Incidence of Adverse Biological Effects Within Ranges of Chemical Concentrations in Marine and Estuarine Sediments[1]

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ABSTRACT / Matching biological and chemical data were compiled from numerous modeling, laboratory, and field studies performed in marine and estuarine sediments. Using these data, two guideline values (an effects range-low and an effects range-median) were determined for nine trace metals, total PCBs, two pesticides, 13 polynuclear aromatic hydrocarbons (PAHs), and three classes of PAHs. The two values defined concentration ranges that were: (1) rarely, (2) occasionally, or (3) frequently associated with adverse effects. The values generally agreed within a factor of 3 or less with those developed with the same methods applied to other data and to those developed with other effects-based methods. The incidence of adverse effects was quantified within each of the three concentration ranges as the number of cases in which effects were observed divided by the total number of observations. The incidence of effects increased markedly with increasing concentrations of all of the individual PAHs, the three classes of PAHs, and most of the trace metals. Relatively poor relationships were observed between the incidence of effects and the concentrations of mercury, nickel, total PCB, total DDT and p,p'-DDE. Based upon this evaluation, the approach provided reliable guidelines for use in sediment quality assessments. This method is being used as a basis for developing National sediment quality guidelines for Canada and informal, sediment quality guidelines for Florida.

Chemical analyses indicate that coastal sediments in some areas of North America are contaminated (Bolton and others 1985, O'Connor 1991, US NOAA 1991, Wells and Rolston 1991, Goyette and Boyd 1989). However, data on the mixtures and concentrations of contaminants in sediments, alone, do not provide an effective basis for estimating the potential for adverse effects to living resources. Moreover, interpretive tools are needed to relate ambient sediment chemistry data to the potential for adverse biological effects. A variety of biological measures (including toxicity and/or bioaccumulation tests) can be performed to determine the biological significance of sediment-associated contaminants (Burton 1992). Furthermore, numerical, effects-based, sediment quality guidelines can be used as screening tools to evaluate sediment chemistry data and to identify and prioritize potential problem areas (Di Toro and others 1991, Persaud 1992, MacDonald 1993, Long and Morgan 1990, Smith and MacDonald 1992, US EPA 1989a, 1992a). In this respect, effects-based guidelines can be used to help identify those areas in which the potential for biological effects is greatest.

KEY WORDS: Sediment quality guidelines; Ecological risk assessment; Contaminants; Biological effects; Marine; Estuarine

[1]The methods and guidelines presented in this report do not necessarily represent the policy of the National Oceanic and Atmospheric Administration, Environment Canada, or Florida Department of Environmental Protection.

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A variety of biological effects-based approaches to the development of sediment quality guidelines have been reviewed by many investigators (US EPA 1989a, 1992a, Adams and others 1992, Chapman 1989, MacDonald and others 1992). These approaches can be grouped into three categories: equilibrium-partitioning modeling, laboratory bioassays, and field studies. Each approach has particular strengths and weaknesses and each defines guidelines in different ways. Thus far, there is no general agreement as to which approach will provide the most reliable, flexible, and credible guidelines for evaluating sediment quality. However, sediment quality guidelines derived from the combination of the results of multiple methods have been recommended for a broad range of applications (Adams and others 1992, US EPA 1989b, Lorenzato and others 1991).

Using data available from all the major approaches to the development of effects-based criteria, Long and Morgan (1990) prepared informal guidelines for use by the National Oceanic and Atmospheric Administration (NOAA). Subsequently, the data base with which these values were prepared was updated and expanded and the approach was refined (MacDonald 1993, Smith and MacDonald 1992). In both the NOAA (Long and Morgan 1990) and Florida (MacDonald 1993) studies, two guideline values were developed for each chemical. These values defined three ranges in chemical concentrations that were anticipated to be: (1) rarely, (2) occasionally, or (3) frequently associated with effects. The identification of ranges in chemical concentrations has been recommended in the development of sediment quality criteria (US EPA 1992b).

The objectives of the present study are: (1) to present updated guideline values based upon the expanded data base, (2) to quantify the percent incidence of adverse biological effects associated with the guidelines, and (3) to compare the guidelines with those developed with other data or methods. In this paper we determined the percent incidence of effects as a measure of the "accuracy" of the guidelines.

Methods

The methods used in this study have been described in detail (Long and Morgan 1990, MacDonald 1993, Smith and MacDonald 1992, Long 1992) and will be only summarized here. Sediment chemistry and biological effects data from numerous reports were assembled to support the derivation of the guidelines. The data base used by Long and Morgan (1990) was refined by excluding data from freshwater studies and including data from additional sites, biological test end points, and contaminants (MacDonald 1993, Smith and MacDonald 1992). Briefly, the approach involved three steps: (1) assemble, evaluate, and collate all available information in which measures of adverse biological effects and chemical concentrations in sediments were reported; (2) identify the ranges in chemical concentrations that were rarely, occasionally, or frequently associated with effects; and (3) determine the incidence of biological effects within each of the ranges in concentrations for each chemical as an estimate of guideline accuracy.

Development of a Biological Effects Database for Sediments

A biological effects database for sediments (BEDS) was developed to compile and integrate chemical and biological data from numerous studies conducted throughout North America. Nearly 350 publications were reviewed and screened for possible inclusion in the BEDS. Data from equilibrium-partitioning modeling, laboratory spiked-sediment bioassays, and field studies of sediment toxicity and benthic community composition were critically evaluated. Only matching, synoptically collected biological and chemical data from marine and estuarine studies were included in the database. Data were excluded if the methods were not clearly described. Data were excluded if sediments were frozen before toxicity tests were initiated or if toxicity of controls was higher than commonly acceptable. If there was less than a tenfold difference in the concentrations of all contaminants among sampling stations, all data from that particular field study were excluded. The tenfold criterion was selected to ensure that data were included in the BEDS only from studies in which significant contaminant gradients were reported. Furthermore, data were excluded if the chemical analytical procedures were inappropriate for determining total concentrations in bulk sediments; for example, trace metals data were excluded if strong acid digestions were not used. The majority of the data sets that were excluded were those in which either no biological data or no chemical data were reported. A total of 89 reports met all the screening criteria and were included in the BEDS. The screening criteria and their use were described previously (MacDonald 1993, Smith and MacDonald 1992). The potential limitations of using data "encountered" from many different studies have been described (Long 1992).

The data entered into the BEDS were expressed on a dry weight basis. Only a minority of the reports included measures of factors that are thought to influ-

ence bioavailability (e.g., grain size, total organic carbon, acid-volatile sulfides). Sediment quality guidelines derived from the equilibrium-partitioning approach (US EPA 1988) were converted from units of organic carbon to units of dry weight, assuming a total organic carbon (TOC) concentration of 1.0%. These conversions were based upon a TOC concentration of 1.0% since the overall mean TOC concentration in the BEDS was 1.2%. Data from spiked-sediment bioassays were incorporated directly into the BEDS.

Guideline values derived using the apparent effects threshold (AET approach, Barrick and others 1988) and national screening level concentration (SLC approach, Neff and others 1986) were entered into the BEDS as reported. AET and SLC values represent large amounts of data compiled from multiple surveys. Therefore, extremely high and extremely low concentrations in some parts of study areas used to produce these values may be ameliorated by highs and lows in other regions, resulting in intermediate concentrations. Raw data from other individual field surveys that passed the initial screening steps were evaluated in "co-occurrence analyses" with either of two methods (Long 1992). If the statistical significance of the data was reported, then the mean chemical concentrations in the statistical groups (i.e., toxic and nontoxic) were compared. If no such statistical evaluations were reported, the frequency distributions of the biological data were examined, and mean concentrations in subjectively determined groups of samples were compared (e.g., most toxic versus least toxic). The extreme high and low concentrations reported in individual studies, generally performed over relatively small spatial scales, were not masked by merging data from other studies.

To maximize the broad applicability of the guidelines, a wide variety of measures of adverse biological effects was included in the BEDS. The kinds of adverse effects included: (1) measures of altered benthic communities (depressed species richness or total abundance), significantly or relatively elevated sediment toxicity, or histopathological disorders in demersal fish observed in field studies; (2) EC_{50} or LC_{50} concentrations determined in laboratory bioassays of sediments spiked with single compounds or elements; and (3) toxicity predicted by equilibrium-partitioning models. All of the measures of effects were treated as if equivalent. However, by screening prospective data sets and including only those biological data that were in concordance with chemical gradients, the prevalence of data from relatively insensitive measures of effects was minimized.

Each entry was assigned an "effects/no-effects" descriptor. An entry was assigned an "effects" descriptor (identified with an asterisk in the data tables) if: (1) an adverse biological effect, such as acute toxicity, was reported; and (2) concordance was apparent between the observed biological response and the measured chemical concentration.

The documentation supporting each BEDS record included the citation, the type of test or biological effect observed or predicted, the approach that was used, the study area, the test duration (if applicable and reported), the species tested or the benthic community considered, the total organic carbon (TOC) and acid-volatile sulfide (AVS) concentrations (if reported), and the chemical concentration.

In our co-occurrence analyses of field-collected data entered into BEDS, an effects descriptor was assigned to data entries in which adverse biological effects were observed in association with at least a twofold elevation in the chemical concentration above reference concentrations. Either "no gradient," "small gradient," or "no concordance" descriptors were assigned when no differences between stations were reported in the concentration of the chemical of concern, when mean chemical concentrations differed by less than a factor of two between the groups of samples, or when there was no concordance between the severity of the effect and the chemical concentration, respectively. In these cases, we assumed that other factors (whether measured or not) were more important in the etiology of the observed effect than the concentration of the contaminant considered. Finally, a "no effects" descriptor was applied to biological data from background, reference, or control conditions.

Collectively, the effects data sets from the modeling, laboratory, and field studies were assigned an asterisk in the ascending tables and used to derive the guidelines. All of the effects data were given equal weight in the guidelines derivation. Collectively, data assigned no gradient, small gradient, no concordance, and no effects descriptors were regarded as the no-effects data set.

Derivation of Sediment Quality Guidelines

For each chemical, the data from BEDS were retrieved and arranged in ascending order of concentration in a tabular format. These ascending data tables, as reported by Long and Morgan (1990) and updated by MacDonald (1993) and Smith and MacDonald (1992), summarized the available information for each chemical or chemical group that was considered.

Table 1. Summary of available data on effects of sediment-associated acenaphthene (ppb) in coastal sediments

Concentration (±SD)	Area	Analysis type[a]	Test duration[b]	End point measured[c]
1	Puget Sound, WA	COA		Low prevalence of hepatic cellular alterations (0%)
1	Puget Sound, WA	COA		Low prevalence of hepatic lesions (0%)
1	Puget Sound, WA	COA		Low prevalence of hepatic idiopathic lesions (32.5%)
<3	Halifax Harbour, NS	COA	10 d	Significantly toxic (61.7 ± 12.5% mortality)
<3.5 ± 1	Halifax Harbour, NS	COA	10 d	Not significantly toxic (5.2 ± 3.5% mortality)
<3.5 ± 1	Halifax Harbour, NS	COA	20 d	Not significantly toxic (1 ± 2% mortality)
3.92 ± 1.59	Southern California	COA	10 d	Significantly toxic (51.7% mortality)
<5	Halifax Harbour, NS	COA	10 d	Not significantly toxic (3% mortality)
<5	Sidney Tar Pond, NS	COA	10 d	Not significantly toxic (4% mortality)
<5	Sidney Tar Pond, NS	COA	10 d	Not significantly toxic (3% mortality)
6.92 ± 11.8	Southern California	COA	10 d	Not significantly toxic (23.2% mortality)
<8.8 ± 5.3	Sidney Tar Pond, NS	COA	20 d	Not significantly toxic (8 ± 5.66% mortality)
9	San Francisco Bay, CA	AETA	48 h	San Francisco Bay AET
<12.5	Sidney Tar Pond, NS	COA	10 d	Significantly toxic (100% mortality)
<12.5	Sidney Tar Pond, NS	COA	10 d	Significantly toxic (100% mortality)
16				**ER L (10th percentile)**
16	California	AETA	48 h	California AET
16	California	AETA		California AET
16	Northern California	AETA		Northern California AET
<23.5	Sidney Tar Pond NS	COA	20 d	Significantly toxic (52% mortality)
<30.8 ± 25.6	Halifax Harbour, NS	COA	10 d	Not significantly toxic (6.8 ± 7.31% mortality)
<30.8 ± 25.6	Halifax Harbour, NS	COA	10 d	Not significantly toxic (8.5 ± 6.06% mortality)
<30.8 ± 25.6	Halifax Harbour, NS	COA	20 d	Not significantly toxic (0.7 ± 1.63% mortality)
50	Burrard Inlet, BC	SQO		Sediment quality objectives
56	Northern California	AETA	10 d	Northern California AET
56	California	AETA	10 d	California AET
56	San Francisco Bay, CA	AETA	10 d	San Francisco Bay AET
56.7 ± 70	Commencement Bay, WA	COA	48 h	Least toxic (15.1 ± 3.1% abnormality)
63	Puget Sound, WA	AETA		PSDDA screening level concentration
85.9 ± 97	Commencement Bay, WA	COA	10 d	Least toxic (12.5 ± 4.5% mortality)
119 ± 105	Commencement Bay, WA	COA	48 h	Moderately toxic (23 ± 2.3% abnormality)
127 ± 117	Commencement Bay, WA	COA	10 d	Moderately toxic (26 ± 5.2% mortality)
150	Eagle Harbor, WA	COA	4 d	LC_{50}
160	Puget Sound, WA	SQG		Chemical criteria
247 ± 147	Burrard Inlet, BC	COA	10 d	Not toxic (4.5 ± 3.02% emergence)
247 ± 147	Burrard Inlet, BC	COA	10 d	Not toxic (5.21 ± 3.61% emergence)
283 ± 140	Burrard Inlet, BC	COA	10 d	Not toxic (97.2 ± 2.84% reburial)
283 ± 140	Burrard Inlet, BC	COA	10 d	Not toxic (8.9 ± 2.99% mortality)
293 ± 73.8	Elizabeth River, VA	COA	96 h	No significant change in respiration rate
306 ± 604	Commencement Bay, WA	COA	48 h	Highly toxic (44.5 ± 19% abnormality)

The distributions of the effects data were determined using percentiles (Byrkit 1975). Two values were derived for each chemical or chemical group. The lower 10th percentile of the effects data for each chemical was identified and referred to as the effects range-low (ERL). The median, or 50th percentile, of the effects data was identified and referred to as the effects range-median (ERM). Percentiles of aquatic toxicity data were used by Klapow and Lewis (1979) to calculate marine water quality standards; the authors noted that this approach tended to minimize the influence of single (potentially outlier) data points on the development of guidelines. Environment Canada and Florida Department of Environmental Protection used a slight modification to this method, the rationale for which has been documented (MacDonald 1993, Smith and MacDonald 1992).

Determination of Percent Incidence of Adverse Biological Effects

The two guideline values, ERL and ERM, delineate three concentration ranges for a particular chemical. The concentrations below the ERL value represent a minimal-effects range; a range intended to estimate conditions in which effects would be rarely observed. Concentrations equal to and above the ERL, but be-

Species	Life stage[d]	Effects/no effects[e]	TOC (%)[f]	Reference[g]
Parophrys vetulus (English sole)	ADT	NE		1
Parophrys vetulus (English sole)	ADT	NE		1
Parophrys vetulus (English sole)	ADT	NE		1
Rhepoxynius abronius (amphipod)	ADT	NC		2
Corophium volutator (amphipod)	ADT	NE		2
Neanthes sp. (polychaete)	JUV	NE		2
Grandidierella japonica (amphipod)	JUV	NC		3
Rhepoxynius abronius (amphipod)	ADT	NE		2
Corophium volutator (amphipod)	ADT	NE		2
Rhepoxynius abronius (amphipod)	ADT	NE		2
Grandidierella japonica (amphipod)	JUV	NE		3
Neanthes sp. (polychaete)	JUV	NE		2
Oyster, mussel	LAR	*		4
Corophium volutator (amphipod)	ADT	*		2
Rhepoxynius abronius (amphipod)	ADT	*		2
Mytilus edulis (bivalve)	LAR	*		5
Benthic species		*		5
Benthic species		*		5
Neanthes sp. (polychaete)	JUV	*		2
Rhepoxynius abronius (amphipod)	ADT	NE		2
Corophium volutator (amphipod)	ADT	NE		2
Neanthes sp. (polychaete)	JUV	NE		2
Aquatic biota		NE		6
Rhepoxynius abronius (amphipod)	ADT	*		5
Rhepoxynius abronius (amphipod)	ADT	*		5
Rhepoxynius abronius (amphipod)	ADT	*		4
Oyster	LAR	NE		7
Aquatic biota		NE		8
Rhepoxynius abronius (amphipod)	ADT	NE		7
Oyster	LAR	*		7
Rhepoxynius abronius (amphipod)	ADT	SG		7
Rhepoxynius abronius (amphipod)	ADT	*		9
Benthic community		*	1	10
Rhepoxynius abronius (amphipod)	ADT	NE	2.66 ± 2.15	11
Corophium volutator (amphipod)	ADT	NE	3.18 ± 2.1	11
Rhepoxynius abronius (amphipod)	ADT	NE	2.8 ± 1.96	11
Corophium volutator (amphipod)	ADT	NE	2.8 ± 1.96	11
Palaemonetes pugio (grass shrimp)	ADT	NE		12
Oyster	LAR	*		7

(Continued)

low the ERM, represent a possible-effects range within which effects would occasionally occur. Finally, the concentrations equivalent to and above the ERM value represent a probable-effects range within which effects would frequently occur. The incidence of adverse effects within each range was quantified by dividing the number of effects entries by the total number of entries and expressed as a percent. The ERL and ERM values were derived with only the effects data set, whereas the calculations of the percent incidence of effects within each concentration range were based upon both the effects and no-effects data sets.

An evaluation of the reliability of any proposed guidelines is essential to determine their applicability in sediment quality assessments. In this study, the reliability of the guidelines for each chemical was considered to be relatively high when: (1) they agreed closely (within factors of 3.0 or less) with those developed with other methods and/or with guidelines developed with the same methods applied to different data; (2) the incidence of effects was low (<25%) in the minimal-effects ranges; (3) the incidence of effects increased consistently and markedly in concordance with increasing chemical concentrations; and

Table 1. (*Continued*)

Concentration (±SD)	Area	Analysis type[a]	Test duration[b]	End point measured[c]
350 ± 45.8	Burrard Inlet, BC	COA	10 d	Not toxic (7.9 ± 5.12% mortality)
390	Burrard Inlet, BC	COA	10 d	Highly toxic (30.5% emergence)
390	Burrard Inlet, BC	COA	10 d	Highly toxic (23% emergence)
<403	Charleston Harbor, SC	COA		High species richness (14.9 ± 2.04) SRUs
<403	Charleston Harbor, SC	COA		Moderate species richness (9.05 ± 1.33) SRUs
<403	Charleston Harbor, SC	COA		Low species richness (5.16) SRUs
<403	Charleston Harbor, SC	COA		High species diversity (4.15 ± 0.59) SDUs
<403	Charleston Harbor, SC	COA		Moderate species diversity (2.3 ± 0.2) SDUs
<403	Charleston Harbor, SC	COA		Low species diversity (1.16) SDUs
486 ± 714	Elizabeth River, VA	COA	96 h	Not significantly toxic (4.5 ± 3.24% mortality)
500	Puget Sound, WA	AETA	15 m	1986 Puget Sound AET
500	Puget Sound, WA	AETA	48 h	1986 Puget Sound AET
500				**ER M (50th percentile)**
500	Puget Sound, WA	AETA	15 m	1988 Puget Sound AET
500	Puget Sound, WA	AETA	48 h	1988 Puget Sound AET
500	Puget Sound, WA	AETA		1986 Puget Sound AET
630	Puget Sound, WA	AETA	10 d	1986 Puget Sound AET
630	Puget Sound, WA	AETA		PSDDA maximum level criteria
654 ± 1049	Commencement Bay, WA	COA	10 d	Highly toxic (78.5 ± 19.5% mortality)
679 ± 469	Elizabeth River, VA	COA	96 h	Significantly toxic (50.7 ± 39% mortality)
680 ± 814	Elizabeth River, VA	COA	96 h	Significant decrease in respiration rates
730	Puget Sound, WA	AETA		1988 Puget Sound AET
2000	Puget Sound, WA	AETA	10 d	1988 Puget Sound AET
3031 ± 4271	Puget Sound, WA	COA	10 d	High prevalence of hepatic lesions (26.7 ± 6.4%)
3031 ± 4271	Puget Sound, WA	COA		High prevalence of hepatic idiopathic lesions (88.0 ± 3.7%)
3031 ± 4271	Puget Sound, WA	COA		High prevalence of hepatic cellular alterations (44.1 ± 8.5%)
5599 ± 24,392	Eagle Harbor, WA	COA	10 d	Least toxic (13 ± 7% mortality)
6522 ± 8915	Eagle Harbor, WA	COA	10 d	Moderately toxic (41 ± 9% mortality)
16,500	United States	EqPA		Chronic marine EqP threshold
39,557 ± 48,678	Eagle Harbor, WA	COA	10 d	Highly toxic (95.5 ± 8.5 mortality)

[a]Analysis type: COA = co-occurrence analysis; AETA = apparent effects threshold approach; EqPA = equilibrium partitioning approach; SQO = sediment quality objective; SQG = sediment quality guideline; SSBA = spiked sediment bioassay approach; SLCA = screening level criteria approach.

[b]Test duration: d = day; h = hour; m = minute.

[c]End point measured: AET = apparent effects threshold; PSDDA = Puget Sound dredge disposal analysis; LC_{50} = lethal concentration to 50% of the tested organisms; SRUs = species richness units; SDUs = species diversity units.

[d]Life stage: ADT = adult; LAR = larval; JUV = juvenile.

[e]Effects/No effects: NE = no effect; NC = no concordance; SG = small gradient; NG = no gradient; * = effects data used to calculate ERL and ERM values.

[f]1, Malins and others, 1985; 2, Tay and others, 1990; 3, Anderson and others, 1988; 4, Long and Morgan, 1990; 5, Becker and others, 1990; 6, Swain and Nijman, 1991; 7, Tetra-Tech, 1985; 8, US Army Corps of Engineers, 1988; 9, Swartz, and others, 1989; 10, Washington Department of Ecology, 1989; 11, McLeay and others, 1991; 12, Alden and Butt, 1987; 13, Winn and others, 1989; 14, Beller and others, 1986; 15, PTI, Inc., 1988; 16, CH2M-Hill, Inc., 1989; 17, Bolton, 1985.

(4) the incidence of effects was very high (>75%) in the probable-effects ranges. The reliability of the guidelines that failed to meet these evaluation criteria was considered to be lower.

Results

ERL and ERM values were derived for 28 substances: nine trace metals, total PCBs, 13 individual polynuclear aromatic hydrocarbons (PAHs), three classes of PAHs (total low molecular weight, total high molecular weight, and total PAH), and two pesticides (p,p'-DDE and total DDT). The data available for acenaphthene and phenanthrene are shown in Tables 1 and 2, respectively, to illustrate the format and content of the ascending tables with which the guidelines were derived. Space limitations preclude inclusion of equivalent tables for all of the substances.

Species	Life stage[d]	Effects/no effects[e]	TOC (%)[f]	Reference[g]
Rhepoxynius abronius (amphipod)	ADT	NE	2.64 ± 2.14	11
Rhepoxynlus abronius (amphipod)	ADT	SG	3.5	11
Corophium volutator (amphipod)	ADT	SG	3.5	11
Benthic species		NE		13
Benthic species		NG		13
Benthic species		NG		13
Benthic species		NE		13
Benthic species		NG		13
Benthic species		NG		13
Palaemonetes pugio (grass shrimp)	ADT	NE		12
Microtox		*		14
Crassostrea gigas (oyster)	LAR	*		14
Microtox		*		15
Crassostrea gigas (oyster)	LAR	*		15
Benthic species		*		14
Rhepoxynius abronius (amphipod)		*		14
Aquatic biota		*		8
Rhepoxynius abronius (amphipod)	ADT	*		7
Palaemonetes pugio (grass shrimp)	ADT	SG		12
Palaemonetes pugio (grass shrimp)	ADT	*		12
Benthic community		*		15
Rhepoxynius abronius (amphipod)	ADT	*		15
Parophrys vetulus (English sole)	ADT	*		1
Parophrys vetulus (English sole)	ADT	*		1
Parophrys vetulus (English sole)	ADT	*		1
Rhepoxynius abronius (amphipod)	ADT	NE		16
Rhepoxynius abronius (amphipod)	ADT	SG		16
Aquatic biota		*	1	17
Rhepoxynius abronius (amphipod)	ADT	*		16

Adverse effects measured in association with acenaphthene included high amphipod mortality in sediment toxicity tests, low species richness in benthic communities, high prevalence of liver lesions in demersal fish, and chronic toxicity predicted by an equilibrium-partitioning model (Table 1). No data from spiked-sediment bioassays were available. As an example of the kinds of data analyses that were performed for entry into the BEDS, matching sediment chemistry and amphipod mortality data from Commencement Bay (Washington) were evaluated in a co-occurrence analysis. The average concentration of acenaphthene was 85.9 ppb in the samples that were the least toxic to amphipods (12.5 ± 4.5% mortality). This data entry was assigned a no-effects (ne) descriptor. In samples that were moderately toxic (26 ± 5.2% mortality), the average concentration of acenaphthene was 127 ppb. The ratio of 127 ppb to 85.9 ppb was less than 2.0, therefore, the moderately toxic data entry was assigned a small-gradient descriptor. The average acenaphthene concentration associated with highly toxic samples (78.5 ± 19.5% mortality) was 654 ppb, a factor 7.6-fold higher than the average concentration in the least toxic samples. It was assigned an asterisk and used in the calculation of the ERL and ERM values. A total of 30 data entries for acenaphthene were assigned effects designators. No biological effects were reported over the range of 1–8.8 ppb acenaphthene. The lower 10th percentile value of the effects data (the ERL) was 16 ppb and the median value (ERM) was 500 ppb. The percent incidence of adverse effects within the minimal-effects, possible-effects, and probable-effects ranges were 20%, 32%, and 84%, respectively.

Phenanthrene data were available from equilibrium-partitioning studies, spiked sediment bioassays, and numerous field surveys performed in many different areas (Table 2). A total of 51 data entries were assigned effects designators in the phenanthrene database. Adverse effects were not observed in asso-

Table 2. Summary of available data on effects of sediment-associated phenanthrene (ppb) in coastal sediments

Concentration (±SD)	Area	Analysis type[a]	Test duration[b]	End point measured[c]
4.6 ± 1.6	Laboratory	SSBA	~4 mo	No significant change in liver somatic indices
<5	Halifax Harbour, NS	COA	10 d	Not significantly toxic (3% mortality)
<5	Sidney Tar Pond, NS	COA	10 d	Not significantly toxic (3% mortality)
15	Burrard Inlet, BC	SQO		Sediment quality objectives
<20	Sidney Tar Pond, NS	COA	10 d	Not significantly toxic (4% mortality)
39.4 ± 47.6	Laboratory	SSBA	~4 mo	No signifcant change in kidney MFO induction
64.6	San Francisco Bay, CA	COA	48 h	Least toxic (23.3 ± 7.3% abnormal)
66.2 ± 57.5	Laboratory	SSBA	~4 mo	No significant change in spleen condition indices
88	San Francisco Bay, CA	AETA	48 h	San Francisco Bay AET
110	United States	EqPA		99% chronic marine criteria
119	Southern California	COA	10 d	Not significantly toxic (23.2% mortality)
150	Puget Sound, WA	COA		Low occurrence of hepatic cellular alterations (0%)
150	Puget Sound, WA	COA		Low prevalence of hepatic lesions (0%)
150	Puget Sound, WA	COA		Low prevalence of hepatic idiopathic lesions (32.5%)
159	San Francisco Bay, CA	COA	48 h	Not significantly toxic (31.9 ± 15.5% abnormal)
170	California	AETA	48 h	California AET
170	Northern California	AETA		Northern California AET
180 ± 325	Narragansett Bay, RI	COA	10 d	Not significantly toxic (5.28 ± 3.04% mortality)
188	San Francisco Bay, CA	COA	10 d	Least toxic (18 ± 6.6% mortality)
199	San Francisco Bay, CA	COA	10 d	Not significantly toxic (18.4 ± 6.8% mortality)
220	San Francisco Bay, CA	COA	10 d	Significantly toxic (42.9 ± 19.2% mortality)
222 ± 136	Southern California	COA	10 d	Significantly toxic (51.7% mortality)
223 ± 169	Burrard Inlet, BC	COA	10 d	Not toxic (4.5 ± 3.02% emergence)
223 ± 169	Burrard Inlet, BC	COA	10 d	Not toxic (5.21 ± 3.61% emergence)
224	San Francisco Bay, CA	COA	48 h	Moderately toxic (59.4 ± 11.3% abnormal)
228	San Francisco Bay, CA	COA	10 d	Moderately toxic (33.8 ± 4.7 mortality)
233	San Francisco Bay, CA	COA	48 h	Significantly toxic (55.7 ± 22.7% abnormal)
240	United States	EqPA		95% chronic marine criteria
240				**ER L (10th percentile)**
242	San Francisco Bay, CA	COA	10 d	Highly toxic (67 ± 11.8% mortality)
259	United States	SLCA		NSLC-marine
270	California	AETA		California AET values
270	Southern California	AETA		Southern California AET values
>290	Southern California	AETA	10 d	Southern California AET values
297	Commencement Bay, WA	COA	48 h	Least toxic (15.1 ± 3.1% abnormality)
316 ± 582	Elizabeth River, VA	COA	96 h	No significant change in respiration rate
320	Puget Sound, WA	AETA		PSSDA screening level concentration
368	United States	SLCA		NSLC-marine
374 ± 461	Elizabeth River, VA	COA	96 h	Not significantly toxic (4.5 ± 3.24% mortality)
383 ± 332	Laboratory	SSBA	~4 mo	Significant change in liver somatic indices
<403	Charleston Harbor, SC	COA		High species richness (14.9 ± 2.04) SRUs
<403	Charleston Harbor, SC	COA		Moderate species richness (9.05 ± 1.33) SRUs
<403	Charleston Harbor, SC	COA		Low species richness (5.16) SRUs
<403	Charleston Harbor, SC	COA		High species diversity (4.15 ± 0.59) SDUs
<403	Charleston Harbor, SC	COA		Moderate species diversity (2.3 ± 0.2) SDUs
<403	Charleston Harbor, SC	COA		Low species diversity (1.16) SDUs
<408 ± 501	Halifax Harbour, NS	COA	10 d	Not significantly toxic (6.8 ± 7.31% mortality)
<408 ± 501	Halifax Harbour, NS	COA	20 d	Not significantly toxic (0.7 ± 1.63% mortality)
<410 ± 498	Halifax Harbour, NS	COA	10 d	Not significantly toxic (8.5 ± 6.06% mortality)
475	San Francisco Bay, CA	COA	48 h	Highly toxic (92.4 ± 4.5% abnormal)
478	Commencement Bay, WA	COA	10 d	Least toxic (12.5 ± 4.5% mortality)
487 ± 318	Laboratory	SSBA	~4 mo	Significant increase in kidney MFO induction
510	Northern California	AETA	10 d	Northern California AET
510	California	AETA	10 d	California AET
510	San Francisco Bay, CA	AETA	10 d	San Francisco Bay AET
593	Commencement Bay, WA	COA	48 h	Moderately toxic (23 ± 2.3% abnormality)

Species	Life stage[d]	Effects/no effects[e]	TOC (%)	Reference[f]
Pseudopleuronectes americanus (flounder)	ADT	NE		18
Rhepoxynius abronius (amphipod)	ADT	NE		2
Rhepoxynius abronius (amphipod)	ADT	NE		2
Aquatic biota		NE		6
Corophium volutator (amphipod)	ADT	NE		2
Pseudopleuronectes americanus (flounder)	ADT	NE		18
Bivalve	LAR	NE		4
Pseudopleuronectes americanus (flounder)	ADT	NE		18
Oyster, mussel	LAR	*		4
Aquatic organisms		*	1	19
Grandidierella japonica (amphipod)	JUV	NE		3
Parophyrs vetulus (English sole)	ADT	NE		1
Parophyrs vetulus (English sole)	ADT	NE		1
Parophrys vetulus (English sole)	ADT	NE		1
Bivalve	LAR	NE		4
Mytilus edulis (bivalve)	LAR	*		5
Benthic species		*		5
Ampelisca abdita (amphipod)	ADT	NE		20
Rhepoxynius abronius (amphipod)	ADT	NE		4
Rhepoxynius abronius (amphipod)	ADT	NE		4
Rhepoxynius abronius (amphipod)	ADT	SG		4
Grandidierella japonica (amphipod)	JUV	SG		3
Rhepoxynius abronius (amphipod)	ADT	NE	2.68 ± 2.15	11
Corophium volutator (amphipod)	ADT	NE	3.18 ± 2.1	11
Bivalve	LAR	*		4
Rhepoxynius abronius (amphipod)	ADT	SG		4
Bivalve	LAR	SG		4
Aquatic organisms		*	1	19
Rhepoxynius abronius (amphipod)	ADT	SG		4
Benthic species		*	1	21
Benthic species		*		5
Benthic species		*		5
Rhepoxynius abronius (amphipod)	ADT	—		5
Oyster	LAR	NE		7
Palaemonetes puglo (grass shrimp)	ADT	NE		12
Aquatic biota		NE		8
Benthic species		*	1	21
Palaemonetes puglo (grass shrimp)	ADT	NE		12
Pseudopleuronectes americanus (flounder)	ADT	*		18
Benthic species		NE		13
Benthic species		NG		13
Benthic species		NG		13
Benthic species		NE		13
Benthic species		NG		13
Benthic species		NG		13
Rhepoxynius abronius (amphipod)	ADT	NE		2
Neanthes species (polychaete)	JUV	NE		2
Corophium volutator (amphipod)	ADT	NE		2
Bivalve	LAR	*		4
Rhepoxynius abronius (amphipod)	ADT	NE		7
Pseudopleuronectes americanus (flounder)	ADT	*		18
Rhepoxynius abronius (amphipod)	ADT	*		5
Rhepoxynius abronius (amphipod)	ADT	*		5
Rhepoxynius abronius (amphipod)	ADT	*		4
Oyster	LAR	*		7

(*Continued*)

Table 2. (*Continued*)

Concentration (±SD)	Area	Analysis type[a]	Test duration[b]	End point measured[c]
597	Commencement Bay, WA	COA	10 d	Moderately toxic (26 ± 5.2% mortality)
670	Laboratory	SSBA	~4 mo	Significant change in spleen condition indices
918 ± 1395	Burrard Inlet, BC	COA	10 d	Not toxic (97.2 ± 2.84% reburial)
918 ± 1395	Burrard Inlet, BC	COA	10 d	Not toxic (8.9 ± 2.99% mortality)
950	Eagle Harbor, WA	COA	4 d	LC_{50}
987 ± 1654	Elizabeth River, VA	COA	96 h	Significant decrease in respiration rates
1000	Puget Sound, WA	SQG		Chemical criteria
1020	United States	EqPA		Interim marine sediment quality criteria (FCV)
1213 ± 1547	Burrard Inlet, BC	COA	10 d	Not toxic (7.9 ± 5.12% mortality)
<1267 ± 2528	Halifax Harbour, NS	COA	20 d	Not significantly toxic (1 ± 2% mortality)
<1271 ± 2526	Halifax Harbour, NS	COA	10 d	Not significantly toxic (5.2 ± 3.5% mortality)
1379 ± 2545	Commencement Bay, WA	COA	48 h	High toxic (44.5 ± 19% abnormality)
1500	Puget Sound, WA	AETA	15 m	1986 Puget Sound AET
1500	Puget Sound, WA	AETA	48 h	1986 Puget Sound AET
1500	Puget Sound, WA	AETA	15 m	1988 Puget Sound AET
1500				**ER M (50th percentile)**
1500	Puget Sound, WA	AETA	48 h	1988 Puget Sound AET
<1688 ± 2920	Halifax Harbour, NS	COA	96 h	Significantly toxic (61.7 ± 12.5% mortality)
1913 ± 2693	Elizabeth River, VA	COA	10 d	Significantly toxic (50.7% ± 39% mortality)
2142	Eagle Harbor, WA	COA	10 d	Moderately toxic (41 ± 9% mortality)
2600	Eagle Harbor, WA	COA	10 d	Least toxic (13 ± 7% mortality)
2838	Commencement Bay, WA	COA	10 d	Highly toxic (78.5 ± 19.5% mortality)
3000	Burrard Inlet, BC	COA	10 d	Highly toxic (30.5% emergence)
3000	Burrard Inlet, BC	COA	10 d	Highly toxic (23% emergence)
3200	Puget Sound, WA	AETA		PSDDA maximum level criteria
3200	Puget Sound, WA	AETA		1988 Puget Sound AET
3680	Eagle Harbor, WA	COA	4 d	LC_{50}
5400	Puget Sound, WA	AETA	10 d	1986 Puget Sound AET
5400	Puget Sound, WA	AETA		1988 Puget Sound AET
6900	Puget Sound, WA	AETA	10 d	1988 Puget Sound AET
10,000	Laboratory	SSBA	10 d	Significant toxicity
11,656 ± 14,472	Puget Sound, WA	COA		High prevalence of hepatic lesions (26.7 ± 6.4%)
11,656 ± 14,472	Puget Sound, WA	COA		High prevalence of hepatic idiopathic lesions (88.0 ± 3.7%)
11,656 ± 14,472	Puget Sound, WA	COA		High prevalence of hepatic cellular alterations (44.2 ± 8.5%)
14,000	United States	EqPA		Chronic marine EqP threshold
14,000	United States	EqPA		EPA acute marine EqP threshold
>30,000	Laboratory	SSBA	14 d	LC_{50}
>30,000	Laboratory	SSBA	14 d	LC_{50}
33,603	Eagle Harbor, WA	COA	10 d	Highly toxic (95.5 ± 8.5% mortality)
<45,903 ± 64,909	Sidney Tar Pond, NS	COA	20 d	Not significantly toxic (8 ± 5.66% mortality)
91,800	Sidney Tar Pond, NS	COA	10 d	Significantly toxic (100% mortality)
91,800	Sidney Tar Pond, NS	COA	10 d	Significantly toxic (100% mortality)
105,500	Elizabeth River, VA	COA	28 d	LC_{50}
484,000	Sidney Tar Pond, NS	COA	20 d	Significantly toxic (52% mortality)
2,363,200	Elizabeth River, VA	COA	24 h	LC_{50}
4,220,000	Elizabeth River, VA	COA	2 h	Highly toxic (100% mortality)

[a]Analysis type: COA ± co-ocurrence analysis; AETA = apparent effects threshold approach; EqPA = equilibrium partitioning approach; SQO = sediment quality objective; SQG = sediment quality guideline; SSBA = spike sediment bioassay approach; SLCA = screening level criteria approach.

[b]Test duration: d = day; h = hour; min = minute; mo = month.

[c]End point measured: ER L = effects range low; ER M = effects range-median; AET = apparent effects threshold; PSDDA = Puget Sound dredge disposal analysis; organisms; SRUs = species richness units; SDUs = species diversity units; MFO = mixed-function oxidase; FCV = final chronic value; LC_{50} = lethal concentration to 50% of the tested organisms; EPA = Environmental Protection Agency.

Species	Life stage[d]	Effects/no effects[e]	TOC (%)	Reference[f]
Rhepoxynius abronius (amphipod)	ADT	*		7
Pseudopleuronectes americanus (flounder)	ADT	*		18
Rhepoxynius abronius (amphipod)	ADT	NE	2.8 ± 1.96	11
Corophium volutator (amphipod)	ADT	NE	2.8 ± 1.96	11
Rhepoxynius abronius (amphipod)	JUV/ADT	*		9
Palaemonetes puglo (grass shrimp)	ADT	*		12
Benthic community		*	1	10
Benthic community		NE	1	22
Rhepoxynius abronius (amphipod)	ADT	NE	2.64 ± 2.14	11
Neanthes species (polychaete)	JUV	NE		2
Corophium volutator (amphipod)	ADT	NE		2
Oyster	LAR	*		7
Microtox		*		14
Crassostrea gigas (oyster)	LAR	*		15
Microtox		*		
Crassostrea gigas (oyster)	LAR	*		15
Rhepoxynius abronius (amphipod)	ADT	*		2
Palaemonetes pugio (grass shrimp)	ADT	*		12
Rhepoxynius abronius (amphipod)	ADT	NC		16
Rhepoxynius abronius (amphipod)	ADT	NE		16
Rhepoxynius abronius (amphipod)	ADT	*		7
Rhepoxynius abronius (amphipod)	ADT	*	3.5	11
Corophium volutator (amphipod)	ADT	*	3.5	11
Aquatic biota		*		8
Benthic species		*		14
Rhepoxynius abronius (amphipod)	JUV/ADT	*		9
Rhepoxynius abronius (amphipod)	ADT	*		14
Benthic community	ADT	*		15
Rhepoxynius abronius (amphipod)	ADT	*		15
Rhepoxynius abronius (amphipod)	ADT	*	0.9	23
Parophrys vetulus (English sole)	ADT	*		1
Parophrys vetulus (English sole)	ADT	*		1
Parophrys vetulus (English sole)	ADT	*		1
Aquatic biota		*	1	17
Aquatic biota		*	1	24
Grandidierella japonica (amphipod)	ADT	—	0.1	25
Grandidierella japonica (amphipod)	ADT	—	1	25
Rhepoxynius abronius (amphipod)	ADT	*		16
Neanthes species (polychaete)	JUV	NE		2
Corophium volutator (amphipod)	ADT	*		2
Rhepoxynius abronius (amphipod)	ADT	*		2
Leiostomus xanthurus (spot)	JUV	*		26
Neanthes species (polychaete)	JUV	*		2
Leiostomus xanthurus (spot)	JUV	*		26
Leiostomus xanthurus (spot)	JUV	*		26

[d]Life stage: ADT = adult; LAR = larval; JUV = juvenile.

[e]Effects/no effects: NE = no effect; NC = no concordance; SG = small gradient; NG = no gradient; * = effects data used to calculate ERL and ERM values.

[f]1, Malins and others, 1985; 2, Tay and others, 1990; 3, Anderson and others, 1988; 4, Long and Morgan, 1990; 5, Becker and others, 1990; 6, Swain and Nijman, 1991; 7, Tetra-Tech, 1985; 8, US Army Corps of Engineers, 1988; 9, Swartz and others, 1989; 10, Washington Department of Ecology, 1989; 11, McLeay and others, 1991; 12, Alden and Butt, 1987; 13, Winn and others, 1989; 14, Bellar and others, 1986; 15, PTI, Inc., 1988; 16, CH2M-Hill, Inc., 1989; 17, Bolton, 1985; 18, Payne and others, 1988; 19, Pavlou and others, 1987; 21, Neff and others, 1986; 22, US EPA, 1988; 23, Plesha and others, 1988; 24, Lyman and others, 1987; 25, SCCWRP, 1989; 26, Roberts and others, 1989.

Table 3. ERL and ERM guideline values for trace metals (ppm, dry wt) and percent incidence of biological effects in concentration ranges defined by the two values

	Guidelines		Percent (ratios) incidence of effects[a]		
Chemical	ERL	ERM	<ERL	ERL-ERM	>ERM
Arsenic	8.2	70	5.0 (2/40)	11.1 (8/73)	63.0 (17/27)
Cadmium	1.2	9.6	6.6 (7/106)	36.6 (32/87)	65.7 (44/67)
Chromium	81	370	2.9 (3/102)	21.1 (15/71)	95.0 (19/20)
Copper	34	270	9.4 (6/64)	29.1 (32/110)	83.7 (36/43)
Lead	46.7	218	8.0 (7/87)	35.8 (29/81)	90.2 (37/41)
Mercury	0.15	0.71	8.3 (4/48)	23.5 (16/68)	42.3 (22/52)
Nickel	20.9	51.6	1.9 (1/54)	16.7 (8/48)	16.9 (10/59)
Silver	1.0	3.7	2.6 (1/39)	32.3 (11/34)	92.8 (13/14)
Zinc	150	410	6.1 (6/99)	47.0 (31/66)	69.8 (37/53)

[a]Number of data entries within each concentration range in which biological effects were observed divided by the total number of entries within each range.

ciation with phenanthrene concentrations of <5 ppb to 66 ppb. The ERL value for phenanthrene was 240 ppb and the ERM value was 1500 ppb. The percent incidence of adverse effects within the minimal-effects, possible-effects, and probable-effects ranges were 18%, 46%, and 90%, respectively.

The incidence of adverse effects increased with increasing concentrations of all trace metals, except nickel (Table 3). The incidence of effects was 10% or less in the minimal-effects ranges and 11%–47% in the possible-effects ranges from all of the trace metals. The incidence of adverse effects exceeded 75% in the probable-effects ranges for chromium, copper, lead, and silver but was only 42.3% and 16.9% for mercury and nickel, respectively. However, the incidence of effects in the probable-effects range for chromium was greatly influenced and exaggerated by data from multiple tests conducted in only two field surveys.

The incidence of adverse effects consistently and markedly increased with increasing concentrations of all organic compounds, except p,p'-DDE and total DDT (Table 4). The incidence of effects ranged from 5.0% to 27.3% in the minimal-effects ranges for organic compounds and was 25% or less for all but one of the compounds—fluorene. Within the possible-effects ranges, the incidence of effects ranged from 18% to 75%. The incidence of effects ranged from 50% to 100% in the probable-effects ranges and equaled or exceeded 75% for all but four compounds. The incidence of effects in the probable-effects range for total PCBs was relatively low (51%).

Discussion

Guidelines Accuracy

Among the trace metals, the most accurate guidelines appeared to be those for copper, lead, and silver; the incidence of effects were very low (<10%) in the minimal-effects ranges, increased steadily through the possible-effects and probable-effects ranges, and were very high (>83%) in the probable-effects ranges. Among the organic compounds, the guidelines appeared to be highly accurate for all of the classes of PAHs and most of the individual PAHs. Except for fluorene, the incidence of effects was 25% or less at concentrations below the respective ERL values. Except for dibenzo(a,h)anthracene, p,p'-DDE, total DDT, and total PCBs, the incidence of effects was 75% or greater at concentrations that exceeded the respective ERMs. At concentrations in the probable-effects ranges, the incidence of adverse effects was 100% for acenaphthylene, 2-methyl naphthalene, and low-molecular-weight PAHs and 90% or greater for chromium, lead, silver, benz(a)anthracene, and fluoranthene.

The accuracy of the guidelines for some substances appeared to be relatively low. For example, the incidences of effects associated with nickel were 1.9%, 16.7%, and 16.9%, respectively, in the three concentration ranges. The incidence of effects did not increase appreciably with increasing concentrations of nickel and were very low in all three ranges. The incidence of effects in the probable-effects ranges for mercury and total PCBs were relatively low (42.3% and 51.0%, respectively). Furthermore, the incidence of effects did not increase consistently and markedly with increasing concentrations of p,p'-DDE, and total DDT. The p,p'-DDE and total DDT databases may have been unduly influenced by relatively low equilibrium-partitioning values, which were based upon chronic marine water quality criteria intended to protect against bioaccumulation in marine fish and birds, not toxicity to benthic organisms. The incidence of effects in the probable-effects range for chromium

Table 4. ERL and ERM guideline values for organic compounds (ppb, dry wt) and percent incidence of biological effects in concentration ranges defined by the two values

Chemical	Guidelines		Percent (ratios) incidence of effects[a]		
	ERL	ERM	<ERL	ERL-ERM	>ERM
Acenaphthene	16	500	20.0 (3/15)	32.4 (11/34)	84.2 (16/19)
Acenaphthylene	44	640	14.3 (1/7)	17.9 (5/28)	100 (9/9)
Anthracene	85.3	1100	25.0 (4/16)	44.2 (19/43)	85.2 (23/27)
Fluorene	19	540	27.3 (3/11)	36.5 (19/52)	86.7 (26/30)
2-Methyl naphthalene	70	670	12.5 (2/16)	73.3 (11/15)	100 (15/15)
Naphthalene	160	2100	16.0 (4/25)	41.0 (16/39)	88.9 (24/27)
Phenanthrene	240	1500	18.5 (5/27)	46.2 (18/39)	90.3 (28/31)
Low-molecular weight PAH	552	3160	13.0 (3/23)	48.1 (13/27)	100 (16/16)
Benz(a)anthracene	261	1600	21.1 (4/19)	43.8 (14/32)	92.6 (25/27)
Benzo(a)pyrene	430	1600	10.3 (3/29)	63.0 (17/27)	80.0 (24/30)
Chrysene	384	2800	19.0 (4/21)	45.0 (18/40)	88.5 (23/26)
Dibenzo(a,h)anthracene	63.4	260	11.5 (3/26)	54.5 (12/22)	66.7 (16/24)
Fluoranthene	600	5100	20.6 (7/34)	63.6 (28/44)	92.3 (36/39)
Pyrene	665	2600	17.2 (5/29)	53.1 (17/32)	87.5 (28/32)
High molecular weight PAH	1700	9600	10.5 (2/19)	40.0 (10/.25)	81.2 (13/16)
Total PAH	4022	44792	14.3 (3/21)	36.1 (13/36)	85.0 (17/20)
p,p′-DDE	2.2	27	5.0 (1/20)	50.0 (10/20)	50.0 (12/24)
Total DDT	1.58	46.1	20.0 (2/10)	75.0 (12/16)	53.6 (15/28)
Total PCBs	22.7	180	18.5 (5/27)	40.8 (20/49)	51.0 (25/49)

[a]Number of data entries within each concentration range in which biological effects were observed divided by the total number of entries within each range.

ostensibly appeared to be very high but was unduly exaggerated by data from multiple tests performed in only two studies.

Comparisons with Other Guidelines

Agreement within a factor of 3 or less among guidelines developed with different methods has been recommended by a panel of experts as an indication of good precision (Lorenzato and others 1991). In the following discussion, the comparisons of guidelines were conducted by determining the ratios between them, i.e., the larger of the two values was divided by the smaller value.

The ERL and ERM values reported in Tables 3 and 4 were based upon a considerable expansion and revision of the database used by Long and Morgan (1990). The quantities of data used to derive the present values exceeded those used previously by factors of 1.4 to 2.6. About 30%–50% of the data used in the present analysis came from the database used previously. Furthermore, the considerable amounts of freshwater data in the previous database were deleted in the present analysis. Of the 25 ERL values derived in the two analyses, seven remained unchanged, nine decreased, and nine increased. The ratios between the two sets of ERL values ranged from 1.0 to 9.4 (average of 1.88, N = 25). The ERL values for only two substances changed by factors greater than 3.0×: arsenic (decreased by 4.2×); and acenaphthene (decreased by 9.4×). The ratios between the two sets of ERM values ranged from 1.0 to 7.6 (average of 1.63, N = 25). The average ratios between the two sets of ERM values was 1.2 for the individual PAHs and 1.5 for the trace metals; seven remained unchanged, seven decreased, and eight increased. Only one ERM value changed by a factor greater than 3.0: total DDT (decreased by 7.6×). The ERL and ERM values for p,p′-DDE increased by factors of 1.1 and 1.8, respectively. The ERL value for total PAHs remained unchanged and the ERM value increased by a factor of 1.3. The results of these comparisons indicate that the guidelines are relatively insensitive to changes in the database, once the minimum data requirements have been satisfied.

The national sediment quality criteria proposed by the US Environmental Protection Agency for fluoranthene, acenaphthene, and phenanthrene in salt water are based upon equilibrium-partitioning models (US EPA 1993a–c). The proposed mean criterion for fluoranthene is 300 μg/g organic carbon (with 95% confidence limits of 140 and 640 μg/goc). For acenaphthene the mean criterion is 240 μg/goc (with 95% confidence limits of 110 and 500 μg/goc). For phenanthrene the mean criterion is 240 μg/goc (with 95% confidence limits of 110 and 510 μg/goc). Assuming a TOC concentration of 1%, these criteria

values are equivalent to 3000 (1400–6400) ppb dry weight for fluoranthene; 2400 (1100–5000) ppb dry weight for acenaphthene; and 2400 (1100–5100) ppb dry weight for phenanthrene. The mean criteria exceeded the ERM values of 500 ppb for acenaphthene and 1500 ppb for phenanthrene by factors of 4.8, and 1.6, respectively. The criterion for fluoranthene was lower than the ERM by a factor of 1.7. The criteria expressed in units of dry weight would increase with increasing TOC concentrations.

The ERL and ERM values generally agreed within factors of two to three with freshwater effects-based criteria issued by Ontario (Persaud and others 1992). Lowest effect levels and severe effect levels were reported, based upon a screening level concentration (SLC) approach applied to matching benthic community and sediment chemistry data. The ratios between the present ERL values and the lowest effect levels for Ontario ranged from 1.25 to 3.1 (average of 1.7) for eight trace metals (As, Cd, Cr, Cu, Pb, Hg, Ni, Zn). The ratios between the present ERM values and the severe effect levels for Ontario ranged from 1.0 to 3.4 (average of 2.0) for the same eight trace metals. Of the 16 comparisons, the ERL/ERM values were lower than the respective values for Ontario in six cases and higher in ten cases.

Among all of these comparisons, most of the guidelines agreed within the recommended factor of 3.0 or less. In the worse case, two values (previous and present ERL values for acenaphthene) differed by a factor of 9.4.

Merits of the Approach

This approach attempts to identify the concentrations of toxicants that are rarely associated with adverse biological effects and those usually associated with effects, based upon data from many studies. The advantages of this approach are that guidelines can be developed quickly with existing information and that they are based upon data gathered from many different studies. An underlying assumption of the approach is that, if enough data are accumulated, a pattern of increasing incidence of biological effects should emerge with increasing contaminant concentrations.

Data from all available sources were considered in this study, including those from equilibrium-partitioning models, spiked sediment bioassays, and numerous field surveys. The modeling and bioassay methods differ considerably from those used in the field studies, since they generally are performed with single chemicals as if they were acting alone. The field studies invariably involve complex mixtures of contaminants, acting synergistically, additively, or antagonistically. Whereas the modeling studies and spiked sediment bioassays can be used to establish cause–effect relationships for single chemicals, the data from field studies cannot establish such relationships. However, the data from field studies of complex mixtures reflect real-world, natural conditions in ambient sediments. We believe that the most meaningful assessment tools are those that are based upon evidence from and agreement among all three of these methods. If data compiled from different study areas with different pollution histories and physical–chemical properties converge upon ranges of contaminant concentrations that are usually associated with effects, then guidelines derived from those studies should be broadly applicable to many other areas and situations. Therefore, in this report, the data from numerous studies were used to identify the concentrations of individual chemicals that were rarely, occasionally, and usually associated with effects.

The biological data compiled for derivation of the guidelines included a variety of different taxonomic groups and toxicological end points. The sensitivities of the taxa to toxicants may have differed considerably, and, therefore, contributed to variability in the data base. However, we believe that the inclusion of data from multiple taxa ensures the broad applicability of the guidelines and the protection of a diversity of organisms.

The bioavailability of sediment-associated contaminants is controlled to a large degree by certain physical–chemical properties of the sediments. For example, high acid-volatile sulfide (AVS) concentrations appear to reduce the bioavailability of cadmium, and, possibly, other trace metals in sediments (Di Toro and others 1990). Similarly, the influence of increasing TOC concentrations in reducing the bioavailability of many nonionic organic compounds has been demonstrated in modeling and laboratory studies (Di Toro and others 1991, Swartz and others 1990, Pavlou and others 1987). Significant differences in toxicity can occur at similar toxicant concentrations over relatively small ranges in TOC and/or AVS concentrations (Adams and others 1992). It has been argued that sediment quality criteria are indefensible if they do not account for factors that control bioavailability (Di Toro and others 1991). The data evaluated in the present analysis were not normalized to either TOC or AVS concentrations, since only a small minority of the reports that were encountered included results for these parameters. Nevertheless, the present evaluation indicates that the guidelines derived using the approach reported herein are accurate for most

chemicals and agree reasonably well with other guidelines. Therefore, they are likely to be reliable tools in sediment quality assessments.

While factors that are thought to control bioavailability were not considered explicity, surely they were operative in the tests of field-collected sediments and influenced the bioavailability of all of the potential toxicants. However, the data that were encountered indicated that TOC concentrations usually ranged from 1% to 3% in most study areas. In contrast, the concentrations of some chemicals differed by several orders of magnitude among the same samples. These observations suggest that, over these large concentration gradients, the relatively small differences in TOC and/or AVS concentrations may have been relatively unimportant in controlling toxicity or, otherwise, were masked in the data analyses.

Since the data bases used to develop the present guidelines included data from many field studies, the guidelines may tend to be more protective than those based upon only single-chemical approaches. The cumulative (e.g., synergistic) effects of mixtures of toxicants in ambient sediments, including those not quantified may tend to drive the apparent effective concentrations of individual toxicants downward (i.e., toward lower concentrations).

Conclusions

Based upon an evaluation of existing data, three ranges in chemical concentrations were determined for 28 chemicals or chemical classes. These ranges were defined by two guideline values: the lower 10th percentile (ERL) and the 50th percentile (ERM) of the effects data distribution. The incidence of biological effects was quantified for each of these ranges as an estimate of the accuracy of the guidelines. The incidence of effects usually was less than 25% at concentrations below the ERL values. For most chemicals, the incidence of effects increased markedly as the concentrations increased. Furthermore, the incidences of effects often were greater than 75% (occasionally 100%) at concentrations that exceeded the ERM values. However, for a few chemicals (especially mercury, nickel, total PCBs, total DDT, and p,p'-DDE) there were relatively weak relationships between their concentrations and the incidence of effects. The guideline values reported herein generally agreed within factors of 3× or less with guidelines derived earlier using the same methods applied to a different data base and with guidelines developed with other methods. The numerical guidelines should be used as informal screening tools in environmental assessments. They are not intended to preclude the use of toxicity tests or other measures of biological effects. The guidelines should be accompanied by the information on the incidence of effects. The percent incidence data may prove useful in estimating the probability of observing similar adverse effects within the defined concentration ranges of particular contaminants.

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