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Development and Evaluation of Consensus-Based Sediment Quality Guidelines for Freshwater Ecosystems

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Abstract. Numerical sediment quality guidelines (SQGs) for freshwater ecosystems have previously been developed using a variety of approaches. Each approach has certain advantages and limitations which influence their application in the sediment quality assessment process. In an effort to focus on the agreement among these various published SQGs, consensus-based SQGs were developed for 28 chemicals of concern in freshwater sediments (*i.e.*, metals, polycyclic aromatic hydrocarbons, polychlorinated biphenyls, and pesticides). For each contaminant of concern, two SQGs were developed from the published SQGs, including a threshold effect concentration (TEC) and a probable effect concentration (PEC). The resultant SQGs for each chemical were evaluated for reliability using matching sediment chemistry and toxicity data from field studies conducted throughout the United States. The results of this evaluation indicated that most of the TECs (*i.e.*, 21 of 28) provide an accurate basis for predicting the absence of sediment toxicity. Similarly, most of the PECs (*i.e.*, 16 of 28) provide an accurate basis for predicting sediment toxicity. Mean PEC quotients were calculated to evaluate the combined effects of multiple contaminants in sediment. Results of the evaluation indicate that the incidence of toxicity is highly correlated to the mean PEC quotient ($R^2 = 0.98$ for 347 samples). It was concluded that the consensus-based SQGs provide a reliable basis for assessing sediment quality conditions in freshwater ecosystems.

Numerical sediment quality guidelines (SQGs; including sediment quality criteria, sediment quality objectives, and sediment quality standards) have been developed by various federal, state, and provincial agencies in North America for both freshwater and marine ecosystems. Such SQGs have been used in numerous applications, including designing monitoring programs, interpreting historical data, evaluating the need for detailed sediment quality assessments, assessing the quality of prospective dredged materials, conducting remedial investigations and ecological risk assessments, and developing sediment quality remediation objectives (Long and MacDonald 1998). Numerical SQGs have also been used by many scientists and managers to identify contaminants of concern in aquatic ecosystems and to rank areas of concern on a regional or national basis (*e.g.*, US EPA 1997a). It is apparent, therefore, that numerical SQGs, when used in combination with other tools, such as sediment toxicity tests, represent a useful approach for assessing the quality of freshwater and marine sediments (MacDonald *et al.* 1992; US EPA 1992, 1996a, 1997a; Adams *et al.* 1992; Ingersoll *et al.* 1996, 1997).

The SQGs that are currently being used in North America have been developed using a variety of approaches. The approaches that have been selected by individual jurisdictions depend on the receptors that are to be considered (*e.g.*, sediment-dwelling organisms, wildlife, or humans), the degree of protection that is to be afforded, the geographic area to which the values are intended to apply (*e.g.*, site-specific, regional, or national), and their intended uses (*e.g.*, screening tools, remediation objectives, identifying toxic and not-toxic samples, bioaccumulation assessment). Guidelines for assessing sediment quality relative to the potential for adverse effects on sediment-dwelling organisms in freshwater systems have been derived using a combination of theoretical and empirical approaches, primarily including the equilibrium partitioning approach (EqPA; Di Toro *et al.* 1991; NYSDEC 1994; US EPA 1997a), screening level concentration approach (SLCA; Persaud *et al.* 1993), effects range approach (ERA; Long and Morgan 1991; Ingersoll *et al.* 1996), effects level approach (ELA; Smith *et al.* 1996; Ingersoll *et al.* 1996), and apparent effects threshold approach (AETA; Cubbage *et al.* 1997). Application of these methods has resulted in the derivation of numerical SQGs for many chemicals of potential concern in freshwater sediments.

Selection of the most appropriate SQGs for specific applications can be a daunting task for sediment assessors. This task is particularly challenging because limited guidance is currently available on the recommended uses of the various SQGs. In addition, the numerical SQGs for any particular substance can differ by several orders of magnitude, depending on the derivation procedure and intended use. The SQG selection process is further complicated due to uncertainties regarding

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the bioavailability of sediment-associated contaminants, the effects of covarying chemicals and chemical mixtures, and the ecological relevance of the guidelines (MacDonald *et al.* 2000). It is not surprising, therefore, that controversies have occurred over the proper use of these sediment quality assessment tools.

This paper represents the third in a series that is intended to address some of the difficulties associated with the assessment of sediment quality conditions using various numerical SQGs. The first paper was focused on resolving the "mixture paradox" that is associated with the application of empirically derived SQGs for individual PAHs. In this case, the paradox was resolved by developing consensus SQGs for ΣPAHs (*i.e.*, total PAHs; Swartz 1999). The second paper was directed at the development and evaluation of consensus-based sediment effect concentrations for total PCBs, which provided a basis for resolving a similar mixture paradox for that group of contaminants using empirically derived SQGs (MacDonald *et al.* 2000). The results of these investigations demonstrated that consensus-based SQGs provide a unifying synthesis of the existing guidelines, reflect causal rather than correlative effects, and account for the effects of contaminant mixtures in sediment (Swartz 1999).

The purpose of this third paper is to further address uncertainties associated with the application of numerical SQGs by providing a unifying synthesis of the published SQGs for freshwater sediments. To this end, the published SQGs for 28 chemical substances were assembled and classified into two categories in accordance with their original narrative intent. These published SQGs were then used to develop two consensus-based SQGs for each contaminant, including a threshold effect concentration (TEC; below which adverse effects are not expected to occur) and a probable effect concentration (PEC; above which adverse effects are expected to occur more often than not). An evaluation of resultant consensus-based SQGs was conducted to provide a basis for determining the ability of these tools to predict the presence, absence, and frequency of sediment toxicity in field-collected sediments from various locations across the United States.

Materials and Methods

Derivation of the Consensus-Based SQGs

A stepwise approach was used to develop the consensus-based SQGs for common contaminants of concern in freshwater sediments. As a first step, the published SQGs that have been derived by various investigators for assessing the quality of freshwater sediments were collated. Next, the SQGs obtained from all sources were evaluated to determine their applicability to this study. To facilitate this evaluation, the supporting documentation for each of the SQGs was reviewed. The collated SQGs were further considered for use in this study if: (1) the methods that were used to derive the SQGs were readily apparent; (2) the SQGs were based on empirical data that related contaminant concentrations to harmful effects on sediment-dwelling organisms or were intended to be predictive of effects on sediment-dwelling organisms (*i.e.*, not simply an indicator of background contamination); and (3) the SQGs had been derived on a *de novo* basis (*i.e.*, not simply adopted from another jurisdiction or source). It was not the intent of this paper to collate bioaccumulation-based SQGs.

The SQGs that were expressed on an organic carbon–normalized basis were converted to dry weight–normalized values at 1% organic carbon (MacDonald *et al.* 1994, 1996; US EPA 1997a). The dry weight–normalized SQGs were utilized because the results of previous studies have shown that they predicted sediment toxicity as well or better than organic carbon–normalized SQGs in field-collected sediments (Barrick *et al.* 1988; Long *et al.* 1995; Ingersoll *et al.* 1996; US EPA 1996a; MacDonald 1997).

The effects-based SQGs that met the selection criteria were then grouped to facilitate the derivation of consensus-based SQGs (Swartz 1999). Specifically, the previously published SQGs for the protection of sediment-dwelling organisms in freshwater ecosystems were grouped into two categories according to their original narrative intent, including TECs and PECs. The TECs were intended to identify contaminant concentrations below which harmful effects on sediment-dwelling organisms were not expected. TECs include threshold effect levels (TELs; Smith *et al.* 1996; US EPA 1996a), effect range low values (ERLs; Long and Morgan 1991), lowest effect levels (LELs; Persaud *et al.* 1993), minimal effect thresholds (METs; EC and MENVIQ 1992), and sediment quality advisory levels (SQALs; US EPA 1997a). The PECs were intended to identify contaminant concentrations above which harmful effects on sediment-dwelling organisms were expected to occur frequently (MacDonald *et al.* 1996; Swartz 1999). PECs include probable effect levels (PELs; Smith *et al.* 1996; US EPA 1996a), effect range median values (ERMs; Long and Morgan 1991); severe effect levels (SELs; Persaud *et al.* 1993), and toxic effect thresholds (TETs; EC and MENVIQ 1992; Table 1).

Following classification of the published SQGs, consensus-based TECs were calculated by determining the geometric mean of the SQGs that were included in this category (Table 2). Likewise, consensus-based PECs were calculated by determining the geometric mean of the PEC-type values (Table 3). The geometric mean, rather than the arithmetic mean or median, was calculated because it provides an estimate of central tendency that is not unduly affected by extreme values and because the distributions of the SQGs were not known (MacDonald *et al.* 2000). Consensus-based TECs or PECs were calculated only if three of more published SQGs were available for a chemical substance or group of substances.

Evaluation of the SQGs

The consensus-based SQGs were critically evaluated to determine if they would provide effective tools for assessing sediment quality conditions in freshwater ecosystems. Specifically, the reliability of the individual or combined consensus-based TECs and PECs for assessing sediment quality conditions was evaluated by determining their predictive ability. In this study, predictive ability is defined as the ability of the various SQGs to correctly classify field-collected sediments as toxic or not toxic, based on the measured concentrations of chemical contaminants. The predictive ability of the SQGs was evaluated using a three-step process.

In the first step of the SQG evaluation process, matching sediment chemistry and biological effects data were compiled for various freshwater locations in the United States. Because the data sets were generated for a wide variety of purposes, each study was evaluated to assure the quality of the data used for evaluating the predictive ability of the SQGs (Long *et al.* 1998; Ingersoll and MacDonald 1999). As a result of this evaluation, data from the following freshwater locations were identified for use in this paper: Grand Calumet River and Indiana Harbor Canal, IN (Hoke *et al.* 1993; Giesy *et al.* 1993; Burton 1994; Dorkin 1994); Indiana Harbor, IN (US EPA 1993a, 1996a, 1996b); Buffalo River, NY (US EPA 1993c, 1996a); Saginaw River, MI (US EPA 1993b, 1996a); Clark Fork River, MT (USFWS 1993); Milltown Reservoir, MT (USFWS 1993); Lower Columbia River, WA (Johnson and Norton 1988); Lower Fox River and Green Bay, WI (Call *et al.* 1991); Potomac River, DC (Schlekat *et al.* 1994; Wade *et al.* 1994; Velinsky *et al.* 1994); Trinity River, TX (Dickson *et al.* 1989; US EPA 1996a); Upper Mississippi River, MN to MO (US EPA 1996a, 1997b);

Table 1. Descriptions of the published freshwater SQGs that have been developed using various approaches

Type of SQG	Acronym	Approach	Description	Reference
Threshold effect concentration SQGs				
Lowest effect level	LEL	SLCA	Sediments are considered to be clean to marginally polluted. No effects on the majority of sediment-dwelling organisms are expected below this concentration.	Persaud *et al.* (1993)
Threshold effect level	TEL	WEA	Represents the concentration below which adverse effects are expected to occur only rarely.	Smith *et al.* (1996)
Effect range—low	ERL	WEA	Represents the chemical concentration below which adverse effects would be rarely observed.	Long and Morgan (1991)
Threshold effect level for *Hyalella azteca* in 28-day tests	TEL-HA28	WEA	Represents the concentration below which adverse effects on survival or growth of the amphipod *Hyalella azteca* are expected to occur only rarely (in 28-day tests).	US EPA (1996a); Ingersoll *et al.* (1996)
Minimal effect threshold	MET	SLCA	Sediments are considered to be clean to marginally polluted. No effects on the majority of sediment-dwelling organisms are expected below this concentration.	EC and MENVIQ (1992)
Chronic equilibrium partitioning threshold	SQAL	EqPA	Represents the concentration in sediments that is predicted to be associated with concentrations in the interstitial water below a chronic water quality criterion. Adverse effects on sediment-dwelling organisms are predicted to occur only rarely below this concentration.	Bolton *et al.* (1985); Zarba (1992); US EPA (1997a)
Probable effect concentration SQGs				
Severe effect level	SEL	SLCA	Sediments are considered to be heavily polluted. Adverse effects on the majority of sediment-dwelling organisms are expected when this concentration is exceeded.	Persaud *et al.* (1993)
Probable effect level	PEL	WEA	Represents the concentration above which adverse effects are expected to occur frequently.	Smith *et al.* (1996)
Effect range—median	ERM	WEA	Represents the chemical concentration above which adverse effects would frequently occur.	Long and Morgan (1991)
Probable effect level for *Hyalella azteca* in 28-day tests	PEL-HA28	WEA	Represents the concentration above which adverse effects on survival or growth of the amphipod *Hyalella azteca* are expected to occur frequently (in 28-day tests).	US EPA (1996a); Ingersoll *et al.* (1996)
Toxic effect threshold	TET	SLCA	Sediments are considered to be heavily polluted. Adverse effects on sediment-dwelling organisms are expected when this concentration is exceeded.	EC and MENVIQ (1992)

and Waukegan Harbor, IL (US EPA 1996a; Kemble *et al.* 1999). These studies provided 17 data sets (347 sediment samples) with which to evaluate the predictive ability of the SQGs. These studies also represented a broad range in both sediment toxicity and contamination; roughly 50% of these samples were found to be toxic based on the results of the various toxicity tests (the raw data from these studies are summarized in Ingersoll and MacDonald 1999).

In the second step of the evaluation, the measured concentration of each substance in each sediment sample was compared to the corresponding SQG for that substance. Sediment samples were predicted to be not toxic if the measured concentrations of a chemical substance were lower than the corresponding TEC. Similarly, samples were predicted to be toxic if the corresponding PECs were exceeded in field-collected sediments. Samples with contaminant concentrations between the TEC and PEC were neither predicted to be toxic nor nontoxic (*i.e.*, the individual SQGs are not intended to provide guidance within this range of concentrations). The comparisons of measured concentrations to the SQGs were conducted for each of the 28 chemicals of concern for which SQGs were developed.

In the third step of the evaluation, the accuracy of each prediction

Table 2. Sediment quality guidelines for metals in freshwater ecosystems that reflect TECs (*i.e.*, below which harmful effects are unlikely to be observed)

Substance	Threshold Effect Concentrations						
	TEL	LEL	MET	ERL	TEL-HA28	SQAL	Consensus-Based TEC
Metals (in mg/kg DW)							
Arsenic	5.9	6	7	33	11	NG	9.79
Cadmium	0.596	0.6	0.9	5	0.58	NG	0.99
Chromium	37.3	26	55	80	36	NG	43.4
Copper	35.7	16	28	70	28	NG	31.6
Lead	35	31	42	35	37	NG	35.8
Mercury	0.174	0.2	0.2	0.15	NG	NG	0.18
Nickel	18	16	35	30	20	NG	22.7
Zinc	123	120	150	120	98	NG	121
Polycyclic aromatic hydrocarbons (in µg/kg DW)							
Anthracene	NG	220	NG	85	10	NG	57.2
Fluorene	NG	190	NG	35	10	540	77.4
Naphthalene	NG	NG	400	340	15	470	176
Phenanthrene	41.9	560	400	225	19	1,800	204
Benz[a]anthracene	31.7	320	400	230	16	NG	108
Benzo(a)pyrene	31.9	370	500	400	32	NG	150
Chrysene	57.1	340	600	400	27	NG	166
Dibenz[a,h]anthracene	NG	60	NG	60	10	NG	33.0
Fluoranthene	111	750	600	600	31	6,200	423
Pyrene	53	490	700	350	44	NG	195
Total PAHs	NG	4,000	NG	4,000	260	NG	1,610
Polychlorinated biphenyls (in µg/kg DW)							
Total PCBs	34.1	70	200	50	32	NG	59.8
Organochlorine pesticides (in µg/kg DW)							
Chlordane	4.5	7	7	0.5	NG	NG	3.24
Dieldrin	2.85	2	2	0.02	NG	110	1.90
Sum DDD	3.54	8	10	2	NG	NG	4.88
Sum DDE	1.42	5	7	2	NG	NG	3.16
Sum DDT	NG	8	9	1	NG	NG	4.16
Total DDTs	7	7	NG	3	NG	NG	5.28
Endrin	2.67	3	8	0.02	NG	42	2.22
Heptachlor epoxide	0.6	5	5	NG	NG	NG	2.47
Lindane (gamma-BHC)	0.94	3	3	NG	NG	3.7	2.37

TEL = Threshold effect level; dry weight (Smith *et al.* 1996)
LEL = Lowest effect level, dry weight (Persaud *et al.* 1993) ONTARIO
MET = Minimal effect threshold; dry weight (EC and MENVIQ 1992)
ERL = Effect range low; dry weight (Long and Morgan 1991)
TEL-HA28 = Threshold effect level for Hyalella azteca; 28 day test; dry weight (US EPA 1996a)
SQAL = Sediment quality advisory levels; dry weight at 1% OC (US EPA 1997a)
NG = No guideline

was evaluated by determining if the sediment sample actually was toxic to one or more aquatic organisms, as indicated by the results of various sediment toxicity tests (Ingersoll and MacDonald 1999). The following responses of aquatic organisms to contaminant challenges (*i.e.*, toxicity test endpoints) were used as indicators of toxicity in this assessment (*i.e.*, sediment samples were designated as toxic if one or more of the following endpoints were significantly different from the responses observed in reference or control sediments), including amphipod (*Hyalella azteca*) survival, growth, or reproduction; mayfly (*Hexagenia limbata*) survival or growth; midge (*Chironomus tentans* or *Chironomus riparius*) survival or growth; midge deformities; oligochaete (*Lumbriculus variegatus*) survival; daphnid (*Ceriodaphnia dubia*) survival; and bacterial (*Photobacterium phosphoreum*) luminescence (*i.e.*, Microtox). In contrast, sediment samples were designated as nontoxic if they did not cause a significant response in at least one of these test endpoints. In this study, predictive ability was calculated as the ratio of the number of samples that were correctly classified as toxic or nontoxic to the total number of samples that were predicted to be toxic or nontoxic using the various SQGs (predictive ability was expressed as a percentage).

The criteria for evaluating the reliability of the consensus-based PECs were adapted from Long *et al.* (1998). These criteria are intended to reflect the narrative intent of each type of SQG (*i.e.*, sediment toxicity should be observed only rarely below the TEC and should be frequently observed above the PEC). Specifically, the individual TECs were considered to provide a reliable basis for assessing the quality of freshwater sediments if more than 75% of the sediment samples were correctly predicted to be not toxic. Similarly, the individual PEC for each substance was considered to be reliable if greater than 75% of the sediment samples were correctly predicted to toxic using the PEC. Therefore, the target levels of both false positives (*i.e.*, samples incorrectly classified as toxic) and false negatives (*i.e.*, samples incorrectly classified as not toxic) was 25% using the TEC and PEC. To assure that the results of the predictive ability evaluation were

Table 3. Sediment quality guidelines for metals in freshwater ecosystems that reflect PECs (*i.e.*, above which harmful effects are likely to be observed)

	Probable Effect Concentrations					
Substance	PEL	SEL	TET	ERM	PEL-HA28	Consensus-Based PEC
Metals (in mg/kg DW)						
Arsenic	17	33	17	85	48	33.0
Cadmium	3.53	10	3	9	3.2	4.98
Chromium	90	110	100	145	120	111
Copper	197	110	86	390	100	149
Lead	91.3	250	170	110	82	128
Mercury	0.486	2	1	1.3	NG	1.06
Nickel	36	75	61	50	33	48.6
Zinc	315	820	540	270	540	459
Polycyclic aromatic hydrocarbons (in µg/kg DW)						
Anthracene	NG	3,700	NG	960	170	845
Fluorene	NG	1,600	NG	640	150	536
Naphthalene	NG	NG	600	2,100	140	561
Phenanthrene	515	9,500	800	1,380	410	1,170
Benz[a]anthracene	385	14,800	500	1,600	280	1,050
Benzo(a)pyrene	782	14,400	700	2,500	320	1,450
Chrysene	862	4,600	800	2,800	410	1,290
Fluoranthene	2,355	10,200	2,000	3,600	320	2,230
Pyrene	875	8,500	1,000	2,200	490	1,520
Total PAHs	NG	100,000	NG	35,000	3,400	22,800
Polychlorinated biphenyls (in µg/kg DW)						
Total PCBs	277	5,300	1,000	400	240	676
Organochlorine pesticides (in µg/kg DW)						
Chlordane	8.9	60	30	6	NG	17.6
Dieldrin	6.67	910	300	8	NG	61.8
Sum DDD	8.51	60	60	20	NG	28.0
Sum DDE	6.75	190	50	15	NG	31.3
Sum DDT	NG	710	50	7	NG	62.9
Total DDTs	4,450	120	NG	350	NG	572
Endrin	62.4	1,300	500	45	NG	207
Heptachlor Epoxide	2.74	50	30	NG	NG	16.0
Lindane (gamma-BHC)	1.38	10	9	NG	NG	4.99

PEL = Probable effect level; dry weight (Smith *et al.* 1996) Canada
SEL = Severe effect level, dry weight (Persaud *et al.* 1993) ONTARIO
TET = Toxic effect threshold; dry weight (EC and MENVIQ 1992)
ERM = Effect range median; dry weight (Long and Morgan 1991)
PEL-HA28 = Probable effect level for *Hyalella azteca*; 28-day test; dry weight (US EPA 1996a)
NG = No guideline

not unduly influenced by the number of sediment samples available to conduct the evaluation of predictive ability, the various SQGs were considered to be reliable only if a minimum of 20 samples were included in the predictive ability evaluation (CCME 1995).

The initial evaluation of predictive ability was focused on determining the ability of each SQG when applied alone to classify samples correctly as toxic or nontoxic. Because field-collected sediments typically contain complex mixtures of contaminants, the predictability of these sediment quality assessment tools is likely to increase when the SQGs are used together to classify these sediments. For this reason, a second evaluation of the predictive ability of the SQGs was conducted to determine the incidence of effects above and below various mean PEC quotients (*i.e.*, 0.1, 0.5, 1.0, and 1.5). In this evaluation, mean PEC quotients were calculated using the methods of Long *et al.* (1998; *i.e.*, for each sediment sample, the average of the ratios of the concentration of each contaminant to its corresponding PEC was calculated for each sample), with only the PECs that were found to be reliable used in these calculations. The PEC for total PAHs (*i.e.*, instead of the PECs for the individual PAHs) was used in the calculation to avoid double counting of the PAH concentration data.

Results and Discussion

Derivation of Consensus-Based SQGs

A variety of approaches have been developed to support the derivation of numerical SQGs for the protection of sediment-dwelling organisms in the United States and Canada. MacDonald (1994), Ingersoll and MacDonald (1999), and MacDonald *et al.* (2000) provided reviews of the various approaches to SQG development, including descriptions of the derivation methods, the advantages and limitations of the resultant SQGs, and their recommended uses. This information, along with the supporting documentation that was obtained with the published SQGs, was used to evaluate the relevance of the various SQGs in this investigation.

Subsequently, the narrative descriptions of the various SQGs were used to classify the SQGs into appropriate categories (*i.e.*, TECs or PECs; Table 1). The results of this classification process indicated that six sets of SQGs were appropriate for deriving consensus-based TECs for the contaminants of concern in freshwater sediments, including: (1) TELs (Smith *et al.* 1996); (2) LELs (Persaud *et al.* 1993); (3) METs (EC and MENVIQ 1992); (4) ERLs (Long and Morgan 1991); (5) TELs for *H. azteca* in 28-day toxicity tests (US EPA 1996a; Ingersoll *et al.* 1996); and (6) SQALs (US EPA 1997a).

Several other SQGs were also considered for deriving consensus TECs, but they were not included for the following reasons. First, none of the SQGs that have been developed using data on the effects on sediment-associated contaminants in marine sediments only were used to derive TECs. However, the ERLs that were derived using both freshwater and marine data were included (*i.e.*, Long and Morgan 1991). Second, the ERLs that were developed by the US EPA (1996a) were not utilized because they were developed from the same data that were used to derive the TELs (*i.e.*, from several areas of concern in the Great Lakes). In addition, simultaneously extracted metals–acid volatile sulfide (SEM-AVS)–based SQGs were not used because they could not be applied without simultaneous measurements of SEM and AVS concentrations (Di Toro *et al.* 1990). None of the SQGs that were derived using the sediment background approach were used because they were not effects-based. Finally, no bioaccumulation-based SQGs were used to calculate the consensus-based TECs. The published SQGs that corresponded to TECs for metals, PAHs, PCBs, and organochlorine pesticides are presented in Table 2.

Based on the results of the initial evaluation, five sets of SQGs were determined to be appropriate for calculating consensus-based PECs for the contaminants of concern in freshwater sediments, including: (1) probable effect levels (PELs; Smith *et al.* 1996); (2) severe effect levels (SELs; (Persaud *et al.* 1993); (3) toxic effect thresholds (TETs; EC and MENVIQ 1992); (4) effect range median values (ERMs; Long and Morgan 1991); and (5) PELs for *H. azteca* in 28-day toxicity tests (US EPA 1996a; Ingersoll *et al.* 1996).

While several other SQGs were considered for deriving the consensus-based PECs, they were not included for the following reasons. To maximize the applicability of the resultant guidelines to freshwater systems, none of the SQGs that were developed for assessing the quality of marine sediments were used to derive the freshwater PECs. As was the case for the TECs, the ERMs that were derived using both freshwater and marine data (*i.e.*, Long and Morgan 1991) were included, however. The ERMs that were derived using data from various areas of concern in the Great Lakes (*i.e.*, US EPA 1996a) were not included to avoid duplicate representation of these data in the consensus-based PECs. In addition, none of the SEM-AVS–based SQGs were not used in this evaluation. Furthermore, none of the AET or related values (*e.g.*, NECs from Ingersoll *et al.* 1996; PAETs from Cubbage *et al.* 1997) were used because they were not considered to represent toxicity thresholds (rather, they represent contaminant concentrations above which harmful biological effects always occur). The published SQGs that corresponded to PECs for metals, PAHs, PCBs, and organochlorine pesticides are presented in Table 3.

For each substance, consensus-based TECs or PECs were derived if three or more acceptable SQGs were available. The consensus-based TECs or PECs were determined by calculating the geometric mean of the published SQGs and rounding to three significant digits. Application of these procedures facilitated the derivation of numerical SQGs for a total of 28 chemical substances, including 8 trace metals, 10 individual PAHs and PAH classes, total PCBs, and 9 organochlorine pesticides and degradation products. The consensus-based SQGs that were derived for the contaminants of concern in freshwater ecosystems are presented in Tables 2 and 3.

Predictive Ability of the Consensus-Based SQGs

Matching sediment chemistry and toxicity data from various locations in the United States were used to evaluate the predictive ability of the consensus-based SQGs in freshwater sediments. Within this independent data set, the overall incidence of toxicity was about 50% (*i.e.*, 172 of the 347 samples evaluated in these studies were identified as being toxic to one or more sediment-dwelling organisms). Therefore, 50% of the samples with contaminant concentrations below the TEC, between the TEC and the PEC, and above PECs would be predicted to be toxic if sediment toxicity was unrelated to sediment chemistry (*i.e.*, based on random chance alone).

The consensus-based TECs are intended to identify the concentrations of sediment-associated contaminants below which adverse effects on sediment-dwelling organisms are not expected to occur. Sufficient data were available to evaluate the predictive ability of all 28 consensus-based TECs. Based on the results of this assessment, the incidence of sediment toxicity was generally low at contaminant concentrations below the TECs (Table 4). Except for mercury, the predictive ability of the TECs for the trace metals ranged from 72% for chromium to 82% for copper, lead, and zinc. The predictive ability of the TECs for PAHs was similar to that for the trace metals, ranging from 71% to 83%. Among the organochlorine pesticides, the predictive ability of the TECs was highest for chlordane (85%) and lowest for endrin (71%). At 89%, the predictive ability of the TEC for total PCBs was the highest observed among the 28 substances for which SQGs were derived. Overall, the TECs for 21 substances, including four trace metals, eight individual PAHs, total PAHs, total PCBs, and seven organochlorine pesticides, were found to predict accurately the absence of toxicity in freshwater sediments (*i.e.*, predictive ability $\geq$75%; $\geq$20 samples below the TEC; Table 4). Therefore, the consensus-based TECs generally provide an accurate basis for predicting the absence of toxicity to sediment-dwelling organisms in freshwater sediments.

In contrast to the TECs, the consensus-based PECs are intended to define the concentrations of sediment-associated contaminants above which adverse effects on sediment-dwelling organisms are likely to be observed. Sufficient data were available to evaluate the PECs for 17 chemical substances, including 7 trace metals, 6 individual PAHs, total PAHs, total PCBs, and 2 organochlorine pesticides (*i.e.*, $\geq$20 samples predicted to be toxic). The results of the evaluation of predictive ability demonstrate that the PECs for 16 of the 17 substances meet the criteria for predictive ability that

Table 4. Predictive ability of the consensus-based TECs in freshwater sediments

Substance	Number of Samples Evaluated	Number of Samples Predicted to Be Not Toxic	Number of Samples Observed to Be Not Toxic	Percentage of Samples Correctly Predicted to Be Not Toxic
Metals				
Arsenic	150	58	43	74.1
Cadmium	347	102	82	80.4
Chromium	347	132	95	72.0
Copper	347	158	130	82.3
Lead	347	152	124	81.6
Mercury	79	35	12	34.3
Nickel	347	184	133	72.3
Zinc	347	163	133	81.6
Polycyclic aromatic hydrocarbons				
Anthracene	129	75	62	82.7
Fluorene	129	93	66	71.0
Naphthalene	139	85	64	75.3
Phenanthrene	139	79	65	82.3
Benz(a)anthracene	139	76	63	82.9
Benzo(a)pyrene	139	81	66	81.5
Chrysene	139	80	64	80.0
Dibenz(a,h)anthracene	98	77	56	72.7
Fluoranthene	139	96	72	75.0
Pyrene	139	78	62	79.5
Total PAHs	167	81	66	81.5
Polychlorinated biphenyls				
Total PCBs	120	27	24	88.9
Organochlorine pesticides				
Chlordane	193	101	86	85.1
Dieldrin	180	109	91	83.5
Sum DDD	168	101	81	80.2
Sum DDE	180	105	86	81.9
Sum DDT	96	100	77	77.0
Total DDT	110	92	76	82.6
Endrin	170	126	89	70.6
Heptachlor epoxide	138	90	74	82.2
Lindane	180	121	87	71.9

were established in this study (Table 5). Among the seven individual trace metals, the predictive ability of the PECs ranged from 77% for arsenic to 94% for cadmium. The PECs for six individual PAHs and total PAHs were also demonstrated to be reliable, with predictive abilities ranging from 92% to 100%. The predictive ability of the PEC for total PCBs was 82%. While the PEC for Sum DDE was also found to be an accurate predictor of sediment toxicity (*i.e.*, predictive ability of 97%), the predictive ability of the PEC for chlordane was somewhat lower (*i.e.*, 73%). Therefore, the consensus-based PECs for arsenic, cadmium, chromium, copper, lead, nickel, zinc, naphthalene, phenanthrene, benz[a]anthracene, benzo(a)pyrene, chrysene, pyrene, total PAHs, total PCBs, and sum DDE provide an accurate basis for predicting toxicity in freshwater sediments from numerous locations in North America (*i.e.*, predictive ability of ≥75%; Table 5). Insufficient data were available (*i.e.*, fewer than 20 samples predicted to be toxic) to evaluate the PECs for mercury, anthracene, fluorene, fluoranthene, dieldrin, sum DDD, sum DDT, total DDT, endrin, heptachlor epoxide, and lindane (Table 5).

The two types of SQGs define three ranges of concentrations for each chemical substance. It is possible to assess the degree of concordance that exists between chemical concentrations and the incidence of sediment toxicity (Table 6; MacDonald *et al.* 1996) by determining the ratio of toxic samples to the total number of samples within each of these three ranges of concentrations for each substance. The results of this evaluation demonstrate that, for most chemical substances (*i.e.*, 20 of 28), there is a consistent and marked increase in the incidence of toxicity to sediment-dwelling organisms with increasing chemical concentrations. For certain substances, such as naphthalene, mercury, chlordane, dieldrin, and sum DDD, a lower PEC may have produced greater concordance between sediment chemistry and the incidence of effects. Insufficient data were available to evaluate the degree of concordance for several substances, such as endrin, heptachlor epoxide, and lindane. The positive correlation between contaminant concentrations and sediment toxicity that was observed increases the degree of confidence that can be placed in the SQGs for most of the substances.

While the SQGs for the individual chemical substances provide reliable tools for assessing sediment quality conditions, predictive ability should be enhanced when used together in assessments of sediment quality. In addition, it would be helpful to consider the magnitude of the exceedances of the SQGs in such assessments. Long *et al.* (1998) developed a procedure for evaluating the biological significance of contaminant mixtures through the application of mean PEC quotients. A three-

Table 5. Predictive ability of the consensus-based PECs in freshwater sediments

Substance	Number of Samples Evaluated	Number of Samples Predicted to Be Toxic	Number of Samples Observed to Be Toxic	Percentage of Samples Correctly Predicted to Be Toxic
Metals				
Arsenic	150	26	20	76.9
Cadmium	347	126	118	93.7
Chromium	347	109	100	91.7
Copper	347	110	101	91.8
Lead	347	125	112	89.6
Mercury	79	4	4	100
Nickel	347	96	87	90.6
Zinc	347	120	108	90.0
Polycyclic aromatic hydrocarbons				
Anthracene	129	13	13	100
Fluorene	129	13	13	100
Naphthalene	139	26	24	92.3
Phenanthrene	139	25	25	100
Benz(a)anthracene	139	20	20	100
Benzo(a)pyrene	139	24	24	100
Chrysene	139	24	23	95.8
Fluoranthene	139	15	15	100
Pyrene	139	28	27	96.4
Total PAHs	167	20	20	100
Polychlorinated biphenyls				
Total PCBs	120	51	42	82.3
Organochlorine pesticides				
Chlordane	193	37	27	73.0
Dieldrin	180	10	10	100
Sum DDD	168	6	5	83.3
Sum DDE	180	30	29	96.7
Sum DDT	96	12	11	91.7
Total DDT	110	10	10	100
Endrin	170	0	0	NA
Heptachlor epoxide	138	8	3	37.5
Lindane	180	17	14	82.4

NA = Not applicable

step process is used in the present study to calculate mean PEC quotients. In the first step, the concentration of each substance in each sediment sample is divided by its respective consensus-based PEC. PEC quotients are calculated only for those substances for which reliable PECs were available. Subsequently, the sum of the PEC quotients was calculated for each sediment sample by adding the PEC quotients that were determined for each substance; however, only the PECs that were demonstrated to be reliable were used in the calculation. The summed PEC quotients were then normalized to the number of PEC quotients that are calculated for each sediment sample (*i.e.*, to calculate the mean PEC quotient for each sample; Canfield *et al.* 1998; Long *et al.* 1998; Kemble *et al.* 1999). This normalization step is conducted to provide comparable indices of contamination among samples for which different numbers of chemical substances were analyzed.

The predictive ability of the PEC quotients, as calculated using the consensus-based SQGs, was also evaluated using data that were assembled to support the predictive ability assessment for the individual PECs. In this evaluation, sediment samples were predicted to be not toxic if mean PEC quotients were <0.1 or <0.5. In contrast, sediment samples were predicted to be toxic when mean PEC quotients exceeded 0.5, 1.0, or 1.5. The results of this evaluation indicated that the consensus-based SQGs, when used, together provide an accurate basis for predicting the absence of sediment toxicity (Table 7; Figure 1). Sixty-one sediment samples had mean PEC quotients of <0.1; six of these samples were toxic to sediment-dwelling organisms (predictive ability = 90%). Of the 174 samples with mean PEC quotients of < 0.5, only 30 were found to be toxic to sediment-dwelling organisms (predictive ability = 83%; Table 7).

The consensus-based SQGs also provided an accurate basis for predicting sediment toxicity in sediments that contained mixtures of contaminants. Of the 173 sediment samples with mean PEC quotients of > 0.5 (calculated using the PECs for seven trace metals, the PEC for total PAHs [rather than the PECs for individual PAHs], the PEC for PCBs, and the PEC for sum DDE), 147 (85%) were toxic to sediment-dwelling organisms (Table 7; Figure 1). Similarly, 92% of the sediment samples (132 of 143) with mean PEC quotients of > 1.0 were toxic to one or more species of aquatic organisms. Likewise, 94% of the sediment samples (118 of 125) with mean PEC quotients of greater than 1.5 were found to be toxic, based on the results of various freshwater toxicity tests. Therefore, it is apparent that a mean PEC quotient of 0.5 represents a useful

Table 6. Incidence of toxicity within ranges of contaminant concentrations defined by the SQGs

Substance	Number of Samples Evaluated	Incidence of Toxicity (%, number of samples in parentheses)		
		≤TEC Not toxic	TEC-PEC	> PEC Toxic
Metals				
Arsenic	150	25.9% (15 of 58)	57.6% (38 of 66)	76.9% (20 of 26)
Cadmium	347	19.6% (20 of 102)	44.6% (29 of 65)	93.7% (118 of 126)
Chromium	347	28% (37 of 132)	64.4% (38 of 59)	91.7% (100 of 109)
Copper	347	17.7% (28 of 158)	64.0% (48 of 75)	91.8% (101 of 110)
Lead	347	18.4% (28 of 152)	53.6% (37 of 69)	89.6% (112 of 125)
Mercury	79	65.7% (23 of 35)	70.0% (28 of 40)	100% (4 of 4)
Nickel	347	27.7% (51 of 184)	62.7% (32 of 51)	90.6% (87 of 96)
Zinc	347	18.4% (30 of 163)	60.9% (39 of 64)	90.0% (108 of 120)
Polycyclic aromatic hydrocarbons				
Anthracene	129	17.3% (13 of 75)	92.9% (26 of 28)	100% (13 of 13)
Fluorene	129	29% (27 of 93)	85.7% (12 of 14)	100% (13 of 13)
Naphthalene	139	24.7% (21 of 85)	94.1% (16 of 17)	92.3% (24 of 26)
Phenanthrene	139	17.7% (14 of 79)	88.2% (30 of 34)	100% (25 of 25)
Benz(a)anthracene	139	17.1% (13 of 76)	70% (14 of 20)	100% (20 of 20)
Benzo(a)pyrene	139	18.5% (15 of 81)	75.7% (28 of 37)	100% (24 of 24)
Chrysene	139	20% (16 of 80)	68.1% (32 of 47)	95.8% (23 of 24)
Fluoranthene	139	25% (24 of 96)	82.5% (33 of 40)	100% (15 of 15)
Pyrene	139	20.5% (16 of 78)	63.0% (29 of 46)	96.4% (27 of 28)
Total PAHs	167	18.5% (15 of 81)	65.1% (43 of 66)	100% (20 of 20)
Polychlorinated biphenyls				
Total PCBs	120	11.1% (3 of 27)	31.0% (9 of 29)	82.3% (42 of 51)
Organochlorine pesticides				
Chlordane	193	14.9% (15 of 101)	75.0% (15 of 20)	73.0% (27 of 37)
Dieldrin	180	16.5% (18 of 109)	95.2% (20 of 21)	100% (10 of 10)
Sum DDD	168	19.8% (20 of 101)	33.3% (1 of 3)	83.3% (5 of 6)
Sum DDE	180	18.1% (19 of 105)	33.3% (1 of 3)	96.7% (29 of 30)
Sum DDT	96	23% (23 of 100)	0.0% (0 of 1)	91.7% (11 of 12)
Total DDT	110	17.4% (16 of 92)	100% (23 of 23)	100% (10 of 10)
Endrin	170	29.4% (37 of 126)	40.0% (4 of 10)	NA% (0 of 0)
Heptachlor epoxide	138	17.8% (16 of 90)	85.0% (17 of 20)	37.5% (3 of 8)
Lindane	180	28.1% (34 of 121)	65.9% (29 of 44)	82.4% (14 of 17)

Table 7. Predictive ability of mean PEC quotients in freshwater sediments

Mean PEC Quotient	Mean PEC Quotients Calculated with Total PAHs Predictive Ability (%)	Mean PEC Quotients Calculated with Individual PAH Predictive Abilities (%)
<0.1	90.2% (61)	90.2% (61)
<0.5	82.8% (174)	82.9% (175)
>0.5	85% (173)	85.4% (172)
>1.0	93.3% (143)	93.4% (143)
>1.5	94.4% (125)	95% (121)

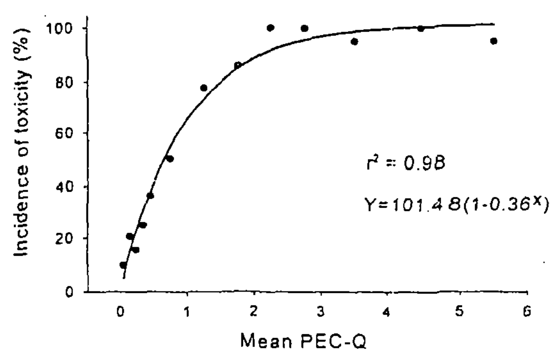

Fig. 1. Relationship between mean PEC quotient and incidence of toxicity in freshwater sediments

threshold that can be used to accurately classify sediment samples as both toxic and not toxic. The results of this evaluation were not substantially different when the PECs for the individuals PAHs (*i.e.*, instead of the PEC for total PAHs) were used to calculate the mean PEC quotients (Table 7). Kemble *et al.* (1999) reported similar results when the mean PEC quotients were evaluated using the results of only 28-day toxicity tests with *H. azteca* (n = 149, 32% of the samples were toxic).

To examine further the relationship between the degree of chemical contamination and probability of observing toxicity in freshwater sediments, the incidence of toxicity within various ranges of mean PEC quotients was calculated (*e.g.*, < 0.1, 0.1–0.2, 0.2–0.3). Next, these data were plotted against the midpoint of each range of mean PEC quotients (Figure 1). Subsequent curve-fitting indicated that the mean PEC-quotient is highly correlated with incidence of toxicity ($r^2 = 0.98$), with the relationship being an exponential function. The resultant equation can be used to estimate the probability of observing sediment toxicity at any mean PEC quotient.

Although it is important to be able to predict accurately the presence and absence of toxicity in field-collected sediments, it is also helpful to be able to identify the factors that are causing or substantially contributing to sediment toxicity. Such information enables environmental managers to focus limited resources on the highest-priority sediment quality issues and concerns. In this context, it has been suggested that the results of spiked sediment toxicity tests provide a basis for identifying the concentrations of sediment-associated contaminants that cause sediment toxicity (Swartz *et al.* 1988; Ingersoll *et al.* 1997). Unfortunately, there is limited relevant data available that assesses effects of spiked sediment in freshwater systems. For example, the available data from spiked sediment toxicity tests is limited to just a few of the chemical substances for which reliable PECs are available, primarily copper and fluoranthene. Additionally, differences in spiking procedures, equilibration time, and lighting conditions during exposures confound the interpretation of the results of sediment spiking studies, especially for PAHs (ASTM 1999). Moreover, many sediment spiking studies were conducted to evaluate bioaccumulation using relatively insensitive test organisms (*e.g., Diporeia* and *Lumbriculus*) or in sediments containing mixtures of chemical substances (Landrum *et al.* 1989, 1991).

In spite of the limitations associated with the available dose-response data, the consensus-based PECs for copper and fluoranthene were compared to the results of spiked sediment toxicity tests. Suedel (1995) conducted a series of sediment spiking studies with copper and reported 48-h to 14-day LC_{50} for four freshwater species, including the waterfleas *Ceriodaphnia dubia* (32–129 mg/kg DW) and *Daphnia magna* (37–170 mg/kg DW), the amphipod *H. azteca* (247–424 mg/kg DW), and the midge *C. tentans* (1.026–4,522 mg/kg DW). An earlier study reported 10-day LC_{50}s of copper for *H. azteca* (1,078 mg/kg) and *C. tentans* (857 mg/kg), with somewhat higher effect concentrations observed in different sediment types (Cairns *et al.* 1984). The PEC for copper (149 mg/kg DW) is higher than or comparable to (*i.e.*, within a factor of three; MacDonald *et al.* 1996; Smith *et al.* 1996) the median lethal concentrations for several of these species. For fluoranthene, Suedel and Rodgers (1993) reported 10-day EC_{50}s of 4.2–15.0 mg/kg, 2.3–7.4 mg/kg, and 3.0–8.7 mg/kg for *D. magna, H. azteca*, and *C. tentans*, respectively. The lower of the values reported for each species are comparable to the PEC for fluoranthene that was derived in this study (*i.e.*, 2.23 mg/kg). Much higher toxicity thresholds have been reported in other studies (*e.g.*, Kane Driscoll *et al.* 1997; Kane Driscoll and Landrum 1997), but it is likely that these results were influenced by the lighting conditions under which the tests were conducted. Although this evaluation was made with limited data, the results suggest that the consensus-based SQGs are comparable to the acute toxicity thresholds that have been obtained from spiking studies.

A second approach—to identify concentrations of sediment-associated contaminants that cause or contribute to toxicity—was to compare our consensus-based PECs to equilibrium partitioning values (Swartz 1999; MacDonald *et al.* 1999). The equilibrium partitioning (EqP) approach provides a theoretical basis for deriving sediment quality guidelines for the protection of freshwater organisms (Di Toro *et al.* 1991; Zarba 1992). Using this approach, the US EPA (1997a) developed SQGs that are intended to represent chronic toxicity thresholds for various sediment-associated contaminants, primarily nonionic organic substances. The concentrations of these contaminants are considered to be sufficient to cause or substantially contribute to sediment toxicity when they exceed the EqP-based SQGs (Berry *et al.* 1996). To evaluate the extent to which the consensus-based SQGs are causally based, the PECs were compared to the chronic toxicity thresholds that have been developed previously using the EqP approach (see Table 2). The results of this evaluation indicate that the consensus-based PECs are generally comparable to the EqP-based SQGs (*i.e.*, within a factor of three; MacDonald *et al.* 1996; Smith *et al.* 1996). Therefore, the consensus-based PECs also define concentrations of sediment-associated contaminants that are sufficient to cause or substantially contribute to sediment toxicity.

Summary

Consensus-based SQGs were derived for 28 common chemicals of concern in freshwater sediments. For each chemical substance, two consensus-based SQGs were derived from the published SQGs. These SQGs reflect the toxicity of sediment-associated contaminants when they occur in mixtures with other contaminants. Therefore, these consensus-based SQGs are likely to be directly relevant for assessing freshwater sediments that are influenced by multiple sources of contaminants. The results of the evaluations of predictive ability demonstrate that the TECs and PECs for most of these chemicals, as well as the PEC quotients, provide a reliable basis for classifying sediments as not toxic and toxic. In addition, positive correlations between sediment chemistry and sediment toxicity indicate that many of these sediment-associated contaminants are associated with the effects that were observed in field-collected sediments. Furthermore, the level of agreement between the available dose-response data, the EqP-based SQGs, and the consensus-based SQGs indicates that sediment-associated contaminants are likely to cause or substantially contribute to, as opposed to simply be associated with, sediment toxicity at concentrations above the PECs.

Overall, the results of the various evaluations demonstrate that the consensus-based SQGs provide a unifying synthesis of the existing SQGs, reflect causal rather than correlative effects, and account for the effects of contaminant mixtures (Swartz 1999). As such, the SQGs can be used to identify hot spots with respect to sediment contamination, determine the potential for and spatial extent of injury to sediment-dwelling organisms, evaluate the need for sediment remediation, and support the development of monitoring programs to further assess the extent of contamination and the effects of contaminated sediments on sediment-dwelling organisms. These applications are strengthened when the SQGs are used in combination with other sediment quality assessment tools (*i.e.*, sediment toxicity tests, bioaccumulation assessments, benthic invertebrate community assessments; Ingersoll *et al.* 1997). In these applications, the TECs should be used to identify sediments that are unlikely to be adversely affected by sediment-associated contaminants. In contrast, the PECs should be used to identify sediments that are likely to be toxic to sediment-dwelling organisms. The PEC quotients should be used to assess sediment that contain complex mixtures of chemical contaminants.

The consensus-based SQGs described in this paper do not consider the potential for bioaccumulation in aquatic organisms nor the associated hazards to the species that consume aquatic organisms (*i.e.*, wildlife and humans). Therefore, it is important to use the consensus-based SQGs in conjunction with other tools, such as bioaccumulation-based SQGs, bioaccumulation tests, and tissue residue guidelines, to evaluate more fully the potential effects of sediment-associated contaminants in the environment. Future investigations should focus of evaluating the predictive ability of these sediment assessment tools on a species- and endpoint-specific basis for various geographic areas.

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