

See discussions, stats, and author profiles for this publication at: <https://www.researchgate.net/publication/7677039>

Deriving Sediment Quality Guidelines from Field-Based Species Sensitivity Distributions

ARTICLE *in* ENVIRONMENTAL SCIENCE AND TECHNOLOGY · AUGUST 2005

Impact Factor: 5.33 · DOI: 10.1021/es050450x · Source: PubMed

CITATIONS

63

READS

41

7 AUTHORS, INCLUDING:



Kenneth Mei yee Leung

The University of Hong Kong

149 PUBLICATIONS **2,529** CITATIONS

SEE PROFILE



Wai Keung Li

The University of Hong Kong

170 PUBLICATIONS **4,157** CITATIONS

SEE PROFILE



Gilbert C. S. Lui

The University of Hong Kong

15 PUBLICATIONS **215** CITATIONS

SEE PROFILE



Paul K S Lam

City University of Hong Kong

407 PUBLICATIONS **12,182** CITATIONS

SEE PROFILE

Deriving Sediment Quality Guidelines from Field-Based Species Sensitivity Distributions

KENNETH M. Y. LEUNG,[†]
ANDERS BJØRGESÆTER,[‡] JOHN S. GRAY,[‡]
W. K. LI,[§] GILBERT C. S. LUI,[§]
YUAN WANG,^{||} AND PAUL K. S. LAM^{*.⊥}

Swire Institute of Marine Science, Department of Ecology and Biodiversity, and Department of Statistics and Actuarial Science, The University of Hong Kong, Pokfulam, Hong Kong, People's Republic of China, Institute of Biology, University of Oslo, Oslo, Norway, School of the Environment, Nanjing University, Nanjing, 210093, People's Republic of China, and Department of Biology and Chemistry, City University of Hong Kong, Hong Kong, People's Republic of China

The determination of predicted no-effect concentrations (PNECs) and sediment quality guidelines (SQGs) of toxic chemicals in marine sediment is extremely important in ecological risk assessment. However, current methods of deriving sediment PNECs or threshold effect levels (TELs) are primarily based on laboratory ecotoxicity bioassays that may not be ecologically and environmentally relevant. This study explores the possibility of utilizing field data of benthic communities and contaminant loadings concurrently measured in sediment samples collected from the Norwegian continental shelf to derive SQGs. This unique dataset contains abundance data for ca. 2200 benthic species measured at over 4200 sampling stations, along with co-occurring concentration data for >25 chemical species. Using barium, cadmium, and total polycyclic aromatic hydrocarbons (PAHs) as examples, this paper describes a novel approach that makes use of the above data set for constructing field-based species sensitivity distributions (f-SSDs). Field-based SQGs are then derived based on the f-SSDs and HCx values [hazardous concentration for x% of species or the (100 – x)% protection level] by the nonparametric bootstrap method. Our results for Cd and total PAHs indicate that there are some discrepancies between the SQGs currently in use in various countries and our field-data-derived SQGs. The field-data-derived criteria appear to be more environmentally relevant and realistic. Here, we suggest that the f-SSDs can be directly used as benchmarks for probabilistic risk assessment, while the field-data-derived SQGs can be used as site-specific guidelines or integrated into current SQGs.

* Corresponding author phone: +852-2788-7681; fax: +852-2788-7406; e-mail: bhpksl@cityu.edu.hk.

[†] Swire Institute of Marine Science, Department of Ecology & Biodiversity, The University of Hong Kong.

[‡] University of Oslo.

[§] Department of Statistics and Actuarial Science, The University of Hong Kong.

^{||} Nanjing University.

[⊥] City University of Hong Kong.

Introduction

The determination of sediment quality guidelines (SQGs) of toxic substances in marine sediment is extremely important to ecological risk assessments (1–3). SQGs commonly include different trigger levels such as threshold effects level (TEL), effect range-low (ERL), effect range-median (ERM), and probable effects level (PEL) (4). Each of these values represents a different level of protection or intensity of the probable harmful effect of a toxic substance to benthic marine organisms. For example, the TEL is similar to a predicted no-effect concentration (PNEC), the concentration below which benthic organisms in the sediment are unlikely to be adversely affected. SQGs are important tools devised to conserve biodiversity and community structure in benthic ecosystems, and increasingly it is recognized that environmental protection needs to be implemented in the context of sustainable development; that is, there ought to be a balance between economic growth/social needs and environmental protection (5, 6). On this basis, the derived PNECs used in ecological risk assessment must not lead to underprotection or overprotection (7, 8). In ecological risk assessment, assessors have to consider all uncertainties during various tiers of extrapolation (e.g., from a few species to many species or community, from acute toxicity to chronic toxicity, from laboratory to the field) (8–10). As well, most current effects data are derived from freshwater systems, due to a lack of information on saltwater and marine organisms (11). Consequently, marine PNECs have been derived from freshwater data by applying unrealistically large assessment (or safety) factors (12). In this paper, we argue that the application of assessment factors in the derivation of PNEC from laboratory-generated effects data may be inappropriate.

Most SQGs or sediment PNECs are derived from effects data generated from laboratory ecotoxicity bioassays (8–10). This laboratory-based method, however, has several shortcomings, often resulting in large uncertainties when these effects data are being extrapolated to assessment end points (e.g., PNECs) in the natural environment (7). First, chemicals are usually tested singly in the bioassay, whereas in the environment, chemicals exist in mixtures and may interact, so that estimates of effects of these complex mixtures under field conditions are arguably only partially predictable at best (13). Second, many routine ecotoxicity tests utilize laboratory-culturable species such as the marine amphipods *Ampelisca abdita* and *Leptocheirus plumulosus*, which often do not present in the site of concern and thus may not be particularly ecologically relevant (7, 14). In the standard procedures, constrained budgets and resources lead to very few species (usually <10 in total, from three different trophic levels) being tested for generating the PNEC value of a particular chemical (15, 16). Third, the bioassay is usually conducted under one set of fixed conditions (i.e., preselected salinity, temperature, sediment type, and carbon content). Under realistic field situations, organisms are exposed to a suite of multiple biological and physical conditions and/or stressors, which may vary from time to time according to weather and climatic situations in the field. Indeed, contaminants often interact with natural stressors such as extremes in temperature, salinity, low dissolved oxygen levels, and reduced food availability in the natural environment (17, 18).

Due to such dynamic biological and physicochemical interactions in natural environments, the relationship between the abundance of a species and the concentration of a toxic chemical in sediment may not be a simple negative dose-related response as predicted by traditional bioassays.

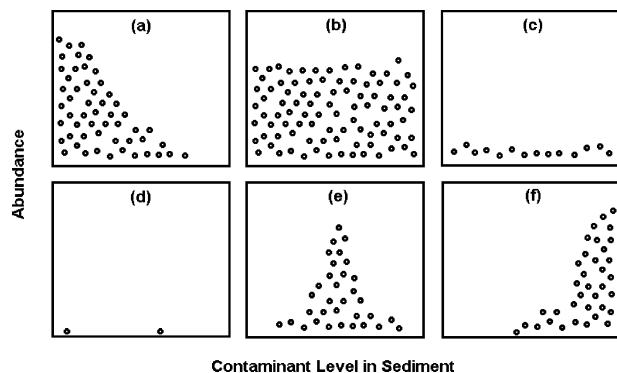


FIGURE 1. Schematic diagrams illustrating various possibilities of benthic marine organisms occurring along an increasing gradient of a contaminant. Each circle represents the abundance of the studied species from an individual sediment sample (i.e., site). A sensitive species (a) will be affected by the contaminant in a dose-response manner, presenting a negative relationship between its abundance and the contaminant level in the sediment. In contrast, an insensitive species (b) exhibits a high abundance regardless of the contaminant level, while the other insensitive species (c) shows a low abundance regardless of the contaminant level. It is common to find rare species (d), which occur only in one or two sampling sites. An opportunistic species (e) increases its abundance when other sensitive species have been killed or have left due to an increased contaminant level, but this opportunistic species subsequently becomes sensitive at higher concentrations. Other highly tolerant opportunistic species (f) may show high abundance even at highly elevated contaminant levels.

Figure 1 hypothesizes that there are at least six different ways for a species to respond to a chemical stressor in the field. Furthermore, it is inconceivable that simple laboratory ecotoxicological bioassays can reflect the importance of biological interactions (e.g., population density effects, competition, and predator-prey relationships), biological attributes (e.g., susceptibility of different life stages and life cycle variability), and seasonality in the natural environment (7). Although it has been suggested that more ecologically relevant tests such as microcosms and mesocosms could be employed to address these issues, they have not been widely adopted because microcosms and mesocosms are much more expensive and logistically difficult to carry out (13).

Therefore, we conclude that the laboratory-based approach has many inherent problems, not least of which are the large uncertainties between the predictions from the laboratory data and actual effects on organisms measured in the field. Most countries have extensive monitoring programs where data are collected on concentrations of contaminants and on the species and their abundances that occur in sediments. It seems sensible to derive guidelines based on data collected under actual field conditions rather than making often incorrect predictions from laboratory data. Over the past two decades, some efforts have been made to apply field data in deriving marine SQGs (e.g., refs 19–22). In general, these field-driven approaches, which primarily derive SQGs based on correlations of biological effects with contaminant concentrations, include the establishment of screening level, apparent effects threshold, and bioeffects/contamination co-occurrence analyses (21). Among them, the screening level approach has been widely used in Canada and the United States. The screening level concentration is determined by use of a database containing co-occurrence information on the concentrations of specific contaminants and the presence/absence data of benthic species in the sediment (19, 23). A cumulative frequency distribution of stations at which a particular species occurred is plotted against the concentration of the target chemical. A species

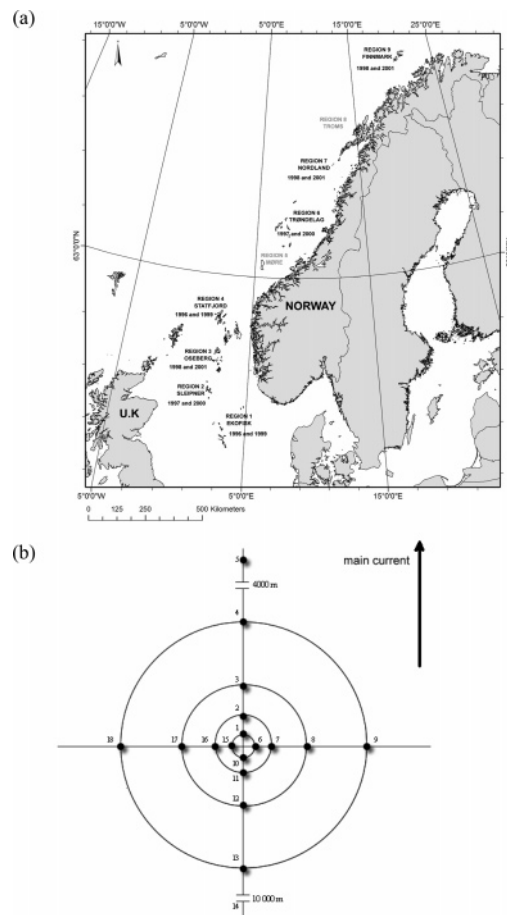


FIGURE 2. Map showing the sampling locations in Norwegian waters (a). The lower schematic diagram (b) illustrates where samples were taken along the two fixed and perpendicular transects at each sampling site.

screening level concentration (SSLC) is then defined as the concentration at the 90th percentile of the frequency distribution. After SLC values are computed for all species found in the sediment, a final cumulative plot of all these SSLC values against contaminant concentrations is constructed. Finally, the screening level of this target chemical is derived as the chemical concentration above which 95% of the SSLCs occurred. However, von Stackelberg and Menzie (24) pointed out that SQGs based on presence/absence data could be invalid, because there are substantial possibilities for misinterpretation due to undersampling of species (i.e., small sample size) and the statistical attributes of log-normal distributions of chemical data. Furthermore, absence of a species from a site cannot be attributed to one single factor, the chemical concentration. Most species are rare in nature (25), and rareness is due to a multitude of factors both natural and anthropogenic. The SSLC approach also fails to consider potential biological interactions.

The problems with current laboratory or field data driven approaches for deriving SQGs can potentially be, at least in part, overcome through establishing field-based species sensitivity distributions (f-SSDs). Provided that data are available, the relationship between the abundance of a particular species and the concentration of a specific chemical in the sediment can be plotted (Figure 1), and any sensitive species is readily revealed (e.g., Figure 1a). In this paper, we will demonstrate how SQGs can be derived by use of the proposed f-SSD approach based on the Norwegian Oil Industry Association (OLF) database. With three chemicals of varying toxicities (barium, cadmium, and total polycyclic

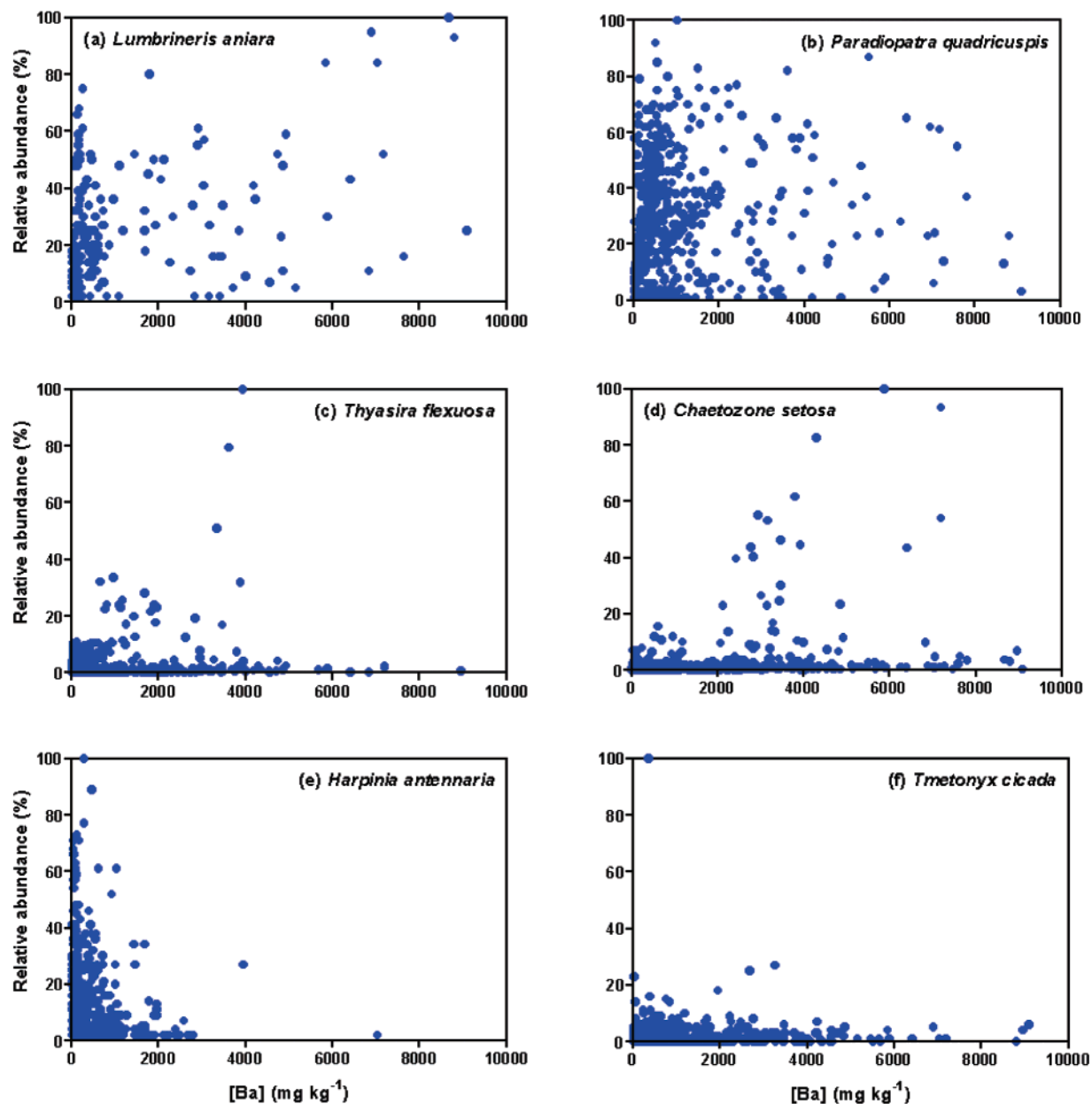


FIGURE 3. Relationships between the relative abundance of six different benthic infaunal species and concentrations of barium in Norwegian marine sediments.

aromatic hydrocarbons) as examples, the SQGs derived by the proposed f-SSD approach will be compared with SQGs currently in use in North American, European, and Asian/Pacific countries. The usefulness and problems of this new approach in deriving site-specific SQGs will be discussed.

Materials and Methods

OLF Database. This study utilized field data in the Norwegian Oil Industry Association (OLF) database that were collected between 1996 and 2001 (Figure 2a). The period included 14 large regional environmental monitoring surveys and several smaller baseline surveys. The database contains abundance data for 1944 species and 1 822 001 individuals in 1902 sampling stations and includes all zones around the oil platforms, that is, highly affected, transitional, and non-affected zones (26, 27). The average depth across all sampling stations was 185 m (± 3.0 m; SEM). Environmental monitoring surveys are conducted in regular intervals of 3 years; thus most of the sampling stations were sampled twice in this period (Figure 2a).

Surveys were conducted according to directives prepared by the Norwegian State Pollution Control Authority (28, 29) and carried out with essentially the same design and procedures throughout the whole period. The basic sampling design is a cross that intersects at the center of the oil platform with sampling stations placed at increasing distances from the oil platform (Figure 2b). At each sampling station, sediment samples were taken with a 0.1 m² van Veen grab, five grab samples for biological analyses and three for chemical analyses. Five additional grab samples were taken at the reference stations (situated in nonaffected zones) for biological analyses and two additional grab samples were taken for additional chemical analyses. The biological samples were washed through a 1 mm round mesh sieve and the material was fixed in formalin. For analysis of metals, subsamples were taken from the upper sediment layer (0–1 cm), dried, homogenized, sieved, and extracted from the sediment by nitric acid digestion (30). For hydrocarbon analysis, subsamples were frozen and extracted from the sediment by saponification followed by solvent extraction.

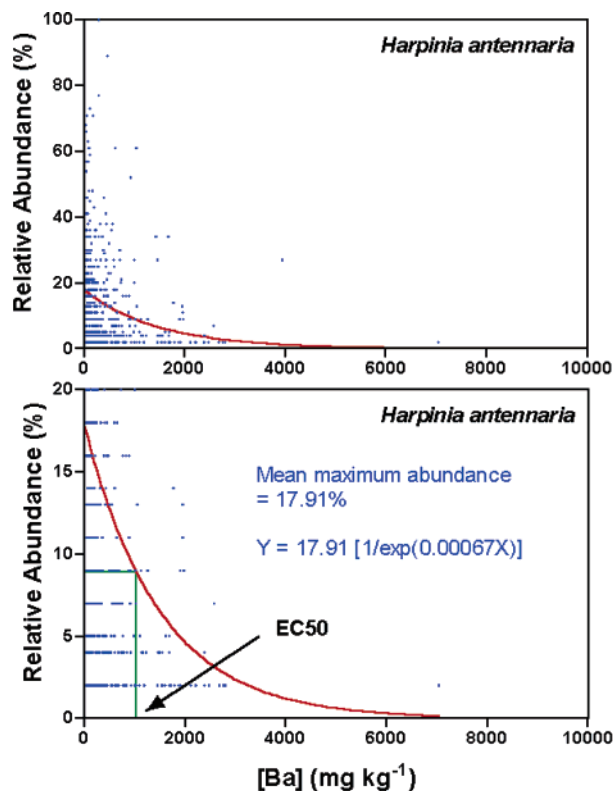


FIGURE 4. Example showing how the EC_{50} is obtained for a species based on normalized relative abundance. First, a best regression model is applied to fit all relative abundance data (upper panel); the regression model can be either nonlinear or linear depending on the goodness of fit. Second, an average maximum of the relative abundance (i.e., interception of the curve) is used as the EC_{100} and then the EC_{50} can be computed on the basis of the regression model.

Barium (Ba) concentrations were determined by inductive coupled plasma–atomic emission spectrometry (ICP-AES), and cadmium (Cd) concentrations by atomic absorption spectrometry (AAS), while total polycyclic aromatic hydrocarbons (PAHs) concentrations were analyzed by gas chromatography/mass spectrometry (GC/MS). Quality assurance procedures were mandatory, and all analyses were carried out by laboratories accredited according to ISO/IEC/EN 17025, formerly ISO Guide 25 and EN45001 (31).

Data Mining. Three substances (Ba, Cd, and PAHs) of varying toxicities were selected to evaluate the proposed field-species sensitivity distribution (f-SSD) approach herein. Ba has low toxicity, while both Cd and PAHs are more toxic to marine benthic organisms. For each chemical, scatter plots between abundance of individual benthic species and chemical concentrations in the sediment were constructed to illustrate species sensitivity and to determine the effect concentration. The species were selected from the OLF database if they fulfilled the following two criteria. First, the species must be found in more than 30 samples with different concentrations to ensure sufficient statistical power. Second, the abundance of the selected species must be greater than 100 individuals to ensure adequate resolution (i.e., percent relative abundance) on the y-axis of the scatter diagram. On the basis of the above criteria, 614, 519, and 239 species for Ba, Cd, and total PAH, respectively, were selected from the OLF database. Moreover, background baseline concentrations of these three substances in the sediment of reference sites were also determined.

Construction of f-SSD. By way of example, typical responses of organisms to Ba are shown in Figure 3. Some species are insensitive (Figure 3a,b; the polychaetes *Lum-*

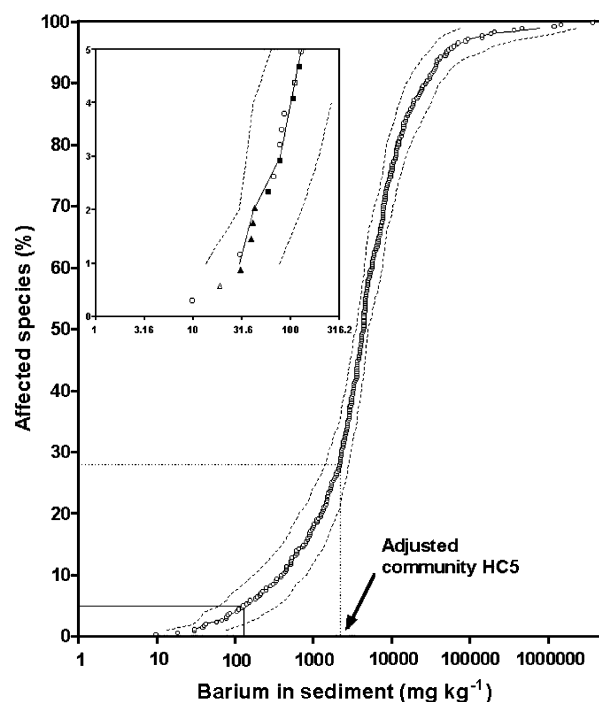


FIGURE 5. Field-based species sensitivity distribution for barium, which is modeled by the basic bootstrap method ($B = 5000$): the solid line shows the median estimate (50%) percentile with lower (5%) and upper (95%) percentiles (dashed lines), which constitutes a 90% two-tailed percentile confidence interval. The inset amplifies the lower percentile (HC0–HC5) region; the different symbols denote different taxonomic groups: Crustacea (■), Mollusca (▲), Polychaeta (○), Echinodermata (□), and Platyhelminthes (△).

brineris aniara and *Paradiopatra quadricuspis*); some species increase and then decrease in abundance in response to increasing concentration of Ba (Figure 3c,d; the bivalve *Thyasira flexuosa* and the polychaete *Chaetozona setosa*), but *T. flexuosa* is more sensitive; some species are sensitive, decreasing in abundance with increasing concentrations of Ba (Figure 3e,f; the crustacean *Harpinia antennaria* and the polychaete *Tmetonyx cicada*). In total, there were 343, 318, and 139 sensitive species identified for Ba, Cd, and total PAHs, respectively. It is here assumed that if these sensitive species are protected, the other species will probably be protected as well. Next, an effect concentration (EC_{50}) corresponding to 50% reduction in relative abundance was computed by use of either a linear or a nonlinear regression model. The choice of the final model was dependent on the goodness of fit and coefficient of determination (r^2). The EC_{50} was selected because it is the estimate with the smallest confidence intervals based on the regression models. Although EC_{10} is closer to the no observable effect concentration (NOEC) and has been used elsewhere in laboratory ecotoxicity tests (9, 10, 32), a reliable EC_{10} value cannot be determined from the scatter plot in this study due to relatively large variances at both ends of the regression models.

For *H. antennaria*, the estimated Ba EC_{50} was 1034 mg kg^{-1} (Figure 4). For each chemical, a f-SSD was constructed by plotting the cumulative EC_{50} values against the rank of these values [$i/(1 + N)$] (16, 32, 33). Various effect levels (i.e., hazardous concentration x (HC_x) values for $x\%$ of species or $100 - x\%$ protection), were subsequently determined by the nonparametric bootstrap method ($B = 5000$) as described by Grist et al. (34).

Effect Levels for Protection. As the f-SSD represents the sensitivity distribution of sensitive species, its HC5 value is likely to be overprotective if this sensitive species HC5 value is used to set the protection level for the community in the

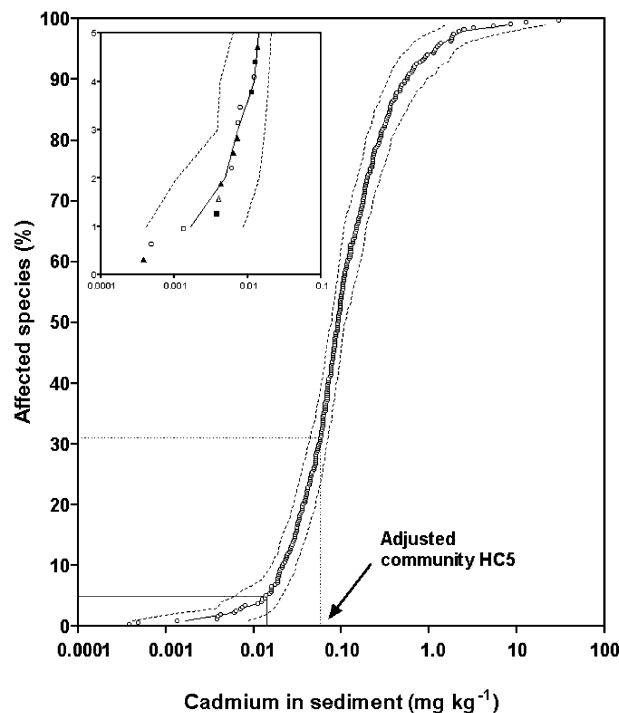


FIGURE 6. Field-based species sensitivity distribution for cadmium. Format is the same as in Figure 5.

natural environment. To counter this problem, we used the following equation to compute the level of x of HC x that would, in theory, protect 95% of the species in the community (i.e., adjusted community HC5):

$$x\% = 5\%(A/S) \quad (1)$$

where A is the total number of species selected from the database ($N = 1943$, 1943 , and 1397 for Ba, Cd, and total PAHs respectively), while S is the total number of sensitive species ($N = 343$, 318 , and 139 , respectively). For the adjusted community HC5 or concentration according 95% protection, the estimated x values for Ba, Cd, and total PAHs were 28%, 31%, and 50%, respectively. These HC x values were selected as the TELs. For probable effect levels (PELs), adjusted community HC10 values from the f-SSDs (i.e., HC56, HC62, and HC99 for Ba, Cd, and total PAHs, correspondingly) were used because they would numerically represent a doubling of the effect at the adjusted community HC5 level in the field. Both field-data-derived TELs and PELs were compared with current SQGs from Australia, Canada, Europe, Hong Kong, and the United States.

Results

f-SSD. Field-based species sensitivity distributions (f-SSD) for Ba, Cd, and total PAHs were constructed and are shown in Figures 5–7. As expected, low toxicities exhibited by Ba were evident from the f-SSD (Figure 5). The sensitive species HC5 of Ba was 130.1 mg kg^{-1} (Table 1). Polychaeta were the most sensitive taxon to Ba, followed by Platyhelminthes and Mollusca (Figure 5, inset; Table 2). In contrast to Ba, Cd was more toxic to the benthic organisms as revealed by the f-SSD, with a sensitive species HC5 value of 0.014 mg kg^{-1} (Figure 6; Table 1). The most sensitive taxonomic groups to Cd (between HC0–HC5 in the f-SSD) included Mollusca, Polychaeta, Echinodermata, Crustacea, and Platyhelminthes (Figure 6, inset; Table 2). As with Cd, total PAHs were more toxic to the benthic organisms as compared to Ba, with a sensitive species HC5 of 0.020 mg kg^{-1} (Table 1). Mollusca is the most sensitive taxonomic group to PAHs, followed by

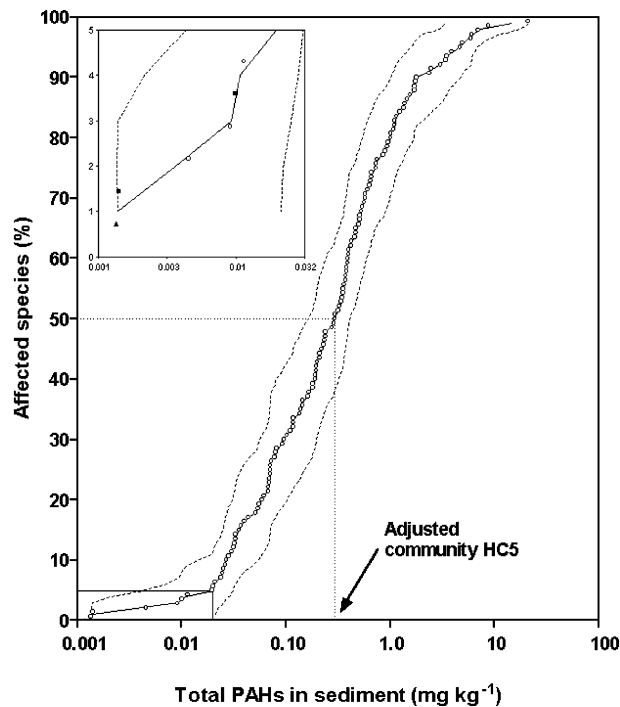


FIGURE 7. Field-based species sensitivity distribution for total PAHs. Format is the same as in Figure 5 except that Echinodermata and Platyhelminthes do not occur within the inset.

Crustacea and Polychaeta (Figure 7 inset; Table 2). It is noted that the f-SSD effectively identifies the most susceptible species in the field (Table 2) and gives results that are therefore potentially useful benchmarks for setting SQGs, for site-specific ecological risk assessments, and for evaluating the effectiveness of environmental remediation or clean up actions.

Comparisons with Current SQGs. Adjusted community HC5 and HC10 values were also computed from the f-SSDs for Ba, Cd, and total PAHs by the bootstrap method (Table 1). The adjusted community HC5 was used as the TEL or PNEC, while the adjusted community HC10 was utilized as the PEL. On this basis, TELs of Ba, Cd, and total PAHs were 2218 , 0.058 , and 0.294 mg kg^{-1} , respectively, while their respective PELs were 4876 , 0.129 , and 14.78 mg kg^{-1} (Table 1). The background baseline concentrations of Ba, Cd, and total PAHs in sediments from the reference stations were $126.8 \pm 7.50 \text{ mg kg}^{-1}$ ($\pm \text{SE}$; $n = 265$), $0.039 \pm 0.002 \text{ mg kg}^{-1}$ ($n = 265$), and $0.092 \pm 0.012 \text{ mg kg}^{-1}$ ($n = 133$), respectively. Hence, all derived TELs are higher than the background concentrations by 17.5-, 1.5-, and 3.2-fold for Ba, Cd, and PAHs, correspondingly.

As there is no existing saltwater SQG for Ba, only the results for Cd and total PAHs were compared with current SQGs from different governments (Figure 8). The SQGs varied considerably among different jurisdictions. It is noted that the TEL of Cd from this f-SSD approach was much smaller than existing SQGs (Figure 8a); such differences varied from 4.3-fold (Sweden SQGs) to 25.9-fold (Australia and Hong Kong SQGs), with an average 14.3-fold difference. On the contrary, our f-SSD-derived TEL for total PAHs was relatively close to the current SQGs across different governments. Specifically, Sweden, Norway, Portugal, and this study shared a similar TEL value of 0.3 mg kg^{-1} (Figure 8c). The Dutch TEL was exceptionally low and even lower than the background level of PAHs in the sediment of Norwegian continental shelf. Excluding the Dutch TEL, the ratios between existing TELs and our TEL for total PAHs ranged only from 1 to 13.6, with an average 4.7-fold difference.

TABLE 1. Medians of Various Hazardous Concentration Values^a Derived from f-SSDs of Sensitive Species by the Bootstrap Technique (34)

	sensitive species HC5	sensitive species HCx adjusted community HC5 used as TEL ^b	adjusted community HC10 used as PEL ^c
barium	130.1 (63.37–350.4)	HC28, 2218 (1436–2821)	HC56, 4876 (4087–6564)
cadmium	0.014 (0.006–0.021)	HC31, 0.058 (0.044–0.070)	HC62, 0.129 (0.098–0.177)
total PAHs	0.020 (0.004–0.031)	HC50, 0.294 (0.164–0.410)	HC99, 14.78 (3.393–21.45)

^a Hazardous concentration (HC) values are given as percent of affected species. Medians are 95% confidence intervals. ^b TEL denotes the threshold effect level, that is, below which harmful effects are unlikely to be observed (4). ^c PEL denotes the probable effect level, that is, above which harmful effects are likely to be observed (4).

TABLE 2. Most Sensitive Species Occurring below the Sensitive Species HC5 Values of the f-SSDs of Barium, Cadmium, and Total PAHs

species name	taxonomic group	rank ^a	% at sensitive species HCx
Barium			
<i>Aricidea</i> spp.	Polychaeta	1	0.29
<i>Oligochaeta</i> spp.	Platyhelminthes	2	0.58
<i>Mysella tumidula</i>	Mollusca	3	0.87
<i>Sosane sulcata</i>	Polychaeta	4	1.17
<i>Spisula elliptica</i>	Mollusca	5	1.46
<i>Yoldiella fraterna</i>	Mollusca	6	1.75
<i>Limopsis aurita</i>	Mollusca	7	2.04
<i>Vargula norvegica</i>	Crustacea	8	2.33
<i>Myriochele</i> spp.	Polychaeta	9	2.62
<i>Eurydice</i> spp.	Crustacea	10	2.92
<i>Exogone</i> spp.	Polychaeta	11	3.21
<i>Ophelia borealis</i>	Polychaeta	12	3.50
<i>Sabellidae</i> spp.	Polychaeta	13	3.79
<i>Unciola planipes</i>	Crustacea	14	4.08
<i>Ophiura robusta</i>	Echinodermata	15	4.37
<i>Haploopsis tubicola</i>	Crustacea	16	4.66
<i>Aricidea cerrutii</i>	Polychaeta	17	4.95
Cadmium			
<i>Mysella</i> spp.	Mollusca	1	0.31
<i>Diplocirrus hirsutus</i>	Polychaeta	2	0.63
<i>Ophiactis balli</i>	Echinodermata	3	0.94
<i>Bathyporeia</i> spp.	Crustacea	4	1.26
<i>Eugyra arenosa</i>	Platyhelminthes	5	1.57
<i>Mysella tumidula</i>	Mollusca	6	1.89
<i>Aricidea cerrutii</i>	Polychaeta	7	2.20
<i>Spisula elliptica</i>	Mollusca	8	2.52
<i>Yoldiella fraterna</i>	Mollusca	9	2.83
<i>Ophiura affinis</i>	Echinodermata	10	3.14
<i>Spio decorata</i>	Polychaeta	11	3.46
<i>Isaeidae</i> spp.	Crustacea	12	3.77
<i>Aglaophamus rubella</i>	Polychaeta	13	4.09
<i>Themisto compressa</i>	Crustacea	14	4.40
<i>Mysella bidentata</i>	Mollusca	15	4.72
Total PAHs			
<i>Spisula elliptica</i>	Mollusca	1	0.72
<i>Bathyporeia</i> spp.	Crustacea	2	1.44
<i>Aricidea cerrutii</i>	Polychaeta	3	2.16
<i>Spio filicornis</i>	Polychaeta	4	2.88
<i>Ampelisca typica</i>	Crustacea	5	3.60
<i>Aglaophamus rubella</i>	Polychaeta	6	4.32

^a Rank 1 = most sensitive.

Discussion

This new approach to derive field-based species sensitivity distributions (f-SSDs) is based on contaminant concentrations of a variety of chemicals and abundance data on a large number of species. The criterion used as an effect is when

abundances decrease by 50% from the abundances found at the background concentrations of the chemical in question. Yet at background concentrations there are large variations in abundances since there is no expectation of a given abundance at any one point in space or time where the sample is taken. The large scatter in the data reflects the natural, undisturbed environment. As concentrations increase, sensitive species (as defined here) decrease in abundance and the scatter becomes less as one single dominant factor acts (Figure 3e,f). Thus the apparently large variation in Figure 4 and relatively poor fit to the regression line is probably an accurate representation of the response of the species in the field.

The f-SSDs shown in Figures 5–7 show the response of the sensitive species to the named contaminant in the presence of other contaminants. For example, the data on Cd (Figure 6) represent the response of species to Cd given that there was a wide range of other chemicals present. There has been much concern that laboratory-based predictions of effects are not able to take account of possible interacting effects of chemicals in the field, and especially synergistic effects. In fact the f-SSD of Cd is measuring the response of benthic species to Cd while other chemicals are acting together. The fact that the species (Table 2) and their responses to Cd (Figure 6) are quite different than those of Ba (Figure 5) and total PAHs (Figure 7), suggests that we are not measuring a common response to all chemicals but rather that of a specific chemical in the presence of others (and also to varying physical factors such as sediment grain size and biotic factors such as competition and predation). Clearly we are not able to separate out the effect of Cd alone, but this is neither needed nor desired. For decades, ecotoxicologists have been seeking to predict effects of interacting chemicals in the field, and here is a method that, we believe, can give those data.

From the data shown in Figure 8, the considerable discrepancies between the laboratory-derived TELs and our field-based TEL for Cd and, to a lesser extent, total PAHs are probably associated with both biological and physicochemical factors. It is also noteworthy that most of the current SQGs have been derived for coastal benthic environments by use of shallow water species or “convenient” laboratory surrogate species, whereas the field-based SQGs developed in this paper were derived from much deeper waters, with an average depth of 185 m, where both environmental conditions (e.g., extent of light penetration, water temperature, and dissolved oxygen levels) and species composition are very different from those occurring in coastal areas. In addition, as mentioned above, different contaminants may interact in the field. Our data show that if a certain concentration of Cd (or total PAHs) is exceeded, there will be adverse effects as long as the other chemicals are present. A recent study, for example, has also demonstrated that, in the presence of PAHs, metal SQGs are abnormally low (Peter

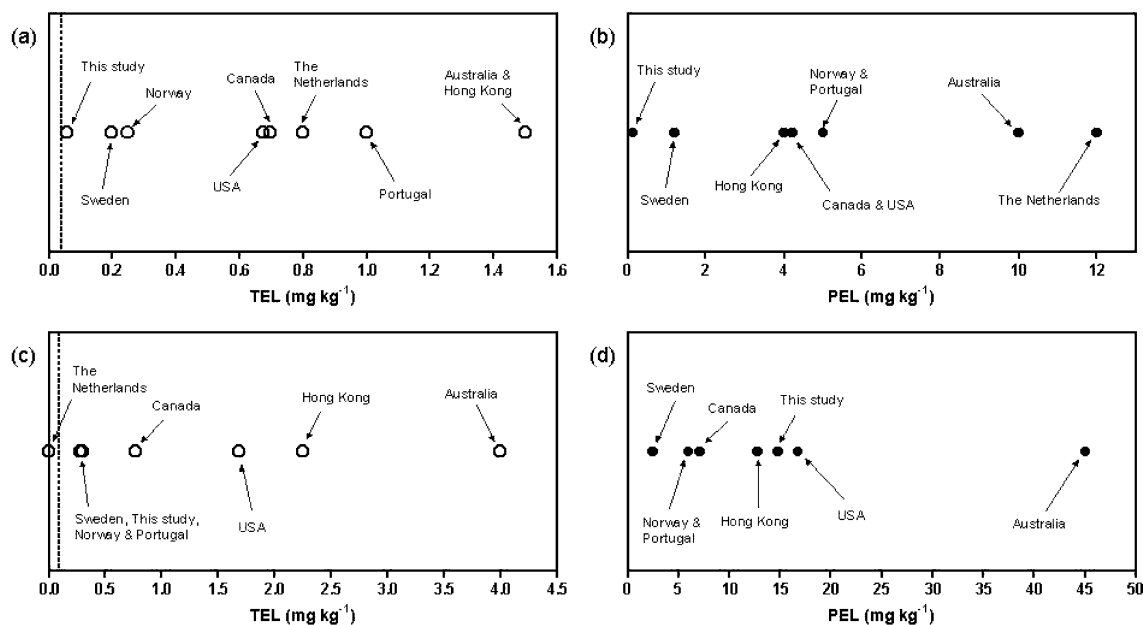


FIGURE 8. Comparison of SQGs, in terms of TEL (○) and PEL (●) values, for cadmium (a, b) and total PAHs (c, d) among different governments and those derived from the present study. Our TELs and PELs are compared with current SQGs from the United States (35), Canada (36), European countries (Norway, Sweden, The Netherlands, and Portugal) (37, 38), Australia (39, 40) and Hong Kong (41). Dotted line represents the background concentration of Cd (a) and PAHs (c) in the sediment of Norwegian continental shelf. Scales are different among the four panels.

Chapman, Golder Associates, Canada; personal communication). These findings suggest that it is not appropriate to define a safe level for Cd (or total PAHs) solely on the basis of a laboratory-derived PNEC. Furthermore, the present study utilized the EC_{50} data generated from highly sensitive species found in the field assemblages, and their sensitivities to Cd (or total PAHs) are highly likely to be different from those of common laboratory test species. Consequently, it is inappropriate to use laboratory-based PNEC values, even though they might be derived from multiple chronic bioassays with different species for a single substance, to accurately predict the likelihood of effects of the test substance in the field. The comparison of the SQGs derived from f-SSDs for Cd and total PAHs with those already in use by various governments suggests that many of the latter may be underprotective for natural benthic communities. We argue that the present site-specific, field-derived PNECs (or TELs) are better estimates of the 5% effect level for Cd and total PAHs with respect to the communities in the offshore Norwegian continental shelf. It is important to point out that prospective users of any SQGs, including the present ones, should take note of regional differences and variabilities when assessing sediment quality by use of SQGs derived from other regions (1, 2, 33). Although the approach outlined in this paper is applicable to a range of ecological systems, the guidelines derived are site-specific.

Nevertheless, there are some limitations and drawbacks of this f-SSD approach. First, this field-based approach will only work retrospectively. Second, it may only work when at least some species have been impacted (i.e., enough species can be identified as sensitive). Third, such an approach cannot be used for predicting acceptable exposures for new chemicals that have yet to be released into the environment. Fourth, such an f-SSD will require a reasonably large data set, which has to be constructed from data accumulated over a long period of time (e.g., 6 years in this study). Such a temporal factor may potentially act as another confounding factor in the derivation of SQGs. At present, we are exploring the possibility of adding a “time” factor to the current model with a view to increasing its applicability. For instance, if concentrations of certain contaminants declined between

the first and second sampling, and the abundances of certain species increased, this could provide additional useful information.

One important message from this study is that the importance of establishing monitoring programs and appropriate databases for sediment chemical and biota information needs to be stressed. The availability of good quality and comprehensive data sets is crucial as the usefulness and effectiveness of the f-SSD approach described in this paper will depend ultimately on the availability of such data sets. The Marine Habitat Committee of the International Council for the Exploration of the Sea has recently initiated an endeavor to collect such data on a regular basis, to build an inventory of national sediment trends (37). Recently, regulatory authorities in the United States and Canada have also been further reevaluating similar field-based approaches for setting site-specific SQGs and assessing ecological risks of benthic communities in both fresh and saltwater environments (13, 42). Likewise, the Government of Hong Kong Special Administrative Region (HKSAR) has been supporting a similar monitoring program in its Victoria Harbor. We concur with Chapman and Mann (3) that the key to successful use of SQGs lies in understanding how they are derived and their limitations. More importantly, SQGs should not be used alone for remediation decisions except for cases of “simple contamination where adverse biological effects are clearly observable. Notwithstanding, SQGs can be used alone when the costs of further investigation outweigh the costs of remediation, and there is agreement to act instead of conducting further investigations” (1). Presently, we are constructing f-SSDs for a wider range of chemicals and are investigating possible combined effects of substances with a similar mode of action. In addition, it is conceivable that there may be correlations between contaminant levels and sediment parameters (e.g., particle size and carbon content). Previous laboratory and field studies (e.g., refs 43 and 44) also showed that partitioning of PAHs is largely dependent on organic carbon content and particle size of the sediment. The potential influence of confounding factors such as depth and particle characteristics is currently being examined.

In conclusion, the proposed site-specific f-SSD approach alleviates the problem of current laboratory-driven methodologies for deriving SQGs that extrapolate laboratory-based observations to field situations, from a few species to many species, and from a single chemical to a mixture. The f-SSD approach is a specific measurement of the biological community response to a specific chemical in the presence of other chemicals in the sediment that takes into account varying physical and biotic factors. Our results indicate that this f-SSD approach is achievable, generating SQGs that are more environmentally and ecologically relevant, and can be integrated into the current laboratory-derived SQGs or consensus SQGs (4, 23). Moreover, f-SSDs can be directly used as benchmarks for conducting probabilistic risk assessments and for evaluating the cost-effectiveness of environmental clean-up activities.

Acknowledgments

The work described in this paper is supported by the Area of Excellence Scheme under the University Grants Committee of the Hong Kong Special Administration Region, China (Project AoE/P-04/2004). We thank Dr. Peter Chapman of Golder Associates, Canada, and three anonymous reviewers for their invaluable and constructive comments on an early draft of the manuscript.

Supporting Information Available

Table showing raw data of Figure 8, that is, a comparison between SQGs for Cd and total PAHs from different jurisdictions and those derived in the present study (Word, PDF). This information is available free of charge via the Internet at <http://pubs.acs.org>.

Literature Cited

- Wenning, R. J.; Ingersoll, C. G. Summary of the SETAC Pellston Workshop on Use of Sediment Quality Guidelines and Related Tools for the Assessment of Contaminated Sediments, 17–22 August 2002, Fairmont, MT; SETAC: Pensacola, FL, 2002.
- Long, E. R.; Field, L. J.; MacDonald, D. D. Predicting toxicity in marine sediments with numerical sediment guidelines. *Environ. Toxicol. Chem./SETAC* **1998**, *17*, 714–727.
- Chapman, P. M.; Mann, G. S. Sediment quality value (SQVs) and ecological risk assessment (ERA). *Mar. Pollut. Bull.* **1999**, *38*, 339–334.
- MacDonald, D. D.; Ingersoll, C. G.; Berger, T. A. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Arch. Environ. Contam. Toxicol.* **2000**, *39*, 20–31.
- Campbell, D. E. Proposal for including what is valuable to ecosystems in environmental assessments. *Environ. Sci. Technol.* **2001**, *35*, 2867–73.
- Pollard, S. J.; Yearsley, R.; Reynard, N.; Meadowcroft, I. C.; Duarte-Davidson, R.; Duerden, S. L. Current directions in the practice of environmental risk assessment in the United Kingdom. *Environ. Sci. Technol.* **2002**, *36*, 530–538.
- Calow, P.; Forbes, V. E. Does ecotoxicology inform ecological risk assessment? *Environ. Sci. Technol.* **2003**, *37*, 146A–151A.
- Calow, P., Eds; *Handbook of Environmental Risk Assessment and Management*; Blackwell: London, 1998.
- U.S. Environmental Protection Agency. *Ecological Risk Assessment Federal Guidelines*; Government Institutes: Rockville, MD, 1998.
- Sergeant, A. Ecological risk assessment: history and fundamentals. In *Human and Ecological Risk Assessment*; Paustenbach, D. J., Ed.; Wiley Interscience: New York, 2002.
- Leung, K. M. Y.; Morritt, D.; Wheeler, J. R.; Whitehouse, P.; Sorokin, N.; Toy, R.; Holt, M.; Crane, M. Can saltwater toxicity be predicted from freshwater data? *Mar. Pollut. Bull.* **2001**, *42*, 1007–13.
- Wheeler, J. R.; Leung, K. M. Y.; Morritt, D.; Whitehouse, P.; Sorokin, N.; Toy, R.; Holt, M.; Crane, M. 2002. Freshwater to saltwater toxicity extrapolation using species sensitivity distributions. *Environ. Toxicol. Chem./SETAC* **2002**, *21*, 2459–2467.
- Mount, D. R.; Ankley, G. T.; Brix, K. V.; Clements, W. H.; Dixon, D. G.; Fairbrother, A.; Hickey, C. W.; Lanno, R. P.; Lee, C. M.; Munns, W. R.; Ringer, R. K.; Staveley, J. P.; Wood, C. M.; Erickson, R. J.; Hodson, P. V. Effects assessment. In *Reevaluation of the State of the Science for Water Quality Criteria Development*; Reiley, M. C.; Stubblefield, W. A.; Adams, W. J.; Di Toro, D. M.; Hodson, P. V.; Erickson, R. J.; Keating, F. J., Jr., Eds.; SETAC: Pensacola, FL, 2003.
- Forbes, V. E.; Calow, P.; Sibly, R. M. Are current species extrapolation models a good basis for ecological risk assessment? *Environ. Toxicol. Chem./SETAC* **2001**, *20*, 442–447.
- OECD. *Guidelines for the Testing of Chemicals*; Organization for Economic Cooperation and Development: Paris, 1993.
- Wheeler, J. R.; Grist, E. P. M.; Leung, K. M. Y.; Morritt, D.; Crane, M. Species sensitivity distribution: Data and model choice. *Mar. Pollut. Bull.* **2002**, *45*, 192–202.
- Van Straalen, N. Ecotoxicology becomes stress ecology. *Environ. Sci. Technol.* **2003**, *37*, 324A–330A.
- Kwok, K. W. H.; Leung, K. M. Y. Toxicity of antifouling biocides to the intertidal harpacticoid copepod *Tigriopus japonicus* (Crustacea, Copepoda): effects of temperature and salinity. *Mar. Pollut. Bull.* **2005** (in press).
- Neff, J. M.; Bean, D. J.; Cornaby, B. W.; Vaga, R. M.; Gulbransen, T. C.; Sealon, J. A. *Sediment Quality Criteria Methodology Validation: Calculation of Screening Level Concentrations from Field Data—Final Report*; U. S. Environmental Protection Agency: Washington, DC, 1986.
- Long, E. R.; MacDonald, D. D.; Smith, S. L.; Calder, F. D. Incidence of adverse effect within ranges of chemical concentrations in marine and estuarine sediment. *Environ. Manage.* **1995**, *19*, 81–97.
- Morrissey, D. J.; Underwood, A. J.; Howitt, L. Development of sediment quality criteria: a proposal from experimental field-studies of the effects of copper on benthic organisms. *Mar. Pollut. Bull.* **1995**, *31*, 372–377.
- DelValls, T. A.; Chapman, P. M. Site-specific sediment quality values in the Gulf of Cadiz (Spain) and in San Francisco Bay (USA) using the sediment quality triad and multivariate analysis. *Cienc. Mar.* **1998**, *24*, 313–336.
- Swartz, R. C. Consensus sediment quality guidelines for polycyclic aromatic hydrocarbon mixtures. *Environ. Toxicol. Chem./SETAC* **1999**, *18*, 780–787.
- von Stackelberg, K.; Menzie, C. A. A cautionary note on the use of species presence and absence data in deriving sediment criteria. *Environ. Toxicol. Chem./SETAC* **2002**, *21*, 466–472.
- Ellingsen, K. E. Soft-sediment benthic biodiversity on the continental shelf in relation to environmental variability. *Mar. Ecol. Prog. Ser.* **2002**, *232*, 15–27.
- Olsfard, F.; Gray, J. S. A comprehensive analysis of the effects of offshore oil and gas exploration and production on the benthic communities of the Norwegian Continental Shelf. *Mar. Ecol. Prog. Ser.* **1995**, *122*, 277–306.
- Gray, J. S.; Bakke, T.; Beck, H. J.; Nilssen, I. Managing the environmental effects of the Norwegian oil and gas industry: from conflict to consensus. *Mar. Pollut. Bull.* **1999**, *38*, 525–530.
- Manual for Baseline Monitoring Studies in the Vicinity of Petroleum Installations in Norwegian Sea Areas*. Norwegian Pollution Control Authority Report 90:01, 1990 (in Norwegian).
- Guidelines for Environmental Monitoring of Petroleum Activities on the Norwegian Shelf*. Norwegian Pollution Control Authority Report 99:01, 1999 (in Norwegian).
- Norwegian Standard 4770. Water Analysis-Determination of Metals by Atomic Absorption Spectrophotometry, Atomization in Flame – General Principles and Guidelines, NS 4770, 1994.
- ISO/IEC/EN 17025. ISO, 05.01.2005, <http://www.fasor.com/iso25/>, 2005.
- Hanson, M. L.; Solomon, K. R. New technique for estimating thresholds of toxicity in ecological risk assessment. *Environ. Sci. Technol.* **2002**, *36*, 3257–3264.
- Posthuma, L.; Suter, G. W., II; Traas, T. P., Eds. *Species Sensitivity Distributions in Ecotoxicology*; Lewis Publishers: Boca Raton, FL, 2002.
- Grist, E. P. M.; Leung, K. M. Y.; Wheeler, J. R.; Crane, M. Better bootstrap estimation of hazardous concentration thresholds for aquatic assemblages. *Environ. Toxicol. Chem./SETAC* **2002**, *21*, 1515–1524.
- Buchman, M. F. *NOAA Screening Quick Reference Tables*; NOAA HAZMAT Report 99-1; Coastal Protection and Restoration Division, National Oceanic and Atmospheric Administration: Seattle, WA, 1999.
- CCME. *Canadian Environmental Quality Guidelines*; Canadian Council of Ministers of the Environment: Winnipeg, MB, Canada, 1999.

- (37) ICES. *Report of the Working Group on Marine Sediments in Relation to Pollution*; Report ICES CM 2003/E:04 Ref. ACME; Marine Habitat Committee, International Council for the Exploration of the Sea: Copenhagen, Denmark, 2003.
- (38) MHSPE. *Setting Integrated Environmental Quality Standards for Substances in The Netherlands*; Ministry of Housing, Spatial Planning and the Environment: The Hague, The Netherlands, 1999.
- (39) ANZECC and ARMCANZ. *Australian and New Zealand Guidelines for Fresh and Marine Water Quality, Volume 2, Aquatic Ecosystems*; National Water Quality Management Strategy Paper 4; Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand: Canberra, Australia, 2002; Chapt. 8.4, Sediment Quality Guidelines.
- (40) Environment Australia. *National Ocean Disposal Guidelines for Dredged Material*; The Commonwealth Department of the Environment and Heritage: Canberra, Australia, 2002.
- (41) EPD. *Marine Water Quality in Hong Kong in 2002*; Environmental Protection Department, The Government of Hong Kong Special Administrative Region: Hong Kong, 2003.
- (42) Nord, M. A. Perspective: Recommendations for the implementation of a national sediment quality policy in the United States. *Hum. Ecol. Risk Assess.* **2001**, 7, 641–650.
- (43) Cornelissen, G.; van Zuilen, H.; van Noort, P. C. M. Particle size dependence of slow desorption of in situ PAHs from sediments. *Chemosphere* **1999**, 38, 2369–80.
- (44) Wang, X. C.; Zhang, Y. X.; Chen, R. F. Distribution and partitioning of polycyclic aromatic hydrocarbons (PAHs) in different size fractions in sediments from Boston Harbor, United States. *Mar. Pollut. Bull.* **2001**, 42, 1139–49.

Received for review March 6, 2005. Revised manuscript received May 13, 2005. Accepted May 16, 2005.

ES050450X