators. The integrated results are summarized as 1 of 5 categories, each describing the relative impact of site sediment contamination on human health. These categories range from Unimpacted (best sediment quality) to Clearly Impacted (greatest deviation from protected condition described in the SQO) and are structured similarly to the categories used to assess California's SQO for direct effects on benthic communities (Bay and Weisberg 2010).

- Unimpacted: Site sediments have minimal impact, due to very low consumption risk overall
- Likely unimpacted: Elevated health risk from site sediment contamination is present for a small proportion of consumers, or sediments are not responsible for the elevated risk
- Possibly impacted: Unacceptable health risk for many consumers, but site sediment contamination has a minor influence
- Likely impacted: Unacceptable health risk is likely and strongly linked to site sediment contamination
- Clearly impacted: Site sediment contamination is the dominant factor responsible for unacceptable health risk to many consumers

Tier 3

Tier 3 is an optional assessment to address unique situations or evaluate additional factors affecting the assessment of risk not considered in Tier 2. Tier 3 uses the same framework, indicators and decision criteria described in Tier 2 (Figures 2, 4, and 5; Table 3). However, in Tier 3, specific parameters can be modified or refined to improve accuracy and precision of the assessment, evaluate different risk related assumptions, incorporate spatial and temporal variation into the assessment, or evaluate specific subareas, contaminant gradients, or potential hotspots.

Tier 3 is performed only after the Tier 2 assessment is completed. To proceed with Tier 3 assessment, the conceptual site model or monitoring data should suggest site conditions sufficiently different from the Tier 2 assumptions to potentially affect the consumption risk or sediment linkage categorical outcomes. If a Tier 3 analysis is used, the specific modifications to the approach should be determined by the management and information needs for the site in question.

For Tier 3 evaluation of consumption risk, modifications of the analysis might include use of data from local seafood consumption surveys (e.g., rate of seafood consumption, species consumed, or dietary proportions). Potential Tier 3 modifications to the sediment linkage analysis based on local data could include revisions to standardized bioaccumulation model parameters (e.g., growth or assimilation rates), food web structure, seafood movement patterns and range, and sediment concentration measurements more representative of actual fish forage area. Additionally, refinements to the contaminant bioaccumulation model may be performed based on site-specific information on contaminant fate and transport, including sediment dynamics, seasonal dynamics, and external inputs (e.g., from the watershed).

DISCUSSION

The assessment approach is designed to meet California's regulatory need to determine whether sediment contamination meets the narrative SQO (i.e., no unacceptable risk to human health). The present approach, therefore, focuses entirely on bioaccumulation and seafood exposure, using measured concentrations in sediments and seafood, and a mechanistic model. This differs from several prior sediment quality assessment frameworks, which include bioaccumulation as just one of multiple lines of evidence (MLOE) (USEPA

Table 3. Provisional site assessment for indirect effects SQO, combining consumption risk and sediment linkage [a]

Consumption risk rating	Sediment linkage rating	Narrative description	Final category
1	1	Virtually all of the seafood consuming population is at an acceptable risk from seafood contamination; very little of the seafood tissue burden is due to site sediments	Unimpacted
1	2	Virtually all of the seafood consuming population is at an acceptable risk from seafood contamination; a limited amount of the seafood tissue burden is due to site sediments	Unimpacted
1	3	Virtually all of the seafood consuming population is at an acceptable risk from seafood contamination; a substantial portion of the seafood tissue burden is due to site sediments	Unimpacted
1	4	Virtually all of the seafood consuming population is at an acceptable risk from seafood contamination; most of the seafood tissue burden is due to site sediments	Unimpacted
2	1	A small portion of seafood consumers with an unacceptable risk from seafood contamination; very little of the seafood tissue burden is due to site sediments	Unimpacted
2	2	A small portion of seafood consumers with an unacceptable risk from seafood contamination; a limited amount of the seafood tissue burden is due to site sediments	Unimpacted
2	3	A small portion of seafood consumers with an unacceptable risk from seafood contamination; a substantial portion of the seafood tissue burden is due to site sediments	Likely unimpacted
2	4	A small portion of seafood consumers with an unacceptable risk from seafood contamination; most of the seafood tissue burden is due to site sediments	Likely unimpacted
3	1	Many seafood consumers with an unacceptable risk from seafood contamination; very little of the seafood tissue burden is due to site sediments	Likely unimpacted
3	2	Many seafood consumers with an unacceptable risk from seafood contamination; a limited amount of the seafood tissue burden is due to site sediments	Possibly impacted
3	3	Many seafood consumers with an unacceptable risk from seafood contamination; a substantial portion of the seafood tissue burden is due to site sediments	Likely impacted
3	4	Many seafood consumers with an unacceptable risk from seafood contamination; most of the seafood tissue burden is due to site sediments	Clearly Impacted
4	1	Most seafood consumers with an unacceptable risk from seafood contamination; very little of the seafood tissue burden is due to site sediments	Likely unimpacted
4	2	Most seafood consumers with an unacceptable risk from seafood contamination; a limited amount of the seafood tissue burden is due to site sediments	Possibly impacted
4	3	Most seafood consumers with an unacceptable risk from seafood contamination; a substantial portion of the seafood tissue burden is due to site sediments	Likely impacted
4	4	Most seafood consumers with an unacceptable risk from seafood contamination; most of the seafood tissue burden is due to site sediments	Clearly impacted

1 = very low; 2 = low; 3 = moderate; 4 = high; SQO = Sediment Quality Objective.

[a] Five categories of result are obtained.

1991, 1998a; McCauley et al. 2000; Chapman and Anderson 2005; Batley and Simpson 2008). The separation of bioaccumulation into its own assessment, rather than as part of a MLOE approach follows from California's present focus on 2 types of sediment contamination impacts. The first type of impact, direct effects to benthic communities, is evaluated based on sediment quality triad data (chemistry, toxicity, and invertebrate community bioassessment) integrated using a validated framework (Bay and Weisberg 2010). The present framework evaluates the second type of impact, namely indirect effects on human health via food web biomagnification. Because of the complexity of indirect effects assessment (Moore et al. 2005; Gobas and Arnot 2010), this separate approach has been developed, based on a tiered framework.

The assessment practices used are based on well-accepted scientific methods and principles. These include application of the risk assessment paradigm (USEPA 1998b; Cura et al. 1999; Faustman and Omenn 2010; Health Canada 2010a, 2010b; Kwok et al. 2014), empirical measurement of contamination in both the environmental and exposure source (i.e., sediment and seafood) (USEPA 2000b; Chapman and Anderson 2005), evaluation of the linkage using well established and mechanistic bioaccumulation models (Burkhard 1998; Arnot and Gobas 2004; Gobas and Arnot 2010; Parkerton and Connolly 2013), and incorporation of methods to address variability and uncertainty (Thompson and Graham 1996). Finally, the use of categories to communicate the assessment results facilitates evaluation and prioritization of sites by end users having varying levels of technical expertise (Canadian Council of Ministers of the Environment 2008; Bay and Weisberg 2010).

Successful environmental assessment in a policy context requires a clearly defined and transparent integration of scientific information with societal value judgments (Kwok et al. 2014). This has been a central goal of the SQO development process and framework. The framework, therefore, contains clear points where parameters and thresholds can be defined based on policy decisions (Kwok et al. 2014). For the consumption risk evaluation, these include the portion of the human population to be protected and the allowable excess cancer risk to that population. For the sediment linkage evaluation, these include the magnitude of linkage (i.e., association between site sediments and observed seafood contamination) that is considered to represent a significant influence of site sediments. Key parameters and decision points in the framework will be explicitly included in public documents and discussion, as part of California's oversight and review process before final consideration by the State Water Board. This will facilitate input from the USEPA, other government agencies, and interested parties regarding what is an acceptable level of exposure. The seafood consumer population to be protected (e.g., general population, fishing population, population who consumes their catch, or subsistence fisher population) will be key to this discussion.

The inclusion of probabilistic information regarding exposure and risk, is intended to facilitate discussion of the management and policy issues that arise when evaluating potential health effects of contaminated sediment (Thompson and Graham 1996). However, a probabilistic framework must also inform sample collection. To ensure that an inference can be drawn regarding the entire site area, probabilistic sediment survey designs should be used, with stratification as needed to address areas of particular concern, rather than targeted or ad hoc sampling (USEPA 2002; Lowe et al. 2004; Stevens and Olsen 2004; Olsen et al. 2009; Bay et al. 2011).

The framework is tiered, which allows rapid screening of multiple sites, focusing resources and effort on evaluating those sediments for which greater uncertainty, risk, or management implications warrant greater attention (Bridges et al. 2005; Chapman and Anderson 2005; Hope 2009). Tier 1 is intended to be a screening assessment, and Tier 3 a complete risk assessment (Hope 2009). Tier 2 is intermediate in nature and can be conceptualized as a focused risk assessment. In Tier 2, some aspects of the evaluation are predetermined, to constrain effort and achieve standardization, whereas highly influential parameters, such as contaminant concentrations, sediment organic carbon, and target seafood species, are measured and evaluated locally. Although one could argue that a complete site-specific risk assessment (i.e., Tier 3) should be undertaken in all circumstances, we feel that this would inhibit implementation of the approach. Requiring the highest level of effort for all situations, including sites likely to pose little risk, would consume excessive time and resources, reduce the overall rate of application of the framework, and delay progress in identifying, prioritizing, and remediating sites of greatest concern. Furthermore, using different site assessment approaches in each evaluation would neglect the considerable progress that has been made in developing consistent approaches for indirect effects assessment, and would eliminate the potential benefits of standardized methods for comparing and prioritizing among sites.

Tier 3 is likely to be applied in complex situations when the standardized assumptions in Tiers 1 and 2 would be too simplistic to evaluate local exposure with sufficient accuracy. Examples include large ports or harbors where there are multiple inputs, multiple contaminants, and high spatial and temporal variability in contamination and exposure. Tier 3 can also be beneficial in identifying data needs for quantitative evaluation of potential management alternatives (USEPA 2005), for which consideration of site-specific attributes is essential.

It is illustrative to compare the SQO framework to the USEPA and USACE 4-tiered evaluation approach (USEPA 1991, 1998a), which is currently recommended by the USEPA for dredged sediment evaluation for off-site disposal (USEPA 2014). The conceptual basis for tiering is the same: focusing resources on evaluating situations with unclear impacts. Both approaches include an initial screening Tier 1 with existing data and an optional final tier (Tier 3 for SQO and Tier 4 in the USEPA/USACE approach) for cases where more detailed evaluation is needed. However, the USEPA/USACE approach combines direct and indirect effects assessment. The USEPA/USACE indirect effects evaluation directly compares laboratory or field benthic invertebrate tests to USFDA Action Levels or to reference sediment results. This neglects the issue of food web trophic transfer increasing exposure to consumers, which the SQO framework addresses via direct measurement of seafood tissue, and food web bioaccumulation models to evaluate sediment linkage. Rather than simply using the USFDA Action Levels, the SQO framework follows USEPA guidance to develop tissue thresholds based on local seafood consumption rates, latest available compound-specific risk information, and acceptable risk.

The proposed SQO framework is limited in scope, and has areas for future expansion. The focus on legacy organochlorine compounds (PCBs and legacy pesticides) was intended to increase the likelihood of successful implementation given technical challenges, stakeholder concerns, and resource and time limitations. There is a need to develop similar approaches for other sediment-associated pollutants of regional concern, including halogenated flame retardants (Klosterhaus et al. 2012), and a variety of contaminants of emerging concern (Oros et al. 2003; Muir and Howard 2006). Mercury, in particular, is a compound of state and region-wide concern (Wiener and Suchanek 2008; Davis et al. 2012), for which there have been recent advances in modeling source apportionment (Knightes et al. 2009; Harris et al. 2012), providing a technical foundation for assessing indirect effects of sediment contamination.

The current framework evaluates risk on an individual contaminant basis, separately examining sum of PCBs, sum of DDTs, and other organochloride pesticides. This implicitly assumes independent action, and is less conservative than combining risks of multiple contaminants via dose addition or synergistic effects (Kortenkamp et al. 2007). Mechanistic approaches to combine effects of multiple contaminants are well established for dioxin-TEQs (van den Berg et al. 2006), and some policies and risk assessments assume additivity for cancer or noncancer effects across contaminant classes (Wilson et al. 2001; Washington Department of Ecology 2013). Our decision to evaluate contaminants individually was consistent with current practice for evaluating fish contamination effects in California (Klasing and Brodberg 2008), and reflects the fact that the different compound classes have different sources and resulting control measures (SWRCB 2006; Connor et al. 2007; Davis et al. 2007).

Finally, although the framework was developed for human health assessment, the approach should also be applicable with minor modifications to evaluating risks to fish and wildlife (Wenning et al. 2005; Gobas and Arnot 2010; Alava et al. 2012). Framework elements appropriate for fish and wildlife assessment include the key questions focusing on consumption risk and sediment contribution, conceptual site model development, 3-tiered approach, bioaccumulation model, combination of statewide and site-specific parameters, feeding guild approach, and explicit consideration of spatial movement versus site area. To address risk to fish and wildlife would require selection of new indicator species and toxicity endpoints, development of risk thresholds for consumption risk to fish and wildlife, and selection of appropriate bioaccumulation model parameters. The inclusion of fish and wildlife would complement the current California SQO approach to sediment assessment, which evaluates both direct effects to benthic organisms and indirect effects to humans. This integration of multiple frameworks and tools will ultimately achieve the goal of a complete and protective approach for implementing sediment quality objectives in California.

Acknowledgment—Key guidance on developing the framework was provided by the SQO Scientific Steering Committee: TS Bridges, RM Burgess, TH Gries, BK Hope, PF Landrum, CA Menzie, ER Long, JP Shine, RF Van Dolah, and DJ Vorhees. We also appreciate the technical contributions by members of the SQO Science Team, including C Beegan, RK Brodberg, MS Connor, JA Arnot, FAPC Gobas, and SB Weisberg. Finally, S Anghera, BB Bernstein, W Gala, J Gully, E Kimura, L Haney, DW Moore, S Paulson, and other members of the Sediment Quality Advisory Committee provided useful feedback throughout the SQO Program development process. TE McKone, C Beegan, and 3 anonymous reviewers reviewed the manuscript. Financial support was provided by the State of California (Contract 11–75-250). BG was also supported by a USEPA STAR Fellowship and a NSF SAGE-IGERT Traineeship. This is SFEI Publication 716. The contents of this article represent the views of the authors alone, and not necessarily those of the funding agencies.

SUPPLEMENTAL DATA

Supplemental text on development of species list, guild categories, indicator species, and food web matrix.

Table S1. Dietary guild categories used for SQO species.

Table S2. Food web structure for marine embayments used in bioaccumulation model.

Table S3. Food web structure for the estuarine Sacramento-San Joaquin Delta used in bioaccumulation model.

Table S4. Biota–sediment accumulation factors (BSAF, g tissue ww/g sediment dw) for chlordanes in Tier I assessment.

Table S5. Biota–sediment accumulation factors (BSAF, g tissue ww/g sediment dw) for DDTs in Tier I assessment.

Table S6. Biota–sediment accumulation factors (BSAF, g tissue ww/g sediment dw) for dieldrin in Tier I assessment.

Table S7. Biota–sediment accumulation factors (BSAF, g tissue ww/g sediment dw) for PCBs in Tier I assessment.

Table S8. Potential categories for the assessment of consumption risk.

Table S9. Potential categories for the assessment of sediment linkage.

REFERENCES

Alava JJ, Ross PS, Lachmuth C, Ford JKB, Hickie BE, Gobas FAPC. 2012. Habitat-based PCB environmental quality criteria for the protection of endangered killer whales (*Orcinus orca*). *Environ Sci Technol* 46:12655–12663.

Anchor Environmental. 2005. Focused ecological risk assessment-bioaccumulation modeling. Rhine Channel sediment remediation Newport Bay, California. Seattle (WA): Prepared for Orange County Coastkeeper.

Anderson DW, Jehl JR, Risebrough RW, Woods LA, Deweese LR, Edgecomb WG. 1975. Brown pelicans: Improved reproduction off the southern California coast. *Science* 190:806–808.

Apitz SE, Davis JW, Finkelstein K, Hohreiter DW, Hoke R, Jensen RH, Jersak J, Kirtay VJ, Mack EE, Magar VS, et al. 2005. Assessing and managing contaminated sediments: Part II, evaluating risk and monitoring sediment remedy effectiveness. *Integr Environ Assess Manag* 1:e1–e14.

Arnot JA, Gobas FAPC. 2004. A food web bioaccumulation model for organic chemicals in aquatic ecosystems. *Environ Toxicol Chem* 23:2343–2355.

Batley G, Simpson S. 2008. Advancing Australia's sediment quality guidelines. *Australas J Ecotoxicol* 14:11.

Bay SM, Greenstein DJ, Jacobe M, Barton C, Sakamoto K, Young D, Ritter KJ, Schiff KC. 2011. Southern California Bight 2008 Regional Monitoring Program: I. Sediment toxicity. Costa Mesa (CA): Southern California Coastal Water Research Project.

Bay SM, Weisberg SB, 2010. Framework for interpreting sediment quality triad data. *Integr Environ Assess Manag* 8:589–596.

Beegan C, Bay SM. 2012. Transitioning sediment quality assessment into regulations: Challenges and solutions in implementing California's sediment quality objectives. *Integr Environ Assess Manag* 8:586–588.

Beyer WN, Heinz GH, Redmon-Norwood AW. 1996. Environmental contaminants in wildlife: Interpreting tissue concentrations. Boca Raton (FL): Lewis. 494 pp.

Birkeland S. 2001. EPA's TMDL Program. *Ecol Law Q* 28:297–305.

Bogen KT, Spear RC. 1987. Integrating uncertainty and interindividual variability in environmental risk assessment. *Risk Anal* 7:427–436.

Bridges TS, Berry WJ, Della Sala S, Dorn PB, Ells SJ, Gries TH, Ireland DS, Maher EM, Menzie CA, Porebski LM, et al. 2005. A framework for assessing and managing risks from contaminated sediments. In: Wenning RJ, Batley GE, Ingersoll CG, Moore DW, editors. Use of sediment quality guidelines and related tools for the assessment of contaminated sediments. Pensacola (FL): Society of Environmental Toxicology and Chemistry. p 227–266.

Burkhard LP. 1998. Comparison of two models for predicting bioaccumulation of hydrophobic organic chemicals in a Great Lakes food web. *Environ Toxicol Chem* 17:383–393.

Burkhard LP, Cook PM, Lukasewycz MT. 2010. Direct application of biota-sediment accumulation factors. *Environ Toxicol Chem* 29:230–236.

Canadian Council of Ministers of the Environment. 1999. Protocol for the derivation of Canadian sediment quality guidelines for the protection of aquatic life. CCME EPC-98E. In: Canadian Environmental Quality Guidelines. Winnipeg, Manitoba.

Canadian Council of Ministers of the Environment. 2008. National Classification System for Contaminated Sites: Guidance document. Winnipeg, Manitoba.

Chapman PM, Anderson J. 2005. A decision-making framework for sediment contamination. *Integr Environ Assess Manag* 1:163–173.

Chapman PM, Anderson B, Carr S, Engle V, Green R, Hameedi J, Harmon M, Haverland P, Hyland J, Ingersoll C, et al. 1997. General guidelines for using the sediment quality triad. *Mar Pollut Bull* 34:368–372.

Chen CY, Borsuk ME, Bugge DM, Hollweg T, Balcom PH, Ward DM, Williams J, Mason RP. 2014. Benthic and pelagic pathways of methylmercury bioaccumulation in estuarine food webs of the northeast United States. *PLoS One* 9: e89305.

Condon CD, Gobas FAPC. 2007. Development, evaluation, and application of a food web bioaccumulation model for PCBs in the Strait of Georgia, British Columbia [MSEM thesis]. Vancouver (BC): Simon Fraser Univ.

Connolly JP. 1991. Application of a food chain model to PCB contamination of the lobster and winter flounder food chains in New Bedford Harbor. *Environ Sci Technol* 25:760–769.

Connor MS, Davis JA, Leatherbarrow J, Greenfield BK, Gunther A, Hardin D, Mumley T, Oram JJ, Werme C. 2007. The slow recovery of San Francisco Bay from the legacy of organochlorine pesticides. *Environ Res* 105:87–100.

Contaminated Sites Management Working Group. 1999. A federal approach to contaminated sites. Ottawa, Ontario: Dillon Consulting Limited.

Cura JJ, Heiger-Bernays W, Bridges TS, Moore DW. 1999. Ecological and human health risk assessment guidance for aquatic environments. Chelmsford (MA): US Army Corps of Engineers.

Davis JA, Hetzel F, Oram JJ, McKee LJ. 2007. Polychlorinated biphenyls (PCBs) in San Francisco Bay. *Environ Res* 105:67–86.

Davis JA, Looker RE, Yee D, Marvin-Dipasquale M, Grenier JL, Austin CM, McKee LJ, Greenfield BK, Brodberg R, Blum JD. 2012. Reducing methylmercury accumulation in the food webs of San Francisco Bay and its local watershed. *Environ Res* 119:3–26.

Davis JA, Schiff K, Melwani AR, Bezalel SN, Hunt JA, Allen R, Ichikawa G, Bonnema A, Heim W, Crane D, et al. 2011. SWAMP/RMP/Bight program report on contaminants in fish from the California coast. Sacramento (CA): California State Water Resources Control Board.

Faustman EM, Omenn GS. 2010. Chapter 4. Risk assessment. In: Klaassen CD, Watkins III JB, editors. Casarett and Doull's essentials of toxicology. 2nd ed. New York (NY): McGraw-Hill. p 139–164.

Gehrke GE, Blum JD, Slotton DG, Greenfield BK. 2011. Mercury isotopes link mercury in San Francisco Bay forage fish to surface sediments. *Environ Sci Technol* 45:1264–1270.

Gewurtz SB, Laposa R, Gandhi N, Christensen GN, Evenset A, Gregor D, Diamond ML. 2006. A comparison of contaminant dynamics in arctic and temperate fish: A modeling approach. *Chemosphere* 63:1328–1341.

Gobas FAPC, Arnot JA. 2010. Food web bioaccumulation model for polychlorinated biphenyls in San Francisco Bay, California, USA. *Environ Toxicol Chem* 29:1385–1395.

Greenfield BK, Allen RM. 2013. Polychlorinated biphenyl spatial patterns in San Francisco Bay forage fish. *Chemosphere* 90:1693–1703.

Gustavson K, von Stackelberg T, Linkov I. 2011. Bioaccumulation models: State of the application at large superfund sites. Vicksburg (MS): Army Engineer Research and Development Center. ERDC TN-DOER-R17.

Gustavson KE, Barnthouse LW, Brierley CL, Clark EH, Ward CH. 2007. Superfund and mining megasites. *Environ Sci Technol* 41:2667–2672.

Hamilton JT, Viscusi WK. 1999. How costly is "Clean"? An analysis of the benefits and costs of superfund site remediations. *J Policy Anal Manag* 18:2–27.

Harris R, Pollman C, Landing W, Evans D, Axelrad D, Hutchinson D, Morey SL, Rumbold D, Dukhovskoy D, Adams DH, et al. 2012. Mercury in the Gulf of Mexico: Sources to receptors. *Eviron Res* 119:42–52.

Health Canada, 2010a. Federal contaminated site risk assessment in Canada, Part I: Guidance on human health preliminary quantitative risk assessment (PQRA), Version 2.0. Ottawa, Ontario: Health Canada.

Health Canada, 2010b. Federal contaminated site risk assessment in Canada, Part V: Guidance on human health detailed quantitative risk assessment (DQRA), Version 2.0. Ottawa, Ontario: Health Canada.

Hesslein RH, Capel MJ, Fox DE, Hallard KA. 1991. Stable isotopes of sulfur, carbon, and nitrogen as indicators of trophic level and fish migration in the Lower Mackenzie River Basin, Canada. *Can J Fish Aquat Sci* 48:2258–2265.

Hin JA, Osté LA, Schmidt CA. 2010. Guidance document for sediment assessment. Methods to determine to what extent the realization of water quality objectives of a water system is impeded by contaminated sediments. Netherlands: Rijkswaterstaat Centre for Water Management.

Hope BK. 1995. A review of models for estimating terrestrial ecological receptor exposure to chemical contaminants. *Chemosphere* 30:2267–2287.

Hope BK. 2009. Will there ever be a role for risk assessments? *Hum Ecol Risk Assess* 15:1–6.

Huang X, Hites RA, Foran JA, Hamilton C, Knuth BA, Schwager SJ, Carpenter DO. 2006. Consumption advisories for salmon based on risk of cancer and noncancer health effects. *Environ Res* 101:263–274.

Jassby AD, Cloern JE, Powell TM. 1993. Organic carbon sources and sinks in San Francisco Bay—Variability induced by river flow. *Mar Ecol Prog Ser* 95:39–54.

Kidd KA, Schindler DW, Muir DC, Lockhart WL, Hesslein RH. 1995. High concentrations of toxaphene in fishes from a subarctic lake. *Science* 269:240–242.

Klasing S, Brodberg R. 2008. Development of fish contaminant goals and advisory tissue levels for common contaminants in sport fish: chlordane, DDTs, dieldrin, methylmercury, PCBs, selenium, and toxaphene. Oakland (CA): Office of Environmental Health Hazard Assessment.

Klosterhaus SL, Stapleton HM, La Guardia MJ, Greig DJ. 2012. Brominated and chlorinated flame retardants in San Francisco Bay sediments and wildlife. *Environ Int* 47:56–65.

Knightes CD, Sunderland EM, Barber MC, Johnston JM, Ambrose RB. 2009. Application of ecosystem-scale fate and bioaccumulation models to predict fish mercury response times to changes in atmospheric deposition. *Environ Toxicol Chem* 28:881–893.

Kortenkamp A, Faust M, Scholze M, Backhaus T. 2007. Low-level exposure to multiple chemicals: Reason for human health concerns? *Environ Health Perspect* 115(Suppl):106–114.

Kwok KWH, Batley GE, Wenning RJ, Zhu L, Vangheluwe M, Lee S. 2014. Sediment quality guidelines: Challenges and opportunities for improving sediment management. *Environ Sci Pollut Res Int* 21:17–27.

Linkov I, Ames MR, Crouch EAC, Satterstrom FK. 2005. Uncertainty in octanol – water partition coefficient: Implications for risk assessment and remedial costs. *Environ Sci Technol* 39:6917–6922.

Linkov I, Burmistrov D, Cura J, Bridges TS. 2002. Risk-based management of contaminated sediments: Consideration of spatial and temporal patterns in exposure modeling. *Environ Sci Technol* 36:238–246.

Long ER, Chapman PM. 1985. A Sediment Quality Triad: Measures of sediment contamination, toxicity and infaunal community composition in Puget Sound. *Mar Pollut Bull* 16:405–415.

Lowe S, Thompson B, Hoenicke R, Leatherbarrow J, Taberski K, Smith R, Stevens D Jr. 2004. Re-design process of the San Francisco Estuary Regional Monitoring Program for trace substances (RMP) status and trends monitoring component for water and sediment. Oakland (CA): San Francisco Estuary Institute. SFEI Contribution Number 109.

Maruya KA, Landrum PF, Burgess RM, Shine JP. 2012. Incorporating contaminant bioavailability into sediment quality assessment frameworks. *Integr Environ Assess Manag* 8:659–673.

McCauley DJ, DeGraeve G, Linton T. 2000. Sediment quality guidelines and assessment: Overview and research needs. *Environ Sci Policy* 3:133–144.

McKone TE, Bogen KT. 1991. Predicting the uncertainties in risk assessment. *Environ Sci Technol* 25:1674–1681.

Melwani AR, Greenfield BK, Byron ER. 2009. Empirical estimation of biota exposure range for calculation of bioaccumulation parameters. *Integr Environ Assess Manag* 5:138–149.

Moore DW, Baudo R, Conder JM, Landrum PF, McFarland VA, Meador JP, Millward RN, Shine JP, Word JQ. 2005. Bioaccumulation in the assessment of sediment quality: Uncertainty and potential application. In: Wenning RJ, Batley GE, Ingersoll CG, Moore DW, editors. Use of sediment quality guidelines and related tools for the assessment of contaminated sediments. Pensacola (FL): SETAC. p 429–495.

Muir DCG, Howard PH. 2006. Are there other persistent organic pollutants? A challenge for environmental chemists. *Environ Sci Technol* 40:7157–7166.

Olsen AR, Snyder BD, Stahl LL, Pitt JL. 2009. Survey design for lakes and reservoirs in the United States to assess contaminants in fish tissue. *Environ Monit Assess* 150:91–100.

Oros DR, Jarman WM, Lowe T, David N, Lowe S, Davis JA. 2003. Surveillance for previously unmonitored organic contaminants in the San Francisco Estuary. *Mar Pollut Bull* 46:1102–1110.

Parkerton TF, Connolly JP. 2013. Modeling bioaccumulation in coupled pelagic-benthic food chains: past insights and future directions. *Environ Toxicol Chem* 32:1931–1934.

Saloranta TM, Andersen T, Naes K. 2006. Flows of dioxins and furans in coastal food webs: Inverse modeling, sensitivity analysis, and applications of linear system theory. *Environ Toxicol Chem* 25:253.

Saloranta TM, Ruus A, Borgå K. 2011. Identification of the most influential factors in the Norwegian guidelines for risk assessment of dispersion of contaminants from sediments. *Integr Environ Assess Manag* 7:657–667.

Schaeffer DJ, Dellinger JA, Needham LL, Hansen LG. 2006. Serum PCB profiles in Native Americans from Wisconsin based on region, diet, age, and gender: Implications for epidemiology studies. *Sci Total Environ* 357:74–87.

San Francisco Estuary Institute. 2000. San Francisco Bay Seafood Consumption Study. Richmond (CA): SFEI, California Department of Health Services. 291 pp.

State of Washington. 1995. Chapter 173–204 WAC, Sediment Management Standards. Olympia (WA): Washington State Department of Ecology.

Stevens DL Jr, Olsen AR. 2004. Spatially balanced sampling of natural resources. *J Am Stat Assoc* 99:262–278.

State Water Resources Control Board. 2006. Total maximum daily loads for organochlorine compounds. San Diego Creek: total DDT and toxaphene. Upper and Lower Newport Bay: total DDT, chlordane, total PCBs. Riverside (CA): California Regional Water Quality Control Board—Santa Ana region.

State Water Resources Control Board. 2009. Water quality control plan for enclosed bays and estuaries. Part 1. Sediment quality. Sacramento (CA): California State Water Resources Control Board. 27 pp.

Thomann RV, Connolly JP, Parkerton TF. 1992. An equilibrium model of organic chemical accumulation in aquatic food webs with sediment interaction. *Environ Toxicol Chem* 11:615–629.

Thompson KM, Graham JD. 1996. Going beyond the single number: Using probabilistic risk assessment to improve risk management. *Hum Ecol Risk Assess* 2:1008–1034.

[USEPA] US Environmental Protection Agency. 1991. Evaluation of dredged material proposed for ocean disposal: Testing manual. US Army Corps of Engineers. Washington DC: USEPA Office of Water. EPA 503/8-91/001.

[USEPA] US Environmental Protection Agency. 1998a. Evaluation of dredged material proposed for discharge in waters of the US: Testing manual (Inland Testing Manual). US Army Corps of Engineers. Washington DC: USEPA. EPA-823-B-98-004.

[USEPA] US Environmental Protection Agency. 1998b. Guidelines for ecological risk assessment. Washington DC: USEPA. EPA/630/R-95/002F.

[USEPA] US Environmental Protection Agency. 2000a Guidance for assessing chemical contaminant data for use in fish advisories. 3rd ed. Vol 2. Risk assessment and fish consumption limits. Washington DC: USEPA. EPA-823-B-00-008.

[USEPA] US Environmental Protection Agency. 2000b Guidance for assessing chemical contaminant data for use in fish advisories. 3rd ed. Vol 1. Fish sampling and analysis. Washington DC: USEPA. EPA-823-B-00-007.

[USEPA] US Environmental Protection Agency. 2002. Guidance on choosing a sampling design for environmental data collection for use in developing a Quality Assurance Project Plan. Washington DC: USEPA Office of Environmental Information. EPA/240/R-02/005.

[USEPA] US Environmental Protection Agency. 2004. Technical memorandum. Origin of 1 meal/week noncommercial fish consumption rate in National Advisory for Mercury. March 11 2004. Washington DC: USEPA. 5 pp. Available from: http://water.epa.gov/scitech/swguidance/fishshellfish/outreach/upload/2004_07_21_fish_advice_1-meal-per-week.pdf

[USEPA] US Environmental Protection Agency. 2005. Contaminated sediment remediation guidance for hazardous waste sites. Washington DC: USEPA Office of Solid Waste and Emergency Response. EPA-540-R-05-012.

[USEPA] US Environmental Protection Agency. 2014. Dredged material testing and evaluation. [cited 2014 June 2]. Available from: http://water.epa.gov/type/oceb/oceandumping/dredged material/testing.cfm.

van den Berg M, Birnbaum LS, Denison M, De Vito M, Farland W, Feeley M, Fiedler H, Hakansson H, Hanberg A, Haws L, et al. 2006. The 2005 World Health Organization reevaluation of human and mammalian toxic equivalency factors for dioxins and dioxin-like compounds. *Toxicol Sci* 93:223–241.

von Stackelberg K, Fogarty K, Cura JJ, DeShields B, Erickson M, Henderson M, Bridges TS. 2003. Comparative risk assessment of dredged material management alternatives, Moss Landing Harbor, Moss Landing, California. Chelmsford (MA): Menzie-Cura and Associates.

von Stackelberg KE, Burmistrova A. 2003. TrophicTrace Users Manual: A tool for assessing risks from trophic transfer of sediment-associated contaminants. Chelmsford (MA): Menzie-Cura and Associates.

von Stackelberg KE, Burmistrov D, Vorhees DJ, Bridges TS, Linkov I. 2002. Importance of uncertainty and variability to predicted risks from trophic transfer of PCBs in dredged sediments. *Risk Anal* 22:499–512.

Washington Department of Ecology. 2013. Sediment management standards. Chapter 173–204 WAC. Publication 13–09-055. Olympia, WA.

Wenning RJ, Batley GE, Ingersoll CG, Moore DW. 2005. Use of sediment quality guidelines and related tools for the assessment of contaminated sediments. Pensacola (FL): SETAC. 815 p.

Wickwire T, Johnson MS, Hope BK, Greenberg MS. 2011. Spatially explicit ecological exposure models: A rationale for and path toward their increased acceptance and use. *Integr Environ Assess Manag* 7:158–168.

Wiener JG, Suchanek TH. 2008. The basis for ecotoxicological concern in aquatic ecosystems contaminated by historical mercury mining. *Ecol Appl* 18:A3–A11.

Wilson ND, Price PS, Paustenbach DJ. 2001. An event-by-event probabilistic methodology for assessing the health risks of persistent chemicals in fish: A case study at the Palos Verdes Shelf. *J Toxicol Environ Health A* 62:595–642.

Wong CS, Capel PD, Nowell LH. 2001. National-scale, field-based evaluation of the biota-sediment accumulation factor model. *Environ Sci Technol* 35:1709–1715.

Zeng EY, Tran K. 2002. Distribution of chlorinated hydrocarbons in overlying water, sediment, polychaete, and hornyhead turbot (*Pleuronichthys verticalis*) in the coastal ocean, southern California, USA. *Environ Toxicol Chem* 21:1600–1608.