SETAC PRESS

Environmental Toxicology and Chemistry, Vol. 19, No. 5, pp. 1403–1413, 2000
Printed in the USA
0730-7268/00 $9.00 + .00

DEVELOPMENT AND EVALUATION OF CONSENSUS-BASED SEDIMENT EFFECT CONCENTRATIONS FOR POLYCHLORINATED BIPHENYLS

DONALD D. MACDONALD,*† LISA M. DIPINTO,‡ JAY FIELD,§ CHRISTOPHER G. INGERSOLL,|| EDWARD R. LONG,# and RICHARD C. SWARTZ††

†MacDonald Environmental Sciences Ltd., 2376 Yellow Point Road, Nanaimo, British Columbia V9X 1W5, Canada
‡ORCA Damage Assessment Center, National Oceanic and Atmospheric Administration, 1305 East-West Highway, Room 10218, Silver Spring, Maryland 20910-3281, USA
§Hazardous Materials Response and Assessment Division, National Oceanic and Atmospheric Administration, 7600 Sand Point Way NE, Seattle, Washington 98115, USA
||Columbia Environmental Research Center, U.S. Geological Survey, 4200 New Haven Road, Columbia, Missouri 65201
#Coastal Monitoring and Bioeffects Assessment Division, National Oceanic and Atmospheric Administration, 7600 Sand Point Way NE, Seattle, Washington DC 98115, USA
††P.O. Box 397, Placida, Florida 33946-0397, USA

(*Received* 19 *March* 1999; *Accepted* 17 *September* 1999)

Abstract—Sediment-quality guidelines (SQGs) have been published for polychlorinated biphenyls (PCBs) using both empirical and theoretical approaches. Empirically based guidelines have been developed using the screening-level concentration, effects range, effects level, and apparent effects threshold approaches. Theoretically based guidelines have been developed using the equilibrium-partitioning approach. Empirically-based guidelines were classified into three general categories, in accordance with their original narrative intents, and used to develop three consensus-based sediment effect concentrations (SECs) for total PCBs (tPCBs), including a threshold effect concentration, a midrange effect concentration, and an extreme effect concentration. Consensus-based SECs were derived because they estimate the central tendency of the published SQGs and, thus, reconcile the guidance values that have been derived using various approaches. Initially, consensus-based SECs for tPCBs were developed separately for freshwater sediments and for marine and estuarine sediments. Because the respective SECs were statistically similar, the underlying SQGs were subsequently merged and used to formulate more generally applicable SECs. The three consensus-based SECs were then evaluated for reliability using matching sediment chemistry and toxicity data from field studies, dose-response data from spiked-sediment toxicity tests, and SQGs derived from the equilibrium-partitioning approach. The results of this evaluation demonstrated that the consensus-based SECs can accurately predict both the presence and absence of toxicity in field-collected sediments. Importantly, the incidence of toxicity increases incrementally with increasing concentrations of tPCBs. Moreover, the consensus-based SECs are comparable to the chronic toxicity thresholds that have been estimated from dose-response data and equilibrium-partitioning models. Therefore, consensus-based SECs provide a unifying synthesis of existing SQGs, reflect causal rather than correlative effects, and accurately predict sediment toxicity in PCB-contaminated sediments.

Keywords—Polychlorinated biphenyls   Mixtures   Sediment quality guidelines   Sediment   Toxicity

INTRODUCTION

A variety of theoretical and empirical approaches have been used to create sediment quality guidelines (SQGs) for polychlorinated biphenyls (PCBs) in freshwater, estuarine, and marine ecosystems. These approaches include the equilibrium-partitioning (EqP) approach [1,2], screening-level concentration approach [3], effects range approach [4], effects level approach [5], and apparent effects threshold (AET) approach [6]. Application of these approaches has resulted in a wide range of SQGs for PCBs. For example, the SQGs for assessing the potential effects of total (tPCBs) in freshwater sediments span more than three orders of magnitude [3,4,7–11]. Similarly, the SQGs for assessing the potential effects of tPCBs in marine sediments span more than two orders of magnitude [5,12–15]. Differences among the numeric SQGs as well as questions regarding the bioavailability of sediment contaminants, effects of varying chemicals and chemical mixtures, ecologic relevance of certain SQGs, and determination of causality, however, have made it difficult for users of SQGs to select the tools that are most relevant for their specific application.

The purpose of this paper is to resolve some of these difficulties by providing a unifying synthesis of the published freshwater, estuarine, and marine SQGs for PCBs. To this end, published SQGs for PCBs were assembled and classified in accordance with their narrative intent, and the SQGs that fell within three general categories were used to develop consensus-based sediment effect concentrations (SECs). Specifically, a threshold effect concentration (TEC; below which adverse effects are unlikely to occur), a midrange effect concentration (MEC; above which adverse effects frequently occur), and an extreme effect concentration (EEC; above which adverse effects usually or always occur) were established. Consensus-based SECs were derived because they provide a means of reconciling SQGs that have been developed using the various empirically based approaches. The consensus-based SECs were then evaluated regarding their ability to predict sediment toxicity in field-collected sediments from various locations in the United States. The consensus-based SECs were also evaluated to determine if they reflected causal rather than correlative effects as indicated using the results of spiked-sediment

* To whom correspondence may be addressed (sff-mesl@island.net).
The views expressed herein are those of the authors and do not necessarily reflect the views of the National Oceanic and Atmospheric Administration or any of its subagencies.

toxicity tests (with single chemicals and simple chemical mixtures) and EqP models.

The consensus-based SECs presented in this paper are intended to provide a basis for assessing the potential effects of PCBs on sediment-dwelling organisms. However, PCBs also bioaccumulate in the tissues of aquatic organisms and cause adverse effects in the food web. Therefore, the consensus-based SECs should not be used alone to assess sediment quality. Other tools, such as bioaccumulation tests, tissue chemistry data, and tissue residue guidelines, are also needed to evaluate the potential effects of PCBs on both wildlife and human health.

MATERIALS AND METHODS

Derivation of consensus-based SECs

A stepwise approach was used to develop the consensus-based SECs for PCBs. First, published SQGs for PCBs that have been developed by various investigators to support quality assessments of freshwater, estuarine, and marine sediments were collected and collated. The published SQGs were compiled directly into spreadsheets in MS Excel® format (Microsoft Corporation, Redmond, WA, USA). The SQGs that were expressed on an organic carbon–normalized basis were converted to dry weight (dry wt)–normalized concentrations assuming 1% organic carbon. The 1% organic carbon value was selected because the average levels of organic carbon in marine and estuarine sediments [5] and in the freshwater sediments [9] were similar to this level (1.2 and 1.5%, respectively). The existing SQGs were compiled on a dry wt-normalized basis, because the results of earlier studies have indicated that such tools predict sediment toxicity as well as, or even better than, the organic carbon–normalized SQGs [15,16] and because many of the underlying SQGs were expressed on a dry-weight basis only.

The SQGs were then classified to facilitate the derivation of consensus-based SECs. The SQGs that applied to freshwater sediments and those that applied to marine and estuarine sediments were initially grouped separately. Next, the SQGs were grouped into three categories according to their original narrative intent, including TECs, which were intended to identify concentrations of PCBs below which adverse effects on sediment-dwelling organisms were unlikely to be observed; MECs, which identify concentrations of PCBs above which adverse effects on sediment-dwelling organisms are frequently observed; and EECs, which identify concentrations of PCBs above which adverse effects on sediment-dwelling organisms are usually or always observed [1]. Only the empirically derived SQGs were used to derive the consensus-based SECs. The theoretically derived SQGs were used subsequently to determine if PCBs are likely to cause, or substantially contribute to, sediment toxicity at concentrations greater than the MEC and EEC (rather than simply being associated with toxicity).

Several indicators of central tendency were considered for calculating consensus-based SECs for PCBs, including the arithmetic mean, geometric mean, and median. Each of these indicators has both advantages and limitations that affect their applicability for calculating consensus-based SECs; no perfect indicator of central tendency exists. In this study, the geometric mean was selected to support the calculation of consensus-based SECs for each category of SQGs. This indicator was used because it tends to minimize the effect of single values on the estimate of central tendency and because the distribution of the SQGs within each category was unknown (i.e., arithmetic mean is most appropriate for normally-distributed [17]).

Three SECs, including a TEC, an MEC, and an EEC, derived both for freshwater and for marine and estuarine sediments. The resultant freshwater and marine SECs were compared to determine if they were statistically similar, indicated by a lack of statistical difference based on the results of modified Student's *t* tests. Comparability of the SECs for the two media types was considered to provide sufficient rationale for merging the underlying SQGs to support the derivation of more generally applicable SECs. Final consensus-based SECs were calculated only if three or more SQGs were available in the pooled data set for a chemical substance or group of substances.

Evaluation of consensus-based SECs

The reliability of the consensus-based SECs for assessing sediment quality conditions was evaluated in several ways: by determining their predictive ability (i.e., their ability to correctly classify sediment samples as toxic or not toxic [ ]), by evaluating the degree of concordance between PCB concentrations and the incidence of adverse effects on sediment-dwelling organisms, and by determining if the empirically derived SECs agreed with the results of spiked-sediment toxicity tests and EqP-based SQGs (i.e., to determine if the SECs can be used to determine if PCBs are likely to cause, or substantially contribute to, sediment toxicity).

To support the evaluation of predictive ability, matching sediment chemistry and biological effects data were assembled from a variety of freshwater, estuarine, and marine locations in the United States. Because the candidate data sets were generated for a variety of purposes, each data set was critically evaluated to ensure the quality of the data used for evaluating the predictive ability of the SECs [18]. Data from the following freshwater locations were used: Grand Calumet River and Indiana Harbor Canal, Indiana, USA [19,20]; Indiana Harbor, Indiana, USA [16]; Lower Fox River and Green Bay, Wisconsin, USA [21]; Potomac River, District of Columbia, USA [22–24]; Saginaw River, Michigan, USA [16]; Trinity River, Texas, USA [25]; Upper Mississippi River, Minnesota to Missouri, USA [16,26]; and Waukegan Harbor, Illinois, USA [16,27]. These studies provided 10 data sets (195 sediment samples) with which to evaluate the predictive ability of the SECs for PCBs. Sediment samples were considered to be toxic if a statistically significant response was observed for any of the following endpoints: amphipod (*Hyalella azteca*) survival and growth, mayfly (*Hexagenia limbata*) survival, midge (*Chironomus tentans* or *Chironomus riparius*) survival and growth, and daphnid (*Ceriodaphnia dubia*) survival.

Matching, synoptically collected sediment chemistry and toxicity data were also compiled from several studies conducted in marine and estuarine locations, including Biscayne Bay, Florida, USA [18; E.R. Long, published data]; Environmental Monitoring and Assessment Program Virginian Province [18]; Hudson-Raritan Estuary, New York, USA [28]; Hudson-Raritan Estuary/Newark Bay, New York/New Jersey, USA [18; unpublished sediment chemistry and toxicity data]; Long Island Sound, USA [18]; Naragansett Bay, Rhode Island, USA [29]; Puget Sound, Washington, USA [30]; San Diego Bay, California, USA [18]; San Francisco Bay, California, USA [31]; San Pedro Bay, California, USA [18]; South Carolina and Georgia, USA [18]; and Tampa Bay, Florida, USA [18]

In total, information on the chemical composition and toxicity of 1,151 sediment samples was obtained. In these studies, sediment toxicity was assessed using the results of toxicity tests conducted on the amphipods *Ampelisca abdita* and *Rhepoxynius abronius*.

In this study, predictive ability was defined as the ability of the SECs to correctly classify sediment samples as being toxic or nontoxic. Predictive ability was calculated as the ratio of the number of samples that were correctly classified as toxic or nontoxic and the number of samples that were predicted to be toxic or nontoxic using the various SECs (predictive ability was expressed as a percentage). In this evaluation, samples with PCB concentrations less than the TEC were predicted to be nontoxic, whereas those with concentrations greater than the MEC or the EEC were predicted to be toxic. Samples with PCB concentrations between the TEC and MEC were neither predicted to be toxic nor to be nontoxic (the SECs are not intended to provide guidance within this range of concentrations).

Criteria for evaluating the predictive ability of the SECs were adapted from those of Long et al. [18]. Specifically, the TEC was considered to provide a reliable basis for assessing sediment quality if more than 75% of the sediment samples were correctly predicted to be nontoxic. Similarly, the EEC was considered to be reliable if more than 75% of the sediment samples were correctly predicted to be toxic. Therefore, the target level for both false-positive classifications (i.e., samples incorrectly classified as being toxic) and false-negative classifications (i.e., samples incorrectly classified as being nontoxic) was 25% using the TEC and EEC. Because the MECs are intended to identify contaminant concentrations greater than that at which adverse effects frequently occur, the MEC was considered to be reliable if the incidence of toxicity was more than 50% at PCB concentrations greater than this level. The degree of concordance between PCB concentrations and sediment toxicity was evaluated by determining the incidence of toxicity within the four ranges of concentrations defined by the three SECs (i.e., <TEC, TEC–MEC, MEC–EEC, and >EEC).

Data from spiked-sediment toxicity tests and EqP models provide specific information for identifying the concentrations of sediment-associated PCBs that are likely to cause toxicity to sediment-dwelling organisms, either when the PCBs occur alone or in simple mixtures with other contaminants. To determine if the empirically derived SECs identified the concentrations of PCBs that are likely to cause adverse effects on sediment-dwelling organisms (as opposed to merely being associated with such effects), the TEC, MEC, and EEC were compared with the results of dose-response studies and EqP models for PCBs. First, the results of spiked-sediment toxicity tests and related toxicological data were reviewed to identify chronic toxicity thresholds for PCBs. Likewise, the results from EqP models were used to identify the concentrations of PCBs above which adverse effects are likely to occur on sensitive, sediment-dwelling organisms (i.e., during longer-term exposures). The consensus-based SECs were considered to be comparable to the chronic effects thresholds if they agreed within a factor of three (i.e., ± a factor of 3, as recommended by Lorenzato et al. [32]).

Description and classification of existing SQG for PCBs

Both empirical and theoretical approaches were considered to support the derivation and evaluation of consensus-based SECs for PCBs, various PCB mixtures, and/or individual PCB congeners, including the screening level concentration approach, effects range approach, effects level approach, AET approach, and EqP approach. Each of these approaches is described in the literature, but some confusion remains concerning how the SQGs are derived and what they actually mean. Therefore, a brief description of each approach is offered to provide sufficient background information to understand the underlying SQGs that were used to derive the consensus-based SECs. Each of the published SQGs was classified as TEC, MEC, or EEC, based on the descriptions of their narrative intents.

Screening level concentration approach

The screening level concentration is a biological effects-based approach that is applicable to the development of SQGs for the protection of benthic organisms. This approach uses matching biological and chemistry data collected in field surveys to calculate a screening level concentration [3], which is an estimate of the highest concentration of a contaminant that can be tolerated by a predefined proportion of benthic infaunal species.

The screening level concentration is determined through use of a database containing information on the concentrations of specific contaminants in sediments and on the co-occurrence of benthic organisms in those same sediments. For each benthic organism for which adequate data are available, a species screening level concentration is calculated. The species screening level concentration is determined by plotting the frequency distribution of the contaminant concentrations over all the sites at which the species occurs; information from at least 10 sites is required to calculate a species screening level concentration. The 90th percentile of this distribution is considered to be the screening level concentration for the species being investigated. Species screening level concentrations for all the species for which adequate data are available are then compiled as a frequency distribution to determine the concentration that can be tolerated by a specific proportion of the species. For example, the fifth percentile of the distribution would provide a screening level concentration that should be tolerated by 95% of the species. This concentration is termed the screening level concentration of the contaminant.

Several jurisdictions have used screening level concentrations to derive numeric SQGs. In the St. Lawrence River, two SQGs were developed for five groups of PCBs using the screening level concentration approach, including a minimal effect threshold and a toxic effect threshold [7]. The minimal effect threshold was calculated as the 15th percentile of the species screening level concentrations, whereas the toxic effect threshold was calculated as the 90th percentile of the species screening level concentration distribution for each substance. Therefore, the minimal effect threshold and toxic effect threshold are considered to provide protection for 85% and 10%, respectively, of the species represented in the database. Similarly, Environment Ontario has develop a lowest effect level and severe effect level for each of five groups of PCBs by using this approach [8]. Neff et al. [3] also developed a screening level concentration for tPCBs primarily by using data from the Great Lakes.

For calculating consensus-based SECs, the minimal effect threshold, lowest effect level, and screening level concentration were considered to represent TECs, because they are expected to protect 85 to 90% of sediment-dwelling organisms.

The toxic effect threshold and severe effect level were considered to represent EECs, because adverse effects are expected on 90% of sediment-dwelling species at greater than such concentrations.

Effects range approach

The effects range approach to derivation of SQGs was developed to provide informal tools for assessing the potential for various contaminants, tested in the National Oceanic and Atmospheric Administration's National Status and Trends Program, to be associated with adverse effects on sediment-dwelling organisms [4]. First, a database was compiled that contained information on the effects of sediment-associated contaminants, including data from spiked-sediment toxicity tests, matching sediment chemistry and biological effects data from field studies in the United States, and SQGs that were derived using various approaches. All the information in the database was weighted equally, regardless of the method that was used to develop it.

Candidate data sets from field studies were evaluated to determine their applicability for incorporation into the database [5]. This evaluation was designed to determine the overall applicability of the data set, the methods used, the endpoints measured, and the degree of concordance between the chemical and the biological data. Data that met the evaluation criteria were incorporated into the database.

The database that was compiled included several types of information from each study. Individual entries consisted of the concentration of the contaminant, the location of the study, the species tested and the endpoint measured, and an indication of any concordance between the observed effect and the concentrations of a specific chemical (i.e., no effect, no or small gradient, no concordance, or a hit, which indicated that an effect was measured in association with elevated sediment chemistry). Data from nontoxic or unaffected samples were assumed to represent background conditions. Data that showed no concordance between chemical and biological variables were included in the database but were not used to calculate the SQGs. Data for which a biological effect was observed in association with elevated chemical concentrations (i.e., hits) were sorted in ascending order of concentration, and the 10th- and 50th-percentile concentrations for each compound were determined. The effects range–low (i.e., 10th-percentile value) was considered to represent a lower threshold value, below which adverse effects on sensitive life stages and/or species occurred infrequently. The effects range–median (i.e., 50th-percentile value) was considered to represent a second threshold value, above which adverse effects were frequently observed. These two parameters were then used as informal SQGs [4,14]. The U.S. Environmental Protection Agency (EPA) [16] used a similar approach to derive effects range–lows (15th-percentile of the effects data set) and effects range–medians (50th-percentile of the effects data set) for assessing sediments from various freshwater locations. Similarly, MacDonald [15] applied the effects range approach to regionally collected field data to derive site-specific SECs for PCBs and DDTs in the Southern California Bight, USA.

For calculating consensus-based SECs, the effects range–low values were considered to represent TECs, because adverse effects are expected to be observed only infrequently at concentrations less than such SQGs. In contrast, the effects range–median [4,14] and SEC [15] values were considered to represent MECs, because adverse effects are likely to be observed at concentrations greater than such values.

Effects level approach

The effects level approach is closely related to the effects range approach described earlier. However, the effects level approach is supported by an expanded version of the database that was used to derive the effects levels [4]. This expanded database contains matching sediment chemistry and biological effects data from spiked-sediment toxicity tests and from field studies conducted throughout North America, including both effects and no-effects data. The expanded database also contains SQGs derived using various approaches. The information contained in the expanded database was evaluated and classified in the same manner the original National Status and Trends Program database was compiled.

In the effects-level approach, the underlying information in the database was used to derive two types of SQGs, including threshold effect levels and probable effect levels. The threshold effect level, which is calculated as the geometric mean of the 15th percentile of the effects data set and the 50th percentile of the no-effects data set, represents the chemical concentration below which adverse effects occurred only infrequently. The probable effect level represents a second threshold value, or the concentration above which adverse effects were frequently observed. The probable effect level is calculated as the geometric mean of the 50th percentile of the effects data set and the 85th percentile of the no-effects data set. These arithmetic procedures have been applied to the expanded database to derive numeric SQGs (i.e., threshold effect levels and probable effect levels) for Florida, USA, coastal waters [5]; U.S. freshwater systems [9]; and Canadian freshwater and marine systems [10].

Because adverse effects are expected to be observed only infrequently at concentrations below the threshold effect levels, they were considered to represent TECs for calculating consensus-based SECs. Similarly, the probable effect levels were considered to represent MECs, because adverse effects are likely to be observed at concentrations above such values.

AET approach

The AET approach to the development of SQGs was developed for use in the Puget Sound area of Washington state [33]. The AET approach is based on empirically defined relationships between measured concentrations of a contaminant in sediments and observed biological effects. This approach is intended to define the concentration of a contaminant in sediment above which significant ($p \leq 0.05$) biological effects are always observed. These biological effects include, but are not limited to, toxicity to benthic and/or water-column species (as measured using sediment toxicity tests), changes in the abundance of various benthic species, and changes in benthic community structure. In Puget Sound, Washington, USA, for example, four AET values have been generated, including AETs for Microtox® (Azur, Carlsbad, CA, USA), oyster larvae, benthic community, and amphipods. The AET values are based on dry wt-normalized contaminant concentrations for metals and either dry wt- or total organic carbon–normalized concentrations for organic substances [6,34]. The state of Washington, USA, has used the various AET values to establish sediment-quality standards and minimum clean-up levels for contaminants of concern in the state.

Recently, Cubbage et al. [11] refined this approach to support the development of probable AETs using matching sediment chemistry and toxicity data for freshwater sediments from the state of Washington, USA. Ingersoll et al. [9] and the U.S. EPA [16] used a similar approach to develop freshwater AETs (termed no-effect concentrations in that study) using data from various freshwater locations.

Classification of AET values is challenging, because the relative sensitivity of each endpoint varies for different chemicals. In this study, AET-type values for tPCBs were classified into three categories to facilitate derivation of consensus-based SECs. The AET values for the most sensitive endpoints (identified here as low-range AETs), including the freshwater AET for Microtox, the California AET for bivalve embryos, and the Puget Sound AET for Microtox, were classified as being TECs, because adverse effects are not expected at concentrations less than these values. The AETs for the endpoints that exhibited intermediate sensitivities (identified here as mid-range AETs) were considered to represent MECs, because adverse effects are likely to be observed for most of the endpoints measured at concentrations greater than such values; these SQGs included the freshwater no-effect concentrations , freshwater probable AETs for amphipods, California AETs for benthic community, and the Puget Sound AETs for oysters and benthic community. The AETs for the least sensitive endpoints (identified here as high-range AETs), including the freshwater AETs for amphipods, the California AETs for amphipods, and the Puget Sound AETs for amphipods, were considered to represent EECs, because adverse effects on all the endpoints measured are expected at concentrations above such values (i.e., high-range AETs are greater than all the other AETs; therefore, adverse effects can be expected on all the endpoints for which AETs were derived).

EqP approach

The water-sediment EqP approach has been one of the most studied and evaluated techniques for developing SQGs for nonpolar organic chemicals and metals [2,35–39]. This approach is based on the premise that the distribution of contaminants among different compartments in the sediment matrix (i.e., sediment solids and interstitial water) is predictable based on their physicochemical properties, assuming that continuous-equilibrium exchange between sediment and interstitial water occurs. This approach has been supported by the results of spiked-sediment toxicity tests, which indicate positive correlations between the biological effects observed and the concentrations of contaminants measured in the interstitial water [2,39,40].

In the EqP approach, water quality criteria developed for the protection of freshwater or marine organisms are used to support the SQG derivation process. As such, water quality criteria formulated to protect the water-column species are assumed to be applicable to benthic organisms [2]. The SQGs are calculated using the appropriate water quality criteria, usually the final chronic values or equivalent criteria [41], in conjunction with the sediment/water partition coefficients for the specific contaminants. The final chronic value is derived from the species mean chronic values that have been calculated using published toxicity data, and it is intended to protect 95% of aquatic species. The calculation procedure for nonionic organic contaminants is

$$SQG = K_p \cdot FCV$$

where SQG is the sediment-quality guideline (μg/kg), K_p is the partition coefficient for the chemical (L/kg), and FCV is the final chronic value (μg/L).

The K_p is a function of the partition coefficient for sediment organic carbon (K_{oc}) of the substance under consideration and the amount of organic carbon in the sediment under investigation (f_{oc}), where $K_{p_i} = K_{oc} \cdot f_{oc}$ [2]. The K_{oc} for nonionic substances can be calculated from its octanol-water partition coefficient (K_{ow}) [2]. For PCBs, the K_{ow} values that have been measured for individual PCB congeners vary over several orders of magnitude. Therefore, derivation of an SQG for total PCBs using this approach necessitates selection of a K_{ow} that is representative of the compounds within this class (i.e., a K_{ow} for Aroclor® 1254 [Monsanto Chemical Company, Sauget, IL, USA], which is a mixture of many PCB congeners).

The EqP approach provides a theoretical basis for identifying chronic effects thresholds for PCBs when they occur alone in sediments. The EqP-based SQG were not used to derive consensus-based SECs. Instead, EqP-based SQGs were used to evaluate consensus-based SECs in terms of their ability to identify PCB concentrations above which PCBs would cause, or substantially contribute to, sediment toxicity. Two sets of EqP-based SQGs were identified for tPCBs, including the freshwater and marine SQGs for New York state, USA [42], and the more generally applicable SQG derived by Bolton et al. [36].

RESULTS

Derivation of consensus-based SECs

Existing SQGs for freshwater sediments that satisfied all the selection criteria are presented in Table 1. Most of the freshwater SQGs for tPCBs were comparable within a factor of three. Of the eight SQGs considered to represent TECs, five were within a factor of three of each other. Similarly, five of the six MEC-type SQGs were within a factor of three of each other, and two of the three EEC-type SQGs fell within a factor of three of each another.

Existing marine SQGs for PCBs are presented in Table 2. Examination of the SQGs that were compiled indicates that the comparability of the marine SQGs for tPCBs was somewhat lower than that for the freshwater SQGs. For example, three of the five TEC-type SQGs fell within a factor of three of each another. The MEC-type SQGs fell within two clusters, each of which had three comparable SQGs. The two EEC-type SQGs varied by slightly more than a factor of three.

Examination of the consensus-based SECs for tPCBs indicated that the freshwater SECs were similar to the marine SECs. In other words, the respective TEC, MEC, and EEC values for freshwater and saltwater were not statistically different from each another based on the results of modified Student's t tests ($p < 0.05$). Therefore, the freshwater, estuarine, and marine SECs were combined to facilitate the determination of consensus-based SECs that apply more generally to various types of waterbodies (Table 3). This decision was supported by toxicological data indicating that the range of acutely lethal or effective concentrations of PCBs for saltwater species (1.0–16,000 μg/L [43]) fully encompasses the range reported for freshwater species (2.0–2400 μg/L [44]). Similarly, the range of species mean acute values for saltwater crustaceans (10.5–12.5 μg/L) falls within the reported range reported for freshwater crustaceans (10–46 μg/L [44]). That the lower end of the effects range is similar for saltwater and freshwater organisms, combined with the high degree of overlap of the effects range, suggests there are no systematic dif-

Table 1. Consensus-based sediment effect concentrations of polychlorinated biphenyls (PCBs) for freshwater ecosystems[a]

Category of SEC	Total PCBs (mg/kg dry wt)	Aroclor 1016[b] (mg/kg dry wt)	Aroclor 1248[b] (mg/kg dry wt)	Aroclor 1254[b] (mg/kg dry wt)	Aroclor 1260[b] (mg/kg dry wt)	Reference
Threshold effect concentrations						
SLC	0.003					[3]
LAET (Microtox)	0.021					[11]
TEL-HA28	0.032					[9]
TEL	0.034					[10]
ERL	0.050					[4]
ERL-HA28	0.050					[9]
LEL	0.070	0.007	0.030	0.060	0.005	[8]
MET	0.200	0.100	0.050	0.060	0.005	[7]
Consensus-based TECs	0.035	NA	NA	NA	NA	
Standard deviation	0.061	NA	NA	NA	NA	
Midrange effect concentrations						
NEC	0.190					[9]
PEL-HA28	0.240					[9]
PEL	0.277					[10]
ERM	0.400					[4]
PAET (amphipod)	0.450					[11]
ERM-HA28	0.730					[9]
Consensus-based MECs	0.34					
Standard deviation	0.20					
Extreme effect concentrations						
HAET (amphipod)	0.820					[11]
TET	1.000	0.400	0.600	0.300	0.200	[7]
SEL	5.300	0.530	1.500	0.340	0.240	[8]
Consensus-based EECs	1.6	NA	NA	NA	NA	
Standard deviation	2.5	NA	NA	NA	NA	

[a] Dry wt = dry weight; EEC = extreme effect concentration; ERL = effects range (low); ERM = effects range (median); NEC = no-effect concentration; PAET = probable-apparent-effects threshold; PEL = probable effect level; HAET = highest-apparent-effects threshhold; HA28 = *Hyalella azteca* 28-d test; LAET = lowest-apparent-effects threshold; LEL = lowest effect level; MEC = moderate effect concentration; MET = moderate effect threshold; NA = not applicable; SEC = sediment effect concentration; SEL = severe effect level; SLC = screening-level concentration; TEC = threshold effect concentration; TEL = threshold effect level; TET = toxic effect threshold.

[b] Monsanto Chemical Company, Sauget, Illinois, USA.

Table 2. Consensus-based sediment effect concentrations of polychlorinated biphenyls (PCBs) for marine and estuarine ecosystems[a]

Category of SEC	Total PCBs (mg/kg dry wt)	Aroclor 1016[b] (mg/kg dry wt)	Aroclor 1248[b] (mg/kg dry wt)	Aroclor 1254[b] (mg/kg dry wt)	Aroclor 1260[b] (mg/kg dry wt)	Reference
Threshold effect concentrations						
TEL	0.022					[5]
ERL	0.023					[14]
SLC	0.043					[12]
LAET-C (bivalve)	0.088					[13]
LAET-PS (Microtox)	0.130					[13]
Consensus-based TECs	0.048					
Standard deviation	0.047					
Midrange effect concentrations						
ERM	0.180					[14]
PEL	0.189					[5]
MAET-C (benthic)	0.360					[13]
SEC	0.835					[15]
MAET-PS (benthic)	1.000					[13]
MAET-PS (oyster)	1.100			0.400		[13]
Consensus-based MECs	0.47					
Standard deviation	0.42					
Extreme effect concentrations						
HAET-C (amphipod)	0.960					[13]
HAET-PS (amphipod)	3.100					[13]
Consensus-based EECs	1.7					
Standard deviation	1.5					

[a] C = California; dry wt = dry weight; EEC = extreme effect concentration; ERL = effects range low; ERM = effects range median; HAET = highest-apparent-effects threshold; LAET = lowest-apparent-effects threshold; MAET = moderate-apparent-effects threshold; MEC = moderate effect concentration; PEL = probable effect level; PS = Puget Sound; SEC = sediment effect concentration; SLC = screening-level concentration; TEC = threshold effect concentration; TEL = threshold effect level

Table 3. Consensus-based sediment effect concentrations of polychlorinated biphenyls (PCBs)[a]

Category of SEC	Total PCBs (mg/kg dry wt)	Reference
Threshold effect concentrations		
SLC	0.003	[3]
LAET (Microtox)	0.021	[11]
TEL	0.022	[5]
ERL	0.023	[14]
TEL-HA28	0.032	[9]
TEL	0.034	[10]
SLC	0.043	[12]
ERL	0.050	[4]
ERL-HA28	0.050	[9]
LEL	0.070	[8]
LAET-C (bivalve)	0.088	[13]
LAET-PS (Microtox)	0.130	[13]
MET	0.200	[7]
Consensus-based TECs	0.040	
Standard deviation	0.054	
Midrange effect concentrations		
ERM	0.180	[14]
PEL	0.189	[5]
NEC	0.190	[9]
PEL-HA28	0.240	[9]
PEL	0.277	[10]
MAET-C (benthic)	0.360	[13]
ERM	0.400	[4]
PAET (amphipod)	0.450	[11]
ERM-HA28	0.730	[9]
SEC	0.835	[15]
MAET-PS (benthic)	1.000	[13]
MAET-PS (oyster)	1.100	[13]
Consensus-based MECs	0.40	
Standard deviation	0.33	
Extreme effect concentrations		
HAET-C (amphipod)	0.820	[13]
HAET (amphipod)	0.960	[11]
TET	1.000	[7]
HAET-PS (amphipod)	3.100	[13]
SEL	5.300	[8]
Consensus-based EECs	1.7	
Standard deviation	2.0	

[a] C = California; dry wt = dry weight; EEC = extreme effect concentration; ERL = effects range low; ERM = effects range median; HA28 = *Hyalella azteca* 28-d test; HAET = highest-apparent-effects threshold; LAET = lowest-apparent-effects threshold; LEL = lowest effect level; MAET = moderate-apparent-effects threshold; MEC = moderate effect concentration; MET = moderate effect threshold; NEC = no-effect concentration; PAET = probable-apparent-effects threshold; PEL = probable effect level; PS = Puget Sound; SEC = sediment effect concentration; SEL = severe effect level; SLC = screening-level concentration; TEC = threshold effect concentration; TEL = threshold effect level; TET = toxic effect threshold.

ferences in the sensitivities of freshwater and saltwater species to PCBs. Therefore, the SQGs for both media types were merged and used to calculate the consensus-based SECs for tPCBs presented in Table 3.

Evaluation of consensus-based SECs

Consensus-based SECs that were derived in this study were evaluated to determine if they provided a reliable basis for identifying the concentrations of PCBs that are likely to substantially contribute to or cause sediment toxicity. This evaluation consisted of four main elements: determination of the predictive ability of the SECs; assessment of the degree of concordance between PCB concentrations and the incidence of sediment toxicity; determination of the level of agreement with the results of spiked-sediment toxicity tests; and assessment of the level of agreement with the EqP-based SQGs.

Predictive ability of consensus-based SECs. Matching sediment chemistry and toxicity data (195 sediment samples in total) were used to evaluate the predictive ability of the consensus-based SECs in freshwater sediments. Within this independent data set, 76 of the 90 samples with tPCB concentrations less than the TEC (0.04 mg/kg dry wt) were nontoxic (predictive ability, 84%). The incidence of adverse biological effects was also low (3 of 42 samples, or 7%) when tPCB concentrations were greater than the TEC but less than the MEC (0.40 mg/kg dry wt). The incidence of toxicity to freshwater biota was much higher (43 of 63 samples, or 68.3%) at tPCB concentrations greater than the MEC. The predictive ability of the EEC (1.7 mg/kg dry wt) was even higher: 33 of the 40 samples with tPCB concentrations in excess of this value were toxic (predictive ability, 83%). The overall incidence of toxicity in the entire freshwater database was 31%.

The predictive ability of the consensus-based SECs in marine and estuarine sediments is similar to that in freshwater sediments (Tables 4 and 5). Of the 599 marine sediment samples with tPCB concentrations less than the TEC (0.040 mg/kg dry wt), 527 were nontoxic based on results of the acute amphipod toxicity tests (predictive ability, 88%). By comparison, 128 of the 391 sediment samples (33%) with tPCB concentrations greater than the TEC but less than the MEC were toxic. Most of the sediment samples with tPCB concentrations greater than the MEC (0.40 mg/kg dry wt) were toxic (90 of 161 sediment samples; predictive ability, 56%). The incidence of toxicity was higher when tPCB concentrations in sediment samples exceeded the EEC (24 of 28 samples, or 86%). Overall, the incidence of toxicity in all studies used to evaluate predictive ability in marine and estuarine sediments was 25%; in other words, 290 of the 1,151 samples evaluated in these studies were significantly toxic to amphipods.

Table 4. Evaluation of the predictive ability of the consensus-based sediment effect concentrations (SECs) in freshwater sediments[a]

Consensus-based SEC	Range of tPCB concentrations defined by SEC	No. samples within range	No. toxic samples within range	Incidence of toxicity (%)	Predictive ability of the SEC (%)	Average survival (%)
<TEC	0.00–0.04 mg/kg dry wt	90	14	15.6	84.4	83.8
TEC–MEC	>0.04–0.40 mg/kg dry wt	42	3	7.1	NA	81.9
>MEC–EEC	>0.40–1.7 mg/kg dry wt	23	10	43.5	NA	71.7
>MEC	>0.4 mg/kg dry wt	63	43	68.3	68.3	70.4
>EEC	>1.7 mg/kg dry wt	40	33	82.5	82.5	69.7
Overall		195	60	30.8	NA	79.0

[a] Dry wt = dry weight; EEC = extreme effect concentration; MEC = moderate effect concentration; NA = not applicable; TEC = threshold effect concentration; tPCB = total polychlorinated biphenyl.

Table 5. Evaluation of the predictive ability of the consensus-based sediment effect concentrations (SECs) in marine and estuarine sediments.[a]

Consensus-based SEC	Range of tPCB concentrations defined by SEC	No. samples within range	No. toxic samples within range	Incidence of toxicity (%)	Predictive ability of the SEC (%)	Average survival (%)
<TEC	0.00–0.04 mg/kg dry wt	599	72	12.0	88.0	89.5
TEC–MEC	>0.04–0.40 mg/kg dry wt	391	128	32.7	NA	75.6
>MEC–EEC	>0.40–1.7 mg/kg dry wt	133	66	49.6	NA	65.8
>MEC	>0.4 mg/kg dry wt	161	90	55.9	55.9	58.3
>EEC	>1.7 mg/kg dry wt	28	24	85.7	85.7	37.7
Overall		1151	290	25.2	NA	80.8

[a] Dry wt = dry weight; EEC = extreme effect concentration; MEC = moderate effect concentration; NA = not applicable; TEC = threshold effect concentration; tPCB = total polychlorinated biphenyl.

Concordance between PCB concentrations and the incidence of toxicity. The matching sediment chemistry and biologic effects data assembled to support evaluations of the predictive ability of SECs was also used to determine relationships between contaminant concentrations and sediment toxicity. Specifically, the three consensus-based SECs (Table 3) were used to delineate four ranges of tPCB concentrations: <TEC, TEC–MEC, MEC–EEC, and >EEC. The incidence of toxicity within these ranges generally increases with increasing concentrations of tPCBs in freshwater sediments (Table 4). This evaluation also demonstrates that the incidence of toxicity in marine and estuarine sediments increases consistently and markedly with increasing tPCB concentrations (Table 5). This high degree of concordance between tPCB concentrations and sediment toxicity indicates that PCBs are strongly associated with toxicity at concentrations greater than the MEC and the EEC (Fig. 1).

Agreement with spiked-sediment toxicity tests. Dose-response data for sediment-dwelling organisms provide a basis for identifying the concentrations of sediment-associated contaminants that would be sufficient to cause sediment toxicity. No information was located on the toxicity of tPCBs per se, but data from five spiked-sediment toxicity tests using formulated mixtures of PCBs provided relevant information for evaluating the consensus-based SECs [45–49]. The results of these studies indicate that PCBs are acutely toxic to sediment-dwelling organisms at concentrations ranging from greater than 0.78 to 251 mg/kg dry wt. A median lethal concentration (LC50) of 8.8 mg/kg dry wt was reported for the amphipod *Rhepoxynius abronius,* when PCBs (Aroclor 1254) alone were tested [48]. The U.S. EPA [44] reported an acute-to-chronic

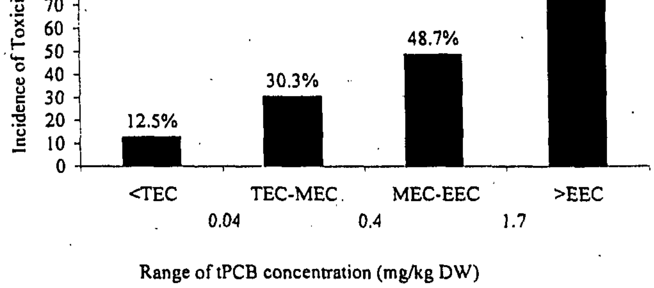

Fig. 1. Incidence of toxicity within the range of polychlorinated biphenyl concentrations defined by the sediment effects concentrations. DW = dry weight; EEC = extreme effect concentration; MEC = midrange effect concentration; TEC = threshold effect concentration; tPCB = total polychlorinated biphenyls.

ratio of 11 for the freshwater amphipod *Gammarus pseudolimnaeus* based on toxicity tests conducted with waterborne PCBs. This ratio is much lower than the acute-to-chronic ratios (27–58) that can be calculated from toxicity tests conducted on the copepod *Microarthridion littorale* [49]. Application of an empirically derived acute-to-chronic ratio for the freshwater amphipod to the 10-d LC50 for the marine amphipod suggests that PCBs, when they are present alone in sediments, are likely to cause chronic toxicity to amphipods at concentrations in the range of 0.8 mg/kg dry wt (i.e., 8.8 mg/kg dry wt ÷ 11 = 0.8 mg/kg dry wt).

Spiked-sediment toxicity tests conducted under controlled laboratory conditions can be used to determine lethal or effective concentrations of many chemical substances. However, such response thresholds could underestimate the ecological effects that occur in the field because of the presence of contaminant mixtures in sediments [1]. As such, sediments containing mixtures of contaminants could be more toxic than sediments containing PCBs alone.

To evaluate the possible interactive effects of PCBs with other contaminants, several investigators have conducted spiked-sediment toxicity tests with mixtures of contaminants. The results of these studies indicate that sediments tend to be more toxic when they contain mixtures of contaminants (i.e., PCBs and other substances). For example, Plesha et al. [47] reported acute toxicity to amphipods (*Rhepoxynius abronius*) in sediments containing several chlorinated hydrocarbons and Aroclor 1254; the concentration of PCBs in these sediments was 1 mg/kg dry wt. Similarly, sediments containing 2.1 mg/kg dry wt of both Aroclor 1254 and fluoranthene were acutely toxic to amphipods (*Rhepoxynius abronius*) [48]. These data indicate that PCB-contaminated sediments are more toxic when they also contain other commonly occurring contaminants (e.g., polycyclic aromatic hydrocarbons [PAHs] [48]). Considering the relationship between the acute LC50 for PCBs (8.8 mg/kg dry wt) and the concentration of PCBs in acutely toxic sediments containing both Aroclor 1254 and fluoranthene (2.1 mg/kg dry wt, giving a ratio of 4.2), PCBs likely contribute to sediment toxicity at concentrations less than the estimated chronic toxicity threshold of 0.8 mg/kg dry wt when they occur with other contaminants. The TEC (0.04 mg/kg dry wt) derived in this study is lower than the chronic effect threshold that was estimated from the spiked-sediment toxicity tests, whereas the MEC (0.40 mg/kg dry wt) and EEC (1.7 mg/kg dry wt) are comparable to such thresholds.

Agreement with EqP-based SQGs. The EqP approach provides a theoretical basis for identifying chronic toxicity thresholds for sediment-associated PCBs. Using this approach, the New York State Department of Environmental Conservation

[42] has developed chronic SQGs for PCBs to protect freshwater and saltwater benthic aquatic life. These guidelines indicate that thresholds for chronic toxicity in freshwater and saltwater sediments are 0.19 and 0.41 mg/kg dry wt at 1% organic carbon, respectively. An EqP-based SQG of 0.07 mg/kg dry wt at 1% organic carbon has also been derived to support the evaluation of sediment-quality conditions at freshwater and saltwater locations in the United States [36]. Together, these EqP-based SQGs suggest that chronic effects on sediment-dwelling organisms are likely to occur at tPCB concentrations in excess of 0.07 to 0.41 mg/kg dry wt. The lowest EqP-based SQG is comparable to the TEC derived in this report (0.04 mg/kg dry wt), whereas the other two EqP-based SQGs are comparable to the MEC (0.40 mg/kg dry wt). The EEC is higher than all the available EqP-based SQGs.

DISCUSSION

Evaluating the toxic effects of PCBs is complicated for several reasons. First, these compounds consist of 209 different congeners, each of which may have unique toxicological characteristics [50–52]. Second, much of the available dose-response data on the toxicity of sediment-associated PCBs from controlled laboratory studies have been generated on several formulated PCB mixtures, including Aroclor 1242 and Aroclor 1254. However, sediments at any particular site under investigation could contain more PCB congeners than would be represented by measurements of Aroclor 1242 or Aroclor 1254 concentrations alone (i.e., *mono-*, *di-*, and *hepta-*chlorobiphenyls may not be fully represented by these measurements). Therefore, field-collected sediments could be more or less toxic than would be indicated by, for example, Aroclor 1254 concentrations alone.

In field-collected sediments, PCBs always occur as complex mixtures of the individual congeners, commonly in association with other contaminants. Toxic effects on sediment-dwelling organisms likely result from the cumulative effects of these mixtures of contaminants. Therefore, SQGs for individual PCB congeners that are developed through experimental determination of toxicological effects (i.e., spiked-sediment bioassays) or with EqP models likely underestimate the ecological effects that occur in the field. Similarly, SQGs for individual PCB congeners that are developed using data from field studies could overestimate the effects that are actually caused by each congener if it occurred alone in sediments. Swartz [1] used the term mixture paradox to describe the dilemma associated with evaluating the toxic effects of contaminant mixtures (PAHs in that case). Swartz [1] resolved this dilemma by deriving consensus-based SECs for mixtures of PAHs (i.e., total PAHs). Applying similar logic to the assessment of PCB-contaminated sediments, it is reasonable to rely on SECs that can be applied to mixtures of PCBs (i.e., tPCBs), provided that such guidelines are reliable.

In this study, several types of information were used to determine the degree of confidence that can be placed in the consensus-based SECs for tPCBs. First, the available data from spiked-sediment toxicity tests demonstrate that PCBs are acutely toxic to sediment-dwelling organisms, which justifies derivation of effects-based SECs for this class of compounds. Second, consensus-based SECs that were derived independently for freshwater sediments and for marine sediments were similar (i.e., not statistically different from one another), which generates confidence that the underlying guideline values are broadly applicable. In addition, the incidence of toxicity generally increases with increasing concentrations of tPCBs in freshwater, estuarine, and marine sediments, which indicates that PCBs are strongly associated with sediment toxicity in freshwater, estuarine, and marine sediments. Importantly, the TEC, MEC, and EEC also provided accurate tools for predicting the presence or absence of toxicity in freshwater, estuarine, and marine sediments.

Results from comparisons of the consensus-based SECs with the empirically and theoretically derived chronic effects thresholds further increase the level of confidence that can be placed in the guidelines. Specifically, the MEC (0.4 mg/kg dry wt) and EEC (1.7 mg/kg dry wt) are both comparable to the chronic effects threshold (0.8 mg/kg dry wt) that was estimated from the results of toxicity tests conducted with PCB-spiked sediments using an empirically derived acute-to-chronic ratio of 11. The MEC and EEC are also likely to be higher than the chronic effects thresholds for PCBs in sediments containing mixtures of other contaminants. In addition, the TEC is comparable to the lowest chronic effects threshold that has been determined using the EqP approach. Furthermore, the MEC and EEC are comparable to, or higher than, all the chronic effects thresholds (0.07–0.41 mg/kg dry wt) that were determined using the EqP approach.

When considered individually, the results of these evaluations again increase the confidence that can be placed in the consensus-based SECs derived in this study. When considered together, however, they provide a weight of evidence for concluding that sediment-associated PCBs are likely to cause, or substantially contribute to, adverse biological effects at concentrations in excess of the MEC or the EEC. Furthermore, PCBs are unlikely to cause, or substantially contribute to, sediment toxicity at concentrations below the TEC.

It has been argued that SQGs can not be causal unless they are normalized to account for the factors that influence bioavailability [40]. However, Ingersoll et al. [9] showed that organic carbon normalization did not improve the performance of SQGs. More importantly, the consensus-based SECs were comparable both to the chronic toxicity thresholds that were derived from EqP models and to spiked-sediment toxicity tests. To the extent that such chronic toxicity thresholds are causally based, the consensus-based SECs also reflect the concentrations of PCBs that are likely to cause, or substantially contribute to, sediment toxicity. Therefore, use of dry wt normalization does not reduce the reliability of the SECs.

The consensus-based SECs reflect the toxicity of PCBs when they occur in mixtures with other contaminants. Therefore, these consensus-based SECs are likely to be directly relevant for assessing freshwater, estuarine, and marine sediments that are influenced by multiple sources of contaminants. Results from the evaluation of predictive ability confirm the applicability of the SECs for assessing the quality of such PCB-contaminated sediments.

Overall, results of the various evaluations demonstrate that the consensus-based SECs provide a unifying synthesis of existing SQGs, reflect causal rather than correlative effects, and account for the effects of contaminant mixtures [1]. As such, SECs can be used to identify hot spots regarding PCB contamination, to determine the potential for and spatial extent of injury to sediment-dwelling organisms, to evaluate the need for sediment remediation, and to support the development of monitoring programs to further assess the extent of PCB contamination and the effects of contaminated sediments on sediment-dwelling organisms. In these applications, the TEC should be used to identify sediments that are unlikely to be adversely affected by PCBs. In contrast, the MEC and EEC should be used to identify sediments that likely are toxic to sediment-dwelling organisms, at least in part because of the presence of PCBs. However, these SECs do not consider the potential for bioaccumulation of PCBs in aquatic organisms or the associated hazards to species that consume the aquatic organisms (i.e., wildlife and humans). Therefore, SECs should be used in conjunction with other tools, such as bioaccumulation assessments, tissue chemistry data, and tissue residue guidelines, to assess the potential environmental effects of PCBs.

Acknowledgement—The authors acknowledge several individuals who contributed to the production of this manuscript, including Michael Huguenin, Josh Habib, Corinne Severn, Jesse Brown, Diana Tao, Tadd Berger, and Mary Lou Haines. The authors also acknowledge Walter Berry, Scott Carr, and two anonymous reviewers for conducting thorough peer reviews of this manuscript. This paper was funded in part by a contract from the National Oceanic and Atmospheric Administration.

REFERENCES

1. Swartz RC. 1999. Consensus sediment quality guidelines for PAH mixtures. *Environ Toxicol Chem* 18:780–787.
2. Di Toro DM, et al. 1991. Technical basis for establishing sediment quality criteria for non-ionic organic chemicals using equilibrium partitioning. *Environ Toxicol Chem* 10:1541–1583.
3. Neff JM, Bean DJ, Cornaby BW, Vaga RM, Gulbransen TC, Scalon JA. 1986. Sediment quality criteria methodology validation: Calculation of screening level concentrations from field data. Final Report. U.S. Environmental Protection Agency, Washington, DC.
4. Long ER, Morgan LG. 1991. The potential for biological effects of sediment-sorbed contaminants tested in the national status and trends program. NOS OMA 52. NOAA Technical Memorandum. National Oceanic and Atmospheric Administration, Seattle, WA, USA.
5. MacDonald DD, Carr RS, Calder FD, Long ER, Ingersoll CG. 1996. Development and evaluation of sediment quality guidelines for Florida coastal waters. *Ecotoxicology* 5:253–278.
6. Barrick R, Becker S, Pastorok R, Brown L, Beller H. 1988. Sediment quality values refinement: 1988 update and evaluation of Puget Sound AET. EPA 68-01-4341. Final Report. U.S. Environmental Protection Agency, Seattle, WA.
7. Environment Canada and Ministere de l'Envionnement du Quebec. 1992. Interim criteria for quality assessment of St. Lawrence River sediment. Final Report. Environment Canada, Ottawa, ON.
8. Persaud D, Jaagumagi R, Hayton A. 1993. Guidelines for the protection and management of aquatic sediment quality in Ontario. Final Report. Ontario Ministry of Environment and Energy, Toronto, ON, Canada.
9. Ingersoll CG, et al. 1996. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the midge *Chironomus riparius*. *J Great Lakes Res* 22:602–623.
10. Smith SL, MacDonald DD, Keenleyside KA, Ingersoll CG, Field J. 1996. A preliminary evaluation of sediment quality assessment values for freshwater ecosystems. *J Great Lakes Res* 22:624–638.
11. Cubbage J, Batts D, Briedenbach S. 1997. Creation and analysis of freshwater sediment quality values in Washington State. Publication 97-323. Technical Report. Washington Department of Ecology, Olympia, WA, USA.
12. Neff JM, Word JQ, Gulbransen TC. 1987. Recalculation of screening level concentrations for nonpolar organic contaminants in marine sediments. Final Report. Washington Environmental Program Office, Washington, DC, USA.
13. Becker DS, Barrick RC, Read LB. 1990. Evaluation of the AET approach for assessing contamination of marine sediments in California. Report 90-3WQ. PTI Environmental Services, Bellevue, WA, USA.
14. Long ER, MacDonald DD, Smith SL, Calder FD. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environ Manage* 19:81–97.
15. MacDonald DD. 1997. Sediment injury in the Southern California bight: Review of the toxic effects of DDTs and PCBs in sediments. Technical Report. National Oceanic and Atmospheric Administration, Long Beach, CA, USA.
16. U.S. Environmental Protection Agency. 1996. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the midge *Chironomus riparius*. EPA 905-R96-008. Chicago, IL.
17. Sokal RR, Rohlf FJ. 1981. *Biometry*, 2nd ed. W.H. Freeman & Company, New York, NY, USA.
18. Long ER, Field LJ, MacDonald DD. 1998. Predicting toxicity in marine sediments with numerical sediment quality guidelines. *Environ Toxicol Chem* 17:714–727.
19. Hoke RA, Giesy JP, Zabik M, Unger M. 1993. Toxicity of sediments and sediment pore waters from the Grand Calumet River-Indiana Harbor, Indiana, area of concern. *Ecotoxicol Environ Saf* 26:86–112.
20. Giesy JP, Hoke RA, Zabik MJ. 1993. Sediment toxicity assessment in the Grand Calumet River system. Cooperative Agreement R005037-01. Department of Fisheries and Wildlife, East Lansing, MI, USA.
21. Call DJ, Balder MD, Brooke LT, Lozano SJ, Vaishnava DD. 1991. Sediment quality evaluation in the Lower Fox River and southern Green Bay of Lake Michigan. U.S. EPA CR-815232. Final Report. University of Wisconsin-Superior, Superior, WI, USA.
22. Schlekat C, McGee BL, Boward DM, Reinharz E, Velinsky DJ, Wade TL. 1994. Biological effects associated with sediment contamination in the Potomac and Anacostia rivers in the Washington, DC, area. *Estuaries* 17:334–344.
23. Wade TL, Velinsky DJ, Reinharz E, Schlekat CE. 1994. Tidal river sediments in the Washington, DC, area. II. Distribution and sources of organic contaminants. *Estuaries* 17:321–333.
24. Velinsky DJ, Wade TL, Schlekat CE, McGee BL, Presley BJ. 1994. Tidal river sediments in the Washington, DC, area. I. Distribution and sources of trace metals. *Estuaries* 17:305–320.
25. Dickson KL, et al. 1989. *A Water Quality and Ecological Survey on the Trinity River*, Vols 1 and 2. Institute of Applied Sciences, University of North Texas, Denton, TX, USA.
26. U.S. Environmental Protection Agency. 1997. An assessment of sediments from the Upper Mississippi River. Final Report. EPA 823-R-97-005. Columbia, MO.
27. Kemble NE, Dwyer FJ, Hardesty DK, Ingersoll CG, Johnson BT, MacDonald DD. 1999. Evaluation of the toxicity and bioaccumulation of contaminants in sediment samples from Waukegan Harbor, Illinois. U.S. Environmental Protection Agency, Chicago, IL.
28. Rice CA, Plesha PD, Casillas E, Misitano DA, Meador JP. 1995. Growth and survival of three marine invertebrate species in sediments from the Hudson-Raritan Estuary, New York. *Environ Toxicol Chem* 14:1931–1940.
29. Munns WR Jr, Mueller C, Cobb DJ, Gleason TR, Pesch GG, Johnston RK. 1991. Risk assessment pilot study, phase I. Naval Construction Battalion Center. Final Report. U.S. Environmental Protection Agency, Naragansett, RI.
30. Pastorok RA, Becker DS. 1990. Comparative sensitivity of sediment toxicity bioassays at three Superfund sites in Puget Sound. In Landis WG, van der Schalie WH, eds, *Aquatic Toxicology and Risk Assessment*, Vol 13. STP 1096. American Society for Testing and Materials, Philadelphia, PA, pp 123–139.
31. Chapman PM, Dexter RN, Long ER. 1987. Synoptic measures of sediment contamination, toxicity and infaunal community composition (the sediment quality triad) in San Francisco Bay. *Mar Ecol Prog Ser* 37:75–96.
32. Lorenzato SG, Gunther AJ, O'Connor JM. 1991. Summary of a workshop concerning sediment quality assessment and development of sediment quality objectives. 0-081-250-0. Final Report. California State Water Resources Control Board, Sacramento, CA, USA.
33. Tetra Tech. 1986. Development of sediment quality values for Puget Sound, Vol 1. TC3090-02. Final Report. U.S. Army Corps of Engineers, Seattle, WA.
34. Washington Department of Ecology. 1990. Sediment management standards. 204 WAC. Final Report. Olympia, WA, USA.
35. Pavlou SP, Weston DP. 1983. Initial evaluation of alternatives for development of sediment related criteria for toxic contaminants in marine waters (Puget Sound). Phase I: Development of con-

ceptual framework. EPA 68-01-6388. Final Report. U.S. Environmental Protection Agency, Washington, DC.

36. Bolton HS, Breteler RJ, Vigon BW, Scanlon JA, Clark SL. 1985. National perspective on sediment quality. EPA 68-01-6986. U.S. Environmental Protection Agency, Washington, DC.

37. Kadeg RD, Pavlou SP, Duxbury AS. 1986. Sediment criteria methodology validation: Elaboration of sediment normalization theory for non-polar hydrophobic organic chemicals. Final Report. U.S. Environmental Protection Agency, Washington, DC.

38. Pavlou SP. 1987. The use of equilibrium partitioning approach in determining safe levels of contaminants in marine sediments. In Dickson KL, Maki AW, Brugs WA, eds, *Fate and Effects of Sediment-Bound Chemicals in Aquatic Systems. Proceedings,* Sixth Pellston Workshop. Pergamon, Elmsford, NY, USA, pp 388–395.

39. Hansen DJ, et al. 1996. Predicting the toxicity of metal-contaminated field sediments using interstitial concentration of metals and acid-volatile sulfide normalizations. *Environ Toxicol Chem* 15:2080–2094.

40. Berry WJ, et al. 1996. Predicting the toxicity of metal-spiked laboratory sediments using acid-volatile sulfide and interstitial water normalizations. *Environ Toxicol Chem* 15:2067–2079.

41. U.S. Environmental Protection Agency. 1997. The incidence and severity of sediment contamination in surface waters of the United States, Vol 1: National sediment quality survey. EPA 823-R-97-006. Washington, DC.

42. New York State Department of Environmental Conservation. 1994. Technical guidance for screening contaminated sediments. Technical Report. Division of Fish and Wildlife, Division of Marine Resources, Albany, NY, USA.

43. Moore DR, Walker SL. 1991. Canadian water quality guidelines for polychlorinated biphenyls in coastal and estuarine waters. Scientific Series 186. Technical Report. Environment Canada, Ottawa, ON.

44. U.S. Environmental Protection Agency. 1980. Ambient water quality criteria for polychlorinated biphenyls. EPA 440/5-80-068. Office of Water Regulations and Standards, Washington, DC.

45. McLeese DW, Metcalfe CD. 1980. Toxicities of eight organochlorine compounds in sediment and seawater to *Crangon septemspinosa. Bull Environ Contam Toxicol* 25:921–928.

46. Polikarpov CG, Parsi P, Fowler SW. 1980. Chronic effects of a PCB (DP5) upon *Nereis diversicolor* in spiked Mediterranean sediments. *Rapp Comm Int Mer Medit* 28:167–168.

47. Plesha PD, Stein JE, Schiewe MH, McCain BB, Varanasi U. 1988. Toxicity of marine sediments supplemented with mixtures of selected chlorinated and aromatic hydrocarbons to the infaunal amphipod *Rhepoxynius abronius. Mar Environ Res* 25:85–97.

48. Swartz RC, Kemp PF, Schults DW, Lamberson JO. 1988. Effects of mixtures of sediment contaminants on the marine infaunal amphipod *Rhepoxynius abronius. Environ Toxicol Chem* 7:1013–1020.

49. DiPinto LM, Coull BC, Chandler GT. 1993. Lethal and sublethal effects of the sediment-associated PCB Aroclor® 1254 on a meiobenthic copepod. *Environ Toxicol Chem* 12:1909–1918.

50. Niimi AJ, Oliver BG. 1989. Assessment of relative toxicity of chlorinated dibenzo-*p*-dioxins, dibenzofurans, and biphenyls in Lake Ontario salmonids to mammalian systems using toxic equivalency factors (TEF). *Chemosphere* 18:1413–1423.

51. Safe S. 1990. Polychlorinated biphenyls (PCBs), dibenzo-*p*-dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: Environmental and mechanistic considerations which support the development of toxic equivalency factors (TEFs). *Crit Rev Toxicol* 21:51–88.

52. MacDonald DD, Ikonomou MG, Rantalaine AL, Rogers H, Sutherland D, Van Oostdam J. 1997. Contaminants in white sturgeon (*Acipenser transmontanus*) from the upper Fraser River, British Columbia, Canada. *Environ Toxicol Chem* 16:479–490.