Use of Sediment Quality Guidelines and Related Tools for the Assessment of Contaminated Sediments

Executive Summary Booklet of a SETAC Pellston Workshop

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Abstract

This publication summarizes the results of a Pellston Workshop sponsored by the Society of Environmental Toxicology and Chemistry (SETAC), held 17-22 August 2002 in Fairmont, Montana, USA. The full technical proceedings of the workshop will be published separately by SETAC in 2003. Previous SETAC workshops have focused on sediment ecological risk assessment (ERA) (Dickson et al. 1987; Ingersoll et al. 1997) and porewater toxicity testing (Carr and Nipper 2003). Another recent workshop addressed the application of weight-of-evidence (WOE) methods in ERA (Burton et al. 2002). These previous workshops focused on how, when, and why ERAs are needed in sediment assessments. However, more focused discussion among scientists, environmental regulators, and environmental managers is now needed to build on this previous work. Specifically, additional guidance is needed on procedures that can be used to integrate the information derived from multiple chemical and biological lines of evidence (LOE). These LOE, which are developed through the application of different assessment tools, often include the use of chemically based sediment quality guidelines (SQGs) to evaluate sediment contamination and to help practitioners in sediment assessment and management formulate risk management decisions. This workshop focused on evaluation of the scientific foundations supporting different chemically based numeric SQGs and methods to improve the integration of SQGs into different sediment quality assessment frameworks that include information derived from multiple chemical and biological LOE.

The approaches used in the development of SQGs can be subdivided as being either empirically based or mechanistically (i.e., equilibrium partitioning [EqP]) based. The scientific underpinnings of SQG derivations vary widely, but none of the guidelines appears to be intrinsically flawed. Nonetheless, interpretations of contaminants in sediment using SQGs should be linked to their derivation and narrative intent. While SQGs are not universally applicable at all sites of interest, SQGs can be used to help identify ranges of contaminants in sediments where adverse effects on benthic organisms are unlikely, uncertain, and likely.

A critical review of published studies from a wide range of laboratory and field studies in freshwater, estuarine, and marine environments (encompassing more than 8000 sediment samples) indicates that effects-based SQGs can be used to assess the probability of observing adverse biological effects to benthic organisms with known levels of statistical confidence. The incidence of effects or the degree of the response increases with increasing sediment contamination based on the SQGs. For both laboratory toxicity tests and benthic community studies, an incremental increase in adverse biological effects has been observed frequently with an incremental increase in contamination based on evaluations with SQGs. Current SQGs and the methods used to develop SQGs are generally appropriate for making some management decisions. Applied appropriately, SQGs can contribute to making determinations about whether contaminants in sediments in different aquatic environments pose relatively low or very high potential for significant toxicity to benthic organisms. While SQGs alone may be sufficient for decision-making, in some situations, multiple LOE developed from

sediment toxicity and bioaccumulation testing, benthic community assessment, and risk assessment should be used to support sediment management decisions. The WOE required for decision-making should be established, in large part, based on the pathways by which risks might exist, receptors for those risks, the spatial extent of the contamination, the regulatory goals, and long-term costs of different management decisions.

Because of the uncertainties inherent in different SQG approaches and the unique or varied environmental and ecological conditions that characterize different freshwater, estuarine, and marine environments, sediment management decisions should be based on site-specific information generated to evaluate the predictive ability of SQGs at a site of interest. Sediment assessment frameworks for different management purposes should be flexible and should be guided by specific questions that address toxicity to and bioaccumulation by sediment-dwelling organisms or risks to wildlife or human health. Further, sediment assessments should be driven by site-specific questions, which in some situations may require a multitiered evaluation involving a suite of assessment tools chosen to appropriately answer the questions established a priori and to generate specific biological or chemical LOE. Development of a relevant set of site-specific questions is best done in conjunction with a site-specific model. A scientifically defensible WOE approach is the appropriate framework in which to place the results from multiple LOE to provide a meaningful interpretation of ecological significance and to make sound management decisions. A review of studies conducted in freshwater, estuarine, and marine aquatic environments was used to examine the strengths and limitations of current approaches for using SQGs in conjunction with other sediment assessment tools. Recommendations for future research developed at the workshop should lead to improvements in the determination of the effects, or the absence of effects, associated with chemical contaminants in sediments.

Introduction

During the past 30 years, environmental scientists, engineers, and regulatory authorities throughout the world have devoted considerable resources to assessment, management, and remediation of chemical contaminants in sediments. Sediment quality has become a serious and potentially costly economical and ecological issue for navigation dredging projects, waterway restoration programs, recreational and commercial fisheries management, water-quality protection, and natural resource restoration.

Numerous scientific and engineering studies have shown that addressing contaminated sediments is a complex undertaking, one that is likely to engage diverse elements of society. Complexities arise from the great variability in the physical and biogeochemical characteristics; human and ecological receptors; and the cultural, social, and economic values associated with different freshwater, estuarine, and marine environments. Because of these complexities, the assessment and management of contaminated sediments in ports and harbors, rivers, lakes, and at hazardous waste sites does not easily lend itself to simple "one-size-fits-all" investigation methods and presumptive management decisions.

In response to society's increasing demands for greater environmental protection of aquatic resources and restoration of impaired or degraded rivers and estuaries, scientists in several countries have developed a variety of methods for evaluating the degree to which sediment-associated chemicals might adversely affect aquatic organisms. These methods have resulted in chemically based numerical or narrative SQGs designed to protect benthic organisms; support or maintain designated uses of freshwater, estuarine, and marine environments; and to assist sediment assessors and managers charged with the interpretation of sediment quality. At the same time, several different sediment frameworks, addressing different management objectives such as navigation channel maintenance and environmental cleanup, have been developed that include the use of SQGs for screening or decision-making purposes. The SQG approaches selected by environmental agencies in various countries to support different sediment management objectives have varied depending on the receptors that need to be protected (e.g., sediment-dwelling organisms, wildlife, or humans), the degree of ecological or human health protection that can be afforded, the geographic area to which the SQGs are intended to apply, and the regulatory context.

"Sediment quality guidelines," as defined at the outset of this Pellston Workshop, are numerical chemical concentrations intended to be either protective of biological resources, or predictive of adverse effects to those resources, or both. SQGs for assessing sediment quality relative to the potential for adverse effects on sediment-dwelling organisms have been derived using both mechanistic and empirical approaches, primarily including the EqP approach (Di Toro, Mahony et al. 1991; Di Toro, Zarba et al. 1991; Ankley et al. 1996; NYSDEC 1998; Di Toro and McGrath 2000), screening-level concentration approach (Persaud et al. 1993; Von Stackelberg and Menzie 2002), effects range–low (ERL) and effects range–median (ERM) approaches (Long et al. 1995; U.S. Environmental Protection Agency [USEPA] 1996), threshold-effects level (TEL) and probable-effects level (PEL) approaches

(MacDonald et al. 1996; Smith et al. 1996; USEPA 1996), the apparent-effects threshold (AET) approach (Barrick et al. 1988; Ginn and Pastorok 1992; Cubbage et al. 1997), and, most recently, the "consensus-based" evaluation approach (Swartz 1999; MacDonald, DiPinto et al. 2000; MacDonald, Ingersoll et al. 2000) and the logistic regression modeling (LRM) approach (Field et al. 1999, 2002).

The underlying supposition in the derivation of effects-based SQGs is that these guidelines can be used as a substitute for direct measures of potential adverse effects of contaminants in sediments on benthic organisms. The mechanistically based SQGs have been developed and tested using laboratory spiked sediments and compared to toxicity tests by using field-collected sediments. The empirically based SQGs have typically been developed using large databases with matching measures of sediment chemistry and toxicity with field-collected samples. These databases have also been used to evaluate the ability of SQGs to predict sediments to be either toxic or non-toxic in laboratory tests or in benthic community assessments in numerous freshwater, estuarine, and marine environments. Generally, the results of analyses conducted with these databases in conjunction with field validation studies involving contaminated sediments suggest that SQGs have been reasonable predictors primarily of acute effects or no effects on benthic organisms. The issue of causality is often contentious for empirically derived SQGs because the cause of observed toxicity or biological degradation from field-collected sediments may or may not be a result of the contaminants measured in sediments but may instead be a response to one or more co-occurring chemicals or confounding factors associated with sediment geochemistry or benthic habitat characteristics. Additional uncertainties arise when the ecological receptors potentially at risk and the characteristics of the sediment are not appropriate surrogates or adequately similar to conditions used in the sediment toxicity studies that led to the development of SQGs.

Increasingly, SQGs are used as informal benchmarks or aids to interpretation of sediment chemistry data, in some cases in spite of their narrative intent, to interpret historical trends, identify potential problem chemicals or reaches in a waterway, interpret or design ambient monitoring programs, classify hot spots, establish baseline conditions in nonurbanized systems, rank contaminated waterways, and help chose sites for more detailed studies (Long and MacDonald 1998). Other applications of SQGs, which have generated considerable controversy within the scientific, industrial, and environmental regulatory communities, include identifying the need for source control measures to address certain chemicals before release, triggering regulatory action as mandatory standards, and establishing target remediation objectives (USEPA 1997; Barnthouse and Stahl 2002).

Numerous studies have shown that the current SQGs perform well relative to their intended purposes of predicting primarily acute adverse effects, or the lack thereof, in field validation studies using laboratory tests or benthic community surveys. Nonetheless, over the past few years, several concerns have been expressed regarding the use of SQGs in sediment quality assessments. One of these concerns is the ability of SQGs to adequately predict the presence or absence of chronic toxicity to sediment-dwelling organisms in field-collected sediments. A second concern relates to the ability

of SQGs to predict effects resulting from bioaccumulation of sediment-associated contaminants. A third concern focuses on the ability of SQGs to establish cause and effect relationships. A fourth concern deals with the ability of SQGs developed for one endpoint (e.g., amphipod mortality measured in the laboratory) to be predictive of effects on organisms exposed in the field. These concerns have not been eased by the growing recognition that SQGs, used in conjunction with other tools such as sediment toxicity tests, bioaccumulation, and benthic community surveys, can provide a WOE for assessing the hazards associated with contaminated sediments (Ingersoll et al. 1997; Chapman et al. 2002). This is due, in large part, to the absence of guidance on what constitutes a WOE approach and the lack of guidance in the implementation of quantitative, rather than subjective or qualitative, procedures for evaluation of multiple biological and chemical LOE generated from field investigations (Burton et al. 2002).

Further difficulties have become apparent in the context of efforts in the U.S., the European Union, and elsewhere to develop or improve sediment assessment frameworks for different management situations. For example, Nord (2001) recently reported that inconsistent approaches in the U.S. for assessing or managing contaminated sediments are due, in part, to the lack of communication or consensus by regulators and the scientific community on a number of key technical issues, which has resulted in inaction or ineffective implementation of regulatory policies concerned with addressing sediments.

Nevertheless, it is widely recognized in a world of limited resources that a balance should be struck between relying solely on numeric SQGs and the use of other tools for different sediment assessment and management scenarios. While there has been considerable emphasis on the use of SQGs for informal (i.e., nonregulatory) purposes and as initial screening values in enforcement and regulatory programs, there has not been broad agreement on how SQGs should be used in different regulatory and management frameworks to define cleanup actions or injuries to natural resources. The Puget Sound Dredged Material Management Program and Washington State's Sediment Management Standards Program have established, at least in part, how regional SQGs are used for various regional regulatory decisions (Chapters 173 through 204 of the Washington State Administrative Code). While SQGs may be appropriate in some situations, scientists generally acknowledge there are several limitations and uncertainties associated with different SQG approaches that have the potential to cause confusion and concern among sediment assessment and management practitioners.

This summary outlines the findings of the SETAC Pellston Workshop, "Use of Sediment quality Guidelines and Related Tools for the Assessment of Contaminated Sediments," 17-22 August 2002, Fairmont, Montana, USA. The full technical workshop proceedings are being prepared for publication by SETAC in 2003. Three previous SETAC workshops, 2 that focused on sediment ERA (Dickson et al. 1987; Ingersoll et al. 1997) and 1 on porewater toxicity testing (Carr and Nipper 2003), along with a recent workshop addressing the application of WOE methods in ERA (Burton et al. 2002) focused on how, when, and why ERAs are needed in sediment assessments. However, more focused discussion among scientists, environmen-

tal regulators, and environmental managers is now needed to build on this previous work. Specifically, additional guidance is needed on procedures that can be used to integrate the information derived from multiple chemical and biological LOE. These LOE, which are developed through the application of different assessment tools, often include chemically based SQGs to evaluate sediment contamination and to help practitioners in sediment assessment and management formulate risk management decisions. This workshop focused on evaluation of the scientific foundation underlying available SQGs and on methods to improve the integration of SQGs into sediment quality assessment frameworks that also include information derived from multiple chemical and biological LOE. Specifically, the discussions among invited experts from 8 countries representing 3 continents focused on the following 5 workgroup topics: 1) the scientific underpinnings of the current available SQG schemes, 2) the predictive ability of SQGs for toxicity and bioaccumulation, 3) application of other related sediment assessment tools, 4) the role of SQGs and other tools in different sediment assessment frameworks, and 5) the role and relative appropriateness of SQGs and other tools in different aquatic environments. In each of these discussions, participants were urged to seek consensus, where possible, on specific technical issues of concern to sediment assessment and management practitioners, and to identify recommendations for future research that could lead to improvements in the existing methods available to quantify the effects, or absence of effects, associated with the presence of chemical contaminants in sediment.

Workshop Purpose and Goals

This workshop brought together 55 experts in the fields of sediment assessment and management from Australia, Canada, France, Germany, Great Britain, Italy, the Netherlands, and the U.S. for 6 days of discussion on the use of SQGs and other sediment assessment tools. Participants (see Appendix 1) included representatives from regulatory and nonregulatory government agencies, academia, industry, environmental groups, and consulting firms involved in assessment, investigation, management, and basic research on contaminants in sediment. Participants were assigned to 5 different workgroups and asked to address one of the following five topics.

Workgroup 1
Review of the scientific underpinnings associated with different SQG approaches

The objective of this workgroup was to critically review the different approaches used to derive SQGs worldwide, focusing on the technical foundation, strengths and limitations, and methodological uncertainties associated with the different approaches with a view to identifying recommendations to improve on the development and application of SQGs. There are a number of complex scientific issues concerning the derivation of SQGs for which clarification could aid in understanding the scientific

foundations of the different approaches and their importance to sediment management decisions. Specifically, participants addressed 5 questions:

1) What are the uncertainties, deficiencies, strengths, and limitations of SQGs derived from matching toxicity and chemistry data with field-collected sediments?

2) What additional qualifiers of SQGs are required to account for bioavailability?

3) What are the strengths and limitations of SQGs derived from EqP?

4) Are there issues of changing sediment chemistry, organism uptake pathways, feeding strategies, and trophic levels that might limit the acceptability of SQGs as predictors of the health of sediment ecosystems?

5) Has the scientific basis for the various uses of SQGs been field validated?

Workgroup 2
The use of SQGs to estimate the potential for effects, or no effects, of sediment-associated contaminants in laboratory toxicity tests and in benthic community assessments

The main objective of this workgroup was to evaluate the predictive ability of effects-based SQGs in laboratory toxicity tests and in benthic community assessments. A primary concern among scientists and regulators evaluating the application of SQG approaches in the assessments of sediment quality is the ability to demonstrate that SQGs are sufficiently predictive of the presence, or absence, of toxicity to sediment-dwelling organisms or to higher-trophic-level organisms. A related concern has been the ability to predict bioaccumulation of chemicals by benthic organisms and the effects of that bioaccumulation either directly on sediment-dwelling organisms or through trophic transfer of contaminants. It is clear that protection of the ecosystem beyond direct effects on benthos is critical to the long-term management of contaminated sediments. The source of the current debate is the extent to which SQGs have been used in some instances without an understanding of their narrative intent or the quantitative extent of their predictive ability. Specifically, participants addressed the following 3 questions:

1) How well do SQGs represent the potential for effects or no effects observed in laboratory toxicity tests and in field studies of benthic communities?

2) How well do SQGs represent the potential for effects or no effects in organisms as a result of contaminant uptake and/or trophic transfer?

3) How have SQGs been applied and validated in the field as part of sediment management and risk management decision-making?

Workgroup 3

The role of other assessment tools available for evaluating sediment contamination

The workgroup objective was to evaluate the use of other tools as either a supplement or as an alternative to SQGs in the assessment of contaminated sediments. An increasingly common emphasis of the different sediment frameworks that have been suggested or applied by various U.S. and international regulatory agencies is the use of decision trees or an integrated WOE approach in sediment quality identification, assessment, and management. A common criticism of the different SQG approaches included in some of these frameworks is their inability to account for site-specific environmental conditions or unique biological characteristics, leading to either inadequate or overly conservative risk management decisions. Improvements in 3 areas have been advocated: 1) the integration of biological information; 2) the integration of routes of exposure considerations; and, 3) the inclusion of biologically based tools to supplement the information provided by more traditional chemically based SQGs. Because different SQGs do not explicitly consider routes of exposure to aquatic biota, it has been suggested that improvements to SQGs that account for different possible exposure scenarios could lead to more scientifically credible risk-based assessments. A closely related issue that has also confounded different sediment frameworks is the interpretation of information generated from multiple chemical and biological LOE. The difficulties attributed to contrasting biological and/or chemical results with SQGs has led sediment assessment and management practitioners to ask for clearer guidance from the scientific community on the use of one or more WOE approaches (Ingersoll et al. 1997; Burton et al. 2002; Chapman et al. 2002). Specifically, participants addressed 5 questions:

1) What types of data should be generated for one or more frameworks at different types of sites to determine the applicability of SQGs and reduce technical uncertainties at a particular site?

2) What can be done to improve the understanding of the uncertainties (or reduce the uncertainties) associated with the more conventional assessment methods (e.g., biological testing, benthic infaunal analysis, etc.) used to derive SQGs?

3) How can we use the understanding of the methods used to derive SQGs to make them more site-specific?

4) Can the accuracy of assessing sediment quality be improved using biologically based thresholds in a WOE process (as opposed to using chemically based threshold SQGs)?

5) What are the uncertainties, deficiencies, strengths, and limitations of biologically based thresholds?

Workgroup 4
The role of SQGs and related chemical and biological assessment tools in different sediment assessment and management frameworks

The objective of this workgroup was to explore whether SQGs alone or as part of a broader framework are sufficient to provide technical advice on the assessment and management of contaminated sediment. Several different sediment assessment and management frameworks have been proposed in the U.S. and elsewhere. For example, in the specific area of dredged material assessment, the 1996 protocol to the London Convention developed by more than 80 countries represents one approach (GIPME 2000). In the U.S., the National Research Council (NRC) report on management of polychlorinated biphenyl (PCB)-contaminated sediments encourages a risk-based framework (NRC 2001). Such a framework would incorporate chemistry, ecotoxicology, bioaccumulation, effects on benthic community structure, and both temporal and spatial considerations of sediment conditions. In these cases, as well as others, the risks and tradeoffs posed by these varied LOE are poorly understood and rarely explicitly evaluated in the different sediment frameworks. With this in mind, participants were charged to address 4 questions:

1) What are the required elements and decision points of sediment assessment frameworks?
2) How can these elements best be assembled in an assessment or WOE decision?
3) What is the utility of different SQG schemes as part of a WOE approach to contaminated sediment assessment and management?
4) How can LOE and WOE best be used in an overall sediment assessment framework to make sediment quality assessment decisions?

Workgroup 5
Use of SQGs and related tools for evaluation of sediments in different aquatic environments

The workshop objective was to evaluate the use of SQGs across a range of different aquatic environments. Additional technical concerns focused on the use of SQGs across a range of different freshwater, estuarine, and marine ecosystems, and their use in environments that involve multiple contaminants, multiple exposure pathways, multiple ecological (and human) receptors, variations in resource use and management, or differences in temporal or spatial scales. There is growing recognition among scientists that 6 factors should be considered in conjunction with SQGs when addressing sediment quality: 1) chemical and biological changes over spatial scales (sometimes measured in meters or kilometers, and at other times measured in centimeters); 2) chemical and biological changes over temporal scales (similarly, sometimes measured in hours, while at other times measured in years); 3) physical heterogeneity of the contaminated material; 4) watershed characteristics; 5) interactions of chemical mixtures; and 6) biological diversity at different trophic levels. For several of these factors, chronic stresses unrelated to contaminant levels may pose more important challenges to resident ecological receptors; therefore, characterizing chronic stresses of sediment

origin is increasingly recognized as an important issue in sediment quality assessment. Consequently, participants addressed the following 2 questions:

1) Are SQGs alone sufficient for management decision-making in different aquatic environments?

2) How applicable are SQGs and different tools for characterization of sediment quality in different aquatic environments?

Workgroup Conclusions

Workgroup 1
Scientific underpinnings of SQGs

The different approaches used to develop SQGs can broadly be subdivided into 2 categories: 1) empirically derived guidelines, and 2) mechanistically based guidelines. Empirical guidelines are derived from databases of sediment chemistry (concentrations of specific sediment contaminants) and observed biological effects (e.g., those derived from sediment toxicity tests and benthic community information). These data are arrayed on a continuum of increasing chemical concentration. Various algorithms are used to define specific concentrations associated with particular levels of effect or no effect. Common examples include ERLs, ERMs, TELs, PELs, and AETs. In contrast, mechanistic guidelines are derived from a theoretical understanding of the factors that govern bioavailability of sediment contaminants, and known relationships between chemical exposure or uptake, and toxicity. EqP theory forms the basis for all current mechanistic guidelines. Mechanistic guidelines account for bioavailability through normalization to sediment characteristics that affect bioavailability, primarily organic carbon for nonionic organic chemicals, and simultaneously extracted metal–acid-volatile sulfide (SEM–AVS), organic carbon, or other sediment fractions for metals. Empirical guidelines may use either dry weight concentrations or normalized concentrations, depending on the guideline.

Though the scientific underpinnings of the different SQG approaches vary widely, none of the approaches appear to be intrinsically flawed. All approaches reviewed are grounded in concepts that, viewed in isolation, are sound. Potential issues of scientific appropriateness of SQGs are, therefore, more a matter of whether a particular application of an SQG is consistent with its underlying principles. Interpretation of SQGs must be properly linked to their derivation and narrative intent.

In the ideal case, SQGs would be able to unequivocally delineate between sediments that cause biological effects and those that do not; in other words, all sediments below the SQG would show no effects, while all those above would show effects (Figure 1A). In reality, the occurrence of biological effects does not show such a clearly delineated relationship. Instead, the distribution of biological effects generally shows a relationship characterized by ranges of chemical concentration where biological effects are rare, where cases of both effects and no effects are found, and where
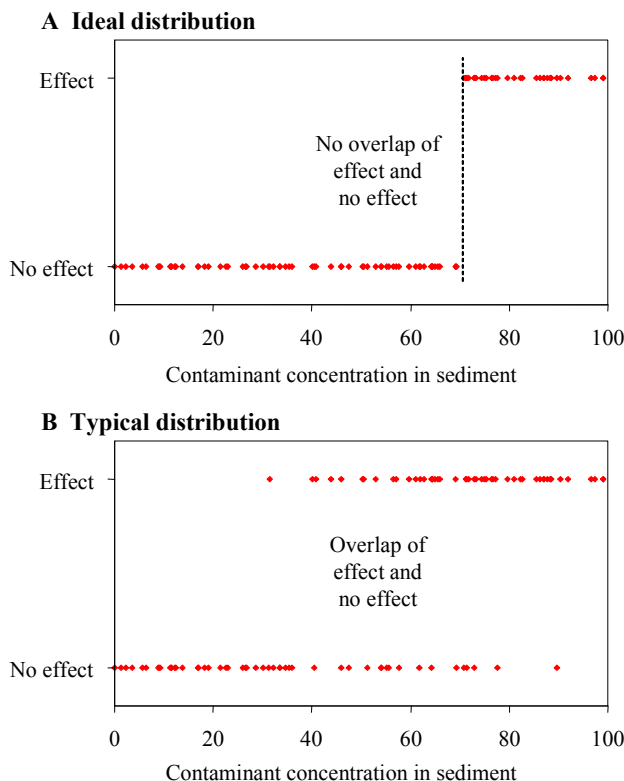
A Ideal distribution

B Typical distribution

Figure 1 Distributions of biological response data across a range of sediment contamination. Panel A shows an ideal case where there is no overlap between concentration ranges that are and are not associated with biological effects. Panel B shows a distribution common in sediment assessments, in which there is overlap in sediment contaminant concentrations associated with effects and no effects in benthic organisms.

biological effects essentially always occur (Figure 1B). Normalization to organic carbon or other characteristics that control bioavailability of sediment contaminants may affect the distribution of effect and no-effect data, but no normalization technique has yet come close to complete discrimination (e.g., Figure 1A), nor is it likely. Factors that cause overlap between effect and no-effect data are many but include contributions of other chemicals to effects, unaccounted for differences in chemical availability, differences in response among organisms, and errors in measurement of either chemical or response.

This generalized concentration–response model, with the probability of effects increasing with increasing chemical concentration, can be used as a framework to consider different SQG approaches (Figure 2). The probability of effects is low until it reaches a threshold-effect (TE) guideline. At the high end of the distribution is the probable-effect (PE) guideline, above which effects almost always occur. Between TE and PE lies a transition zone where adverse biological effects may or may not occur,
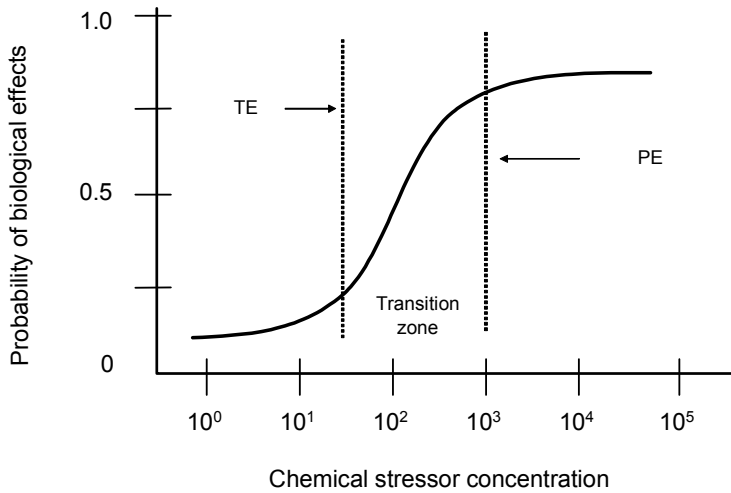
Figure 2 A generalized concentration–response model describing the TE and PE guidelines along a gradient of increasing probabilities of adverse biological effects for a single sediment-bound contaminant.

but within which the probability of adverse effects increases. SQGs can be thought of as vertical lines across this concentration–response curve. Different SQG derivation procedures are intended to represent TE, PE, or mid-range effect (ME) guidelines. A TE guideline is a contaminant concentration below which biological effects are predicted to occur in very few cases, whereas a PE guideline is intended to identify the contaminant concentration above which effects are predicted to occur in almost all cases. Intermediate to these would be ME guidelines, which identify contaminant concentrations within the transition zone wherein the probability of effects is substantively above background, but above which effects are not always observed.

In this context, chemically based numeric SQGs can be effective for identifying concentration ranges where adverse biological effects are unlikely, uncertain, and highly likely to occur. While represented in Figure 2 as probability of effect, a similar concept could be applied to the severity of the effect (e.g., reduced survival or reproductive impairment), which typically increases as the probability of toxicity increases.

While not all SQGs are derived from a concentration–response model like that in Figure 2, this paradigm can still be used to conceptualize differences among SQGs. As indicated previously, SQGs differ as where they would occur along the concentration–response continuum (e.g., vertical lines for TE and PE in Figure 2). In addition, SQGs can vary in the way that chemical concentration is expressed (the X-axis in Figure 2), such as dry weight concentration of a chemical, the organic carbon normalized concentration, or some expression of a mixture, such as total narcotic potency (Di Toro and McGrath 2000). SQGs can also vary in the way that biological effect is expressed (the Y-axis in Figure 2), such as lethality to amphipods versus benthic community response.

By definition, the existence of the transition zone in Figure 2 means that no guideline can unequivocally separate all sediments showing effects from those that do not. Nonetheless, SQGs are often employed such that they are expected to separate "toxic" and "nontoxic" samples. In such applications, the interpretation of exceeding a guideline varies according to where the guideline lies along the concentration–response continuum. If one uses a TE-type SQG to define "toxic" (or unacceptable), then errors will tend to be characterized as false positives; that is, some sediments will be classified as toxic when in fact they are not. On the other hand, selecting a PE-type guideline will result in predominately false negative errors, in which some sediments will be classified as nontoxic when in fact they are toxic. As one moves from a TE guideline toward a PE guideline, the chances of false positives decrease and chances of false negatives increase. ME guidelines can be expected to have intermediate rates of false positives and false negatives relative to TE and PE guidelines, respectively; whether this is more or less desirable depends on the management purpose. In some situations, it may be desirable to have a guideline that will identify almost all potentially toxic samples, even if many of those samples are not actually toxic (TE guideline). In other instances, it may be desirable to have a guideline that predominately identifies sediments that have a high probability of being toxic, even if means that some sediments that may be toxic will be missed. **In all cases, application of SQGs in a "toxic or nontoxic" context must be cognizant of the types and rates of errors associated with each type of SQG.**

Beyond the TE–ME–PE concepts, interpreting exceedances of SQGs must also recognize differences between empirically derived and mechanistically derived guidelines. Unlike most empirically derived SQGs, mechanistically based guidelines are intended to assess the effects of only those chemicals for which the guideline is derived; they address the question "is the concentration of this chemical (or chemicals) sufficient, by itself, to cause effects?" Although the likelihood of adverse effects is thought to be elevated when the guideline is exceeded, the likelihood of a sediment causing adverse effects when it exceeds an individual mechanistically based guideline cannot be predicted. This is because that guideline does not address the potential effects of other chemicals that may be present. For guidelines developed using data from field-collected sediments, which typically contain mixtures of chemicals, there is some implicit consideration of other chemicals present in the sediment, although the nature of this mixture effect is not explicitly quantified.

The issues associated with mixtures of contaminants in the environment are pervasive. **The occurrence of biological effects in field sediments is almost always associated with mixtures of sediment-bound contaminants. Empirically derived SQGs may be used to incorrectly attribute the adverse biological effects caused by a mixture of contaminants to a single contaminant. Conversely, mechanistically derived SQGs based on causality may underestimate the adverse biological effects attributable to a mixture of contaminants in sediment.**

Chemical interactions in sediments and their resulting biological effects are poorly understood. The simplest conceptual model is additivity, which predicts the effects of

a contaminant mixture as the sum of the effects of the individual contaminants. Additivity is generally thought to apply to chemicals with the same mechanism of action. Thus, if concentrations of chemicals with the same mode of action are normalized to a reference concentration like a LC50 (lethal concentration for 50% of test organisms) or a SQG, the sum of the normalized numeric SQGs may be logically used in the concentration–response analysis to predict the probability of biological effects. This additivity model is appropriate for SQG approaches that reflect adverse biological effects caused by specific contaminants (e.g., mechanistic SQGs such as those derived from EqP theory) and has been applied to polycyclic aromatic hydrocarbons (PAHs) (Swartz et al. 1995), narcotic chemicals (Di Toro and McGrath 2000), and metals (Ankley et al. 1996). At present, these additivity approaches are generally restricted to chemicals with similar modes of action, as there is no theoretically rigorous method for modeling or predicting the interactions among contaminants that act with different modes of actions.

Methods for aggregating empirically derived SQGs have also been proposed, such as sum, maximum, or average of SQG quotients (i.e., the ratio of chemical concentration in sediment to SQG). Although intuitively appealing, the theoretical basis for how best to do this is less clear than for mechanistically based guidelines. Individual empirically derived SQGs based on the co-occurrence of effects and chemistry (e.g., ERLs and PELs) may not be causally related to specific contaminants. Instead, empirical SQGs for individual contaminants indicate effects caused by the entire mixture of sediment chemicals. For this reason, adding empirical SQG-normalized concentrations across chemicals may be, in effect, incorporating additional conservatism regarding effects attributable to contaminant mixtures and may not be as justifiable as doing so for mechanistic guidelines. Calculation of average or maximum SQG quotient has the potential to provide more information on the nature of sediment contamination relative to effects, and there is evidence that these approaches improve the predictive ability of empirical SQGs (MacDonald, DiPinto et al. 2000; MacDonald, Ingersoll et al. 2000; USEPA 2000a; Ingersoll et al. 2001; Field et al. 2002). **Despite the technical uncertainties, in most cases the use of mixture models and mean SQG quotients improves the ability of SQGs to predict effects in the laboratory and in the field compared to evaluations based on exceeding single SQGs.**

In the context of Figure 2, the aggregation of multiple SQGs represents an alteration of the X-axis, the quantification of sediment contamination. Another issue affecting this X-axis is the definition and quantification of the bioavailable fraction of sediment contaminants. Normalizing contaminants to binding phases (e.g., PAHs to organic carbon, metals to AVS) theoretically provides a better consideration of bioavailability. Spiked sediment experiments, for both acute and chronic exposures, have demonstrated that the applications of these normalization methods improve the predictive ability of mechanistically based SQGs. However, there are additional factors known to affect bioavailability that have not yet been incorporated into SQGs, such as unusual carbon types (e.g., soot, ash, black carbon) or phases controlling metal bioavailability in oxic sediments. Interestingly, evaluations of empirically based SQGs have found that dry weight normalization often results in predictions of adverse biological effects that

are as good or better than the normalizations thought to reflect bioavailability (e.g., Barrick et al. 1998). The reasons for this apparent discrepancy remain unclear. Nonetheless, it seems an inescapable conclusion that at some level, the accuracy of SQGs will be limited by the ability to measure and incorporate the factors that account for bioavailability in sediments.

Returning again to Figure 2, the third way in which SQGs vary lies in the effects used to define the Y-axis. **The selection of data and biological endpoints used to derive SQGs can affect the relevance to protection of benthic communities and the inferences that SQGs provide regarding contaminant conditions in sediment.** Differences among SQGs often reflect differences in the sensitivity of the biological indicators used to document effects. SQGs may change with test species, life stages, response endpoints, test duration, and taxonomic composition of benthic communities. Some SQGs incorporate information from one or more aquatic toxicity databases and are intended to protect 95% of species from chronic effects (e.g., EqP using water quality criteria endpoints). Some empirical SQGs are based on extensive databases that include a great variety of effects of sediment chemicals on benthic species (e.g., ERL, ERM, TEL, PEL, and AET). Other SQGs are based exclusively on adverse biological effects on a single taxonomic group (e.g., ΣPAH for marine amphipods). The choice of biological indicators can significantly influence the relevance of different SQGs in sediment assessments as screening tools for evaluating the impact of sediment-bound contaminants on benthic organisms or other aquatic biota.

There are inherent limitations in the ability of any SQG approach to accurately and reliably indicate adverse effects on benthic communities, especially at sediment contaminant concentrations within the transition zone. The general expectation for SQGs is that these guidelines index the potential for effects on benthic communities. However, the connectivity between benthic community structure and function and the endpoints on which the SQGs are derived varies and is generally incomplete. A number of factors that affect organism response to contaminated sediments are not, or are not fully, captured by existing SQG approaches. For example, SQGs do not directly address avoidance of sediment contaminants by sediment-dwelling organisms or predator–prey interactions with regard to contaminant trophic transfers (Von Stackelberg and Menzie 2002). There have been some studies that compared SQGs with field surveys of contamination, toxicity, and benthic community structure (e.g., Swartz et al. 1994; Ingersoll et al 1997). These studies generally show a correspondence between increasing contamination and the disappearance or reduction of sensitive benthic taxa in sediments where a numeric SQG has been exceeded. However, impacts on benthic communities have also been observed at contaminant concentrations well below SQGs derived using laboratory toxicity data. Conversely, there have been some cases when adverse effects on benthos were not observed in sediment where one or more SQGs were exceeded.

Despite the uncertainties involved in different SQG approaches, chemically based empirical numeric SQGs derived using different methods and assumptions appear to converge, suggesting important underlying relationships relative to causality. Diverse scientific and statistical methods have been used to

quantify the potential for adverse biological effects at different contaminant concentrations. All of the different methods provide some information on the relationship between sediment contamination and adverse biological effects; however, none of the approaches by themselves is entirely satisfactory. The recent development of a "consensus-based" approach (Swartz 1999; MacDonald, DiPinto et al. 2000; MacDonald, Ingersoll et al. 2000) indicates that, for some contaminants, estimates of TE and ME guidelines for the most widely used chemically based empirical numeric SQGs (e.g., ERLs, ERMs, TELs, PELs, and probable-effects concentrations [PECs]) are within 1 order of magnitude or less. The mechanistic and empirical SQGs for total PAHs are also reasonably consistent, although the individual chemical SQGs are not (Di Toro and McGrath 2000).

Research associated with the development of SQGs has made large contributions to the assessment of sediments and has identified many of the key factors that must be considered. That said, SQGs are based on a variety of assumptions and endpoints, and none can separate with perfect accuracy sediments that do and do not cause biological effects. For that reason, the application of SQGs to decision-making processes must always be consistent with the derivation and uncertainties of those SQGs. While continued development of SQGs will undoubtedly lead to further improvements, there are inherent limitations to the ability to quantify sediment contamination (including bioavailability) and biological effects that will probably always maintain some level of uncertainty, for example, the transition zone shown in Figure 2. **While improvements in SQGs may narrow the range of concentrations where effects are uncertain, SQGs should be incorporated into a larger WOE framework to better evaluate the degree of adverse biological effects in sediments that fall within the transition zone of the concentration–response model.**

Workgroup 2
Predictive ability of SQGs

The quantitative extent to which chemically based numeric SQGs are predictive of the presence, or absence, of toxicity of contaminated sediment to sediment-dwelling organisms or to higher trophic level organisms is a critical concern among scientists and agencies evaluating the application of one or more numeric SQG approaches in assessments of sediment quality. Users of the different SQG approaches should understand how well various SQGs predict the presence or absence and extent of toxicity in sediment samples. **The predictive ability of various SQGs to represent the potential for effects or no effects of contaminants on organisms in freshwater, estuarine, and marine environments were defined by the workgroup as the probability of observing the presence or absence of effects within incremental ranges of sediment contaminant concentrations as defined by SQGs based on the specific endpoints and benthic taxa evaluated.** Results of these evaluations are typically expressed as the percentage of samples expected to be affected (e.g., % toxic samples); however, some evaluations have also been based on the degree of the response (e.g., % mortality). In this context, chemically based numeric SQGs are defined as the concentration of sediment-associated contaminants

that is associated with a high or a low probability of observing adverse biological effects or unacceptable levels of bioaccumulation, depending on its purpose and narrative intent.

A wide range of published laboratory and field studies in freshwater, estuarine, and marine environments encompassing over 8000 sediment samples indicate that empirical effects-based SQGs can be used to assess the probability of observing effects with known statistical levels of confidence. Among the different empirical SQG approaches, the incidence of effects, or the degree of the response, increases with increasing sediment contamination. These comparisons are primarily based on the presence or absence of toxicity in 10-day amphipod tests using whole (i.e., bulk) sediments. Among all of the sediment toxicity data sets examined, the lowest incidence of adverse biological effects (less than about 10%) was identified at contaminant concentrations less than the low-range of empirically derived SQGs (i.e., below a mean quotient of about 0.1); the highest incidence of toxicity (greater than about 75%) was observed at contaminant concentrations above the upper-range of empirical SQGs (i.e., above a mean quotient of about 1.5 to 2.3).

For mechanistic-based EqP SQGs, the predictive ability was demonstrated initially on the results of 10-day toxicity tests using single contaminants followed by mixtures of metals and PAHs in spiked sediments (Di Toro, Mahony et al. 1991; Di Toro, Zarba et al. 1991; Swartz et al. 1995; Hansen et al. 1996). For organic chemicals the range of uncertainty for predicting acute toxicity in sediment (i.e., the LC50 concentration) appears to deviate from the best estimate by about a factor of 2 in concentration (USEPA 1993, 2000b). For metals, the SEM–AVS method initially predicted only the lack of toxicity, with no exceptions. The more recent excess organic carbon normalized SEM method predicts the presence of toxicity as well, with an uncertainty range (25% of the samples) within which both toxicity and no toxicity are found (USEPA 2000c).

Chronic toxicity tests conducted with freshwater sediments and evaluations of benthic community structure in estuaries indicate that responses occur in ranges below those where an empirical SQG would predict toxicity in 10-day toxicity tests. These differences are about 6-fold lower for the chronic responses in freshwater toxicity tests and about 10-fold lower for estuarine benthic community responses. However, the relative influence of natural factors (e.g., grain size or total organic carbon) versus chemical-induced toxicity on benthic communities in estuaries has not been fully quantified. In a limited number of chronic metals and metals mixture experiments, the excess organic carbon normalized SEM method predicts the presence or absence of effects at the lower bound (100 μmol/g of organic carbon) of the acute range in virtually all of the tests conducted to date. Use of chronic laboratory toxicity tests and controlled field-colonization studies and mesocosm studies that control for confounding factors are needed to better estimate the impacts observed on benthic communities exposed to contaminated sediments.

Sediment quality guidelines based on partitioning theory attempt to causally relate sediment concentration to toxicity. The availability and success or failure of the

different mechanistic-based SQG approaches depends on the adequacy of the partitioning model and its parameters and the assumption that exposure is either from pore water or sediment particles. Empirical SQGs for total PAH and total PCBs are similar to comparable guidelines derived using theoretical approaches. This concordance suggests that these mixtures may be causally implicated in the toxicity observed in a substantial number of sediments. However the results for metals are much different, with the SEM–AVS model predicting metal toxicity only in laboratory-spiked sediments with either very high metal concentrations, or in laboratory-spiked sediments containing no AVS and low organic carbon concentrations.

Importantly for both laboratory toxicity tests and benthic community studies, an incremental increase in effects has frequently been observed with an incremental increase in contamination as defined by different SQG approaches. However, direct measurement of toxicity in the laboratory and/or benthic community impacts in the field are required to determine if an individual sample with moderate contamination is toxic or nontoxic.

Many confounding factors can potentially cause spurious conclusions regarding the relationship between sediment-bound contaminants and occurrence of toxicity or nontoxicity. Confounding factors can be biological in that organisms are not exposed to whole sediment or pore water and, thus, may underpredict toxicity (e.g., behavior). Confounding factors can also be physical, whereby one or more sediment characteristics drive a biological change that co-varies with contaminant concentrations (e.g., grain size and total organic carbon). Additionally, the chemical state of the contaminants (e.g., paint chips, lead shot, tar balls, and metal ore), the nature of the sediment matrix (e.g., black carbon, peat, and wood chips), and laboratory artifacts (e.g., water temperature and homogenization or compositing of heterogeneous sediments) can also reduce the predictive ability of various SQG approaches.

Bioaccumulation, in and of itself, is not an effect, and none of the effects-based SQGs (e.g., ERMs, EqP, and AETs) were designed or intended to be predictive of toxic residues for sediment-dwelling organisms as a result of bioaccumulation from sediment. However, there are approaches that may allow connections to be established between measures of effect and no-effect tissue residues that lead to new bioaccumulation-based SQGs. These connections can be derived from tissue-residue effects values, (e.g., median lethal residue [LR50] or from regional background tissue reference values derived using empirically derived biota-sediment accumulation factors (BSAFs) or measured BSAFs for nonionic organic compounds. Current evidence suggests that BSAFs for nonionic organic compounds are relatively consistent on a contaminant-specific basis; however, site-specific geochemical factors may cause deviations from the general bioavailability trends. **While bioaccumulation-based SQGs have been proposed and, in some cases, implemented as chemical-specific numeric SQGs, the predictive ability of this approach has yet to be adequately validated by field experimentation.** Further, establishing connections from sediments through bioaccumulation and trophic transfer should be evaluated on a site-specific basis because of the unique characteristics of different ecological receptors and food web interactions in freshwater, estuarine, and marine environ-

ments. Thus, the probabilities of adverse biological effects due to bioaccumulation at higher trophic levels cannot be readily predicted using current SQG approaches in the absence of ERA.

Workgroup 3
Use of related sediment assessment tools

The primary focus of sediment assessments should be determining the potential for biological impairment. The simplest of all assessments might include the use of a single LOE such as a set of toxicity tests, a benthic community survey, or the use of SQGs to make a decision regarding impairment. However, the initial screening of contaminated sediments may be insufficient for making decisions due to uncertainty. In such cases, practitioners may opt to pursue additional LOE that improve certainty. This may involve utilizing a WOE approach or a more formal ERA, and can include developing a conceptual model, understanding organism linkages, selecting measurement and assessment endpoints, characterizing exposure, and performing a risk characterization. Regardless, the WOE approach must be clearly defined a priori with regard to how complementary and contrasting LOE will be evaluated.

There are at least 4 key LOE that should be developed: 1) sediment contaminant chemistry and geochemical characteristics, 2) benthic invertebrate community structure, 3) sediment toxicity, and 4) bioaccumulation and biomagnification data (Grapentine et al. 2002). The integration of these and other sediment assessment tools should 1) improve the ability to establish impairment of biological systems and to establish reference or benchmark conditions, 2) ensure that sediment assessments include consideration of important ecological and food web linkages, 3) provide guidance on how to formulate LOE in an overall WOE approach to assist in making a decision as to whether or not sediment contamination has resulted in biological impairment, and 4) help to identify potential sources of contaminants introduced into sediments.

The USEPA (1992) ERA framework provides the option of either making a decision with the current level of uncertainty or performing more tests to reduce uncertainty (Ingersoll et al. 1997). Such assessments may not be required in every circumstance as they may be burdensome, delay action and increase cost. **However, each of the 4 components of the current ERA paradigm—1) problem formulation, 2) exposure characterization, 3) effects characterization, and 4) risk characterization—can help guide managers toward an appropriate level of assessment using appropriate tools (Ingersoll et al. 1997).**

The problem formulation stage is arguably the most critical stage of the sediment assessment process. It is during this stage that the appropriate tools for screening and assessing the state of biological impairment should be selected. To assist in making decisions regarding the potential for biological impairment where more than a single LOE is needed, an iterative approach to problem formulation is recommended for collecting data within the sediment ERA framework. The importance of the problem formulation stage and, in particular, the conceptual model development and character-

ization of reference (i.e., background) conditions are frequently overlooked in sediment assessments and lead to incomplete or overly simplified ERAs that may not reflect an accurate assessment of the potential for biological impairment. To avoid situations where sediment quality is mischaracterized as either over- or underprotective, the following study design elements should be considered:

- Identification of relevant exposure pathways associated with low or high flow, ground water upwelling, surface-water down welling, sediment (surface and deep), and food;
- Selection of an appropriate model relating stress and response;
- Selection of appropriate reference locations, background stressor levels, and natural stressors;
- Selection of optimal measurement and assessment endpoints;
- Laboratory and field quality control procedures to ensure observations with the lowest possible variability;
- Environmental sampling to ensure adequate statistical power to detect pre-specified biological changes in responses and spatial and/or temporal characterization; and
- Selection of appropriate statistical methods for LOE analyses and characterization of effects using biologically based methods that integrate resident biota and sediment toxicity into a WOE matrix.

Sediment assessments are frequently based on temporal or spatial data that do not adequately reflect co-occurrence of multiple contaminants or stressors, or the proper exposure pathway considerations. Exposures should be assessed in a manner that reflects species biological and behavioral characteristics, and the critical time periods and locations that reflect co-occurrence between stressor and receptor. With regard to characterization of ecological effects, adverse biological effects in a water body are defined as those that can impact populations such as development, reproduction, and survival. Sediment assessments based on a single LOE or incomplete LOE often overlook the potential for long-term effects or the trophic transfer of contaminants that result in food chain effects.

A relatively simple risk characterization model, involving the comparison of sediment chemistry measurements to one or more SQGs, can be improved by incorporating probabilistic models into the integration of exposure and effects LOE. This, in itself, does not necessarily increase the complexity of the underlying hazard quotient model, but simply adds a useful layer of quantitative interpretation. When confronted by conflicting multiple LOE, the first task in risk characterization is to determine how to interpret apparent conflicting information between different LOE (Grapentine et al. 2002). For example, the results of sediment toxicity tests may indicate no effects when comparison of sediment chemistry with an SQG suggests that toxicity should occur. This may be due to insufficient sensitivity on the part of the toxicity test organism or to overly conservative estimates used to derive the SQG. A transparent, and preferably quantitative, approach is required to integrate the information derived from multiple LOE.

Table 1 Summary of available sediment assessment methods for direct measures of exposure and adverse biological effects on benthic organisms and other aquatic biota.

Method	Measurement endpoint		Applications and strengths	Limitations
	Exposure	Effects		
Methods typically used in Tier 1 sediment assessments				
Toxicity test — lethal	✓	✓	• Provide direct measures of short-term toxicity • Relatively low cost • Based on standardized methods • Can be used in different matrices	• Lab-field extrapolation can be difficult • Does not predict long-term effects • May not provide the most sensitive measures • May not predict nonlethal effects
SQGs	✓	✓	• Generic • Widely available • Identify effects or no effects of contaminants in sediments	• May be imprecise in predicting effects • Methods are not standardized • Lack consistency across regulatory agencies • Lack site-specific data
Physicochemical profiling	✓	–	• May identify co-stressors • Provides data to assess bioavailability	• Cost may be high
Exposure surrogates	✓	–	• Provides an indirect measure of in situ exposure and/or uptake • Easy to use	• May not be predictive of exposure in active organisms • Does not measure dietary exposure

Table 1 continued

Methods typically used in both Tier 1 and Tier 2 sediment assessments

Method	Measurement endpoint		Applications and strengths	Limitations
	Exposure	Effect		
Contaminant profiling	✓	–	• Provides direct measures of contaminants • For routine chemicals, cost is relatively low • Critical for source identification • Based on well-established methods	• May not reflect bioavailable fraction • Prohibitive to measure and identify all chemicals in most systems • Methods lacking for new chemicals • Limited to chemicals measured
Toxicity test–sublethal	–	✓	• Provide direct measures of sublethal toxicity • Relatively more sensitive • Some standardized methods are available • Can be used in different matrices	• Lab-field extrapolation can be difficult • Typically involves increased level of difficulty and cost • May not provide the most sensitive measures
Biomarkers/ histopathology	✓	✓	• Can be diagnostic (chemical specific) • Can provide early indication of effects • Histopathology disorders are relevant to demersal fish	• Unclear association with organism-level and/or community-level effects • Interpretation is often difficult • May indicate noncontaminant effects

Table 1 continued

Method	Measurement endpoint		Applications and strengths	Limitations
	Exposure	Effects		
Methods typically used in Tier 2 sediment assessments				
Microcosm test (multispecies)	✓	–	• Represents Increased relevancy to field conditions • Can allow discrimination of confounding factors • May allow better assessment of long-term effects	• Typically associated with a high level of difficulty and cost • Interpretation of results may be difficult • Reproducibility may be difficult
Toxicity identification evaluation (TIE)	✓	✓	• Links specific chemical classes with effects (causality) • Can be used with different matrices	• Field validation is often needed • Sensitivity limited to toxicity test and/or organisms used • Limited by precision and accuracy of chemical analyses • Typically associated with an increased level of difficulty and cost
In situ testing	–	✓	• Provides direct measure of effects in field • Reflects realistic exposure conditions • Can identify co-stressors	• Logistical challenges in deep or high energy systems • Lack of control over natural variables including nutritional quality of food • Potential organism stress during transport and deployment • Sorption of the chemical to test chamber surfaces or screens, and fouling of the screens
Mapping (GIS)	✓	–	• Facilitates visualize spatial (pattern) and co-occurrence • Can indicate spatial extent of contamination • Can be used with management decisions	• Provides indication of correlation, not causation • May require a large amount of data

Table 1 continued

Method	Measurement endpoint		Applications and strengths	Limitations
	Exposure	Effects		
Methods typically used in Tier 2 sediment assessments				
Probabilistic models	✓	✓	• Uses all available data • Provides quantifiable interpretation of risk and uncertainty associated with assessment	• May require a large amount of data • Results may be difficult to communicate • Robustness of model dependent upon quantity and/or quality of data used • Selection of data distributions can be difficult, and are typically subject to best professional judgment
Environmental fate models	✓	–	• Provides prediction of exposure • Can provide understanding of complex processes controlling exposure (e.g., bioavailability)	• Model must be calibrated to system being assessed • Model assumptions must be recognized and adhered to • Predictions often need validation with empirical data • Robustness of model is dependent upon quantity and/or quality of data used
Food Web Models	✓	✓	• Provides linkage of sediments to higher trophic level effects • Facilitates definition of direct and indirect ecosystem effects	• Model must be calibrated to system being assessed • Model assumptions must be recognized & adhered to in the model • Predictions often need validation with empirical data • Robustness of model is dependent upon quantity and/or quality of data used

A tiered approach to data collection is recommended whereby screening type tools are used in Tier 1 and tools that enable detailed characterization of either exposure or effects, or those that provide interpretation of both exposure and effects data in sediment assessments are used in Tier 2 (Table 1). Tier 1 tools are typically low in cost, easy to use, and may yield information that tends to be general, rather than specific. Tier 2 tools may be more challenging or expensive to use, but often provide more information and greater specificity. Exposure and effects data form the two major LOE in a Tier 2 assessment. **A critical requirement for selecting the appropriate sediment assessment tools is to ensure that the appropriate tools are matched with the appropriate investigation questions.** This will depend, in part, on the stage and complexity of the assessment process; earlier stages and simpler assessments may require only relatively simple tests that facilitate screening (e.g., Tier 1 tests in Table 1). Later stages, such as those deemed necessary following screening, may require more complex tests or tools to better delineate the extent of sediment impairment (e.g., Tier 2 tests in Table 1). The selection of the appropriate tools allows for development of defensible LOE that can be used to determine sediment quality and the potential for biological impairment.

The importance of characterizing both natural and anthropogenic stressors and exposure conditions at multiple levels of the food web is often recognized in principle, though seldom accomplished in practice during the ERA process. Traditional assessment tools such as SQGs do not always provide predictive power when multiple contaminants and stressors are present and may result in unacceptably high levels of false positives and negatives.

It is apparent that more effective ecological assessment approaches are needed to link the magnitude, frequency, and duration of exposure with biological effects and to provide better definition of when adverse ecological effects occur. A more quantitative and logical framework for using the WOE process for sediment assessment is necessary where substantial uncertainty exists. Such a framework should include the following critical elements: conceptual model, linkage of "exposure" and "effects" and conceptual model components, characterization of key natural and anthropogenic stressor exposure profiles, appropriate reference characterization and comparison methods, appropriate quantification methods used to integrate LOE, critique of advantages and limitations of each LOE used, evaluation of each LOE versus causality criteria, and combining the exposure and effects LOE into a WOE matrix for interpretation, showing causality linkages in the conceptual model. Appropriate statistical approaches (e.g., regression, analysis of variance [ANOVA] and multivariate methods) should be used within each LOE to define reference conditions and potentially impaired conditions. This approach is most useful when incorporated into the initial and final study design stages (e.g., the problem formulation and risk characterization stages of an ERA Burton et al. 2002).

In order to establish causality, a multistep process is necessary, using diagnostic protocols and weighing the strength of evidence that supports each potential cause. This can be done via 7 causal considerations: co-occurrence (spatial correlation);

temporality (temporal correlation); effect magnitude (strength of link); consistency of association (at multiple sites); experimental confirmation (field or laboratory); plausibility (likelihood of stressor-effect linkage); and specificity (stressor causes unique effect). The results of these expert judgments can be summarized in a tabular decision matrix, for example, by conveying judgments of different LOE into a hierarchal scale (e.g., 1 to 3 or 4, or "+" and "−" values). Using a ranking scheme, logical comparisons of multiple LOE, comprising both "exposure-" and "effects-" based LOE can be made. **If the WOE process adequately characterizes and links stressor exposure with biological impairment using relationships to reference (i.e., background) conditions, then uncertainty will be better understood in decisions concerning the source, occurrence, and severity of sediment-related impairment of aquatic biota.**

Workgroup 4
Use of SQGs in sediment assessment frameworks

Chemistry data have been used for decades by state and federal regulatory agencies in different countries to assess and manage contaminated sediments. SQGs have been recently developed to better define the relationship between sediment chemistry and toxicity, providing regulatory agencies with additional insight into the importance of sediment chemistry data. Increasing interest in the development of risk-based sediment assessment frameworks to guide assessments and management decisions has led to questions concerning the role of SQGs within a sediment assessment and management process that makes use of multiple LOE to reach management decisions based on a WOE. At present, nearly 20 sediment assessment frameworks have been proposed or used by regulatory authorities in different countries. These frameworks include several key characteristics that should be preserved or refined in the future, including the use of multiple tiers, multiple LOE (including both chemical and biological information), and an iteration process that facilitates refinement of an assessment as data are collected and analyzed. With respect to the use of SQGs in current sediment assessment frameworks, there is strong interest in having a range of SQGs that can be used to assess and classify sediments.

An assessment framework provides a structure and process for conducting a sediment assessment that leads to a management action. As such, a framework that meets programmatic objectives delimits appropriate uses for SQGs, as part of a risk-based evaluation. Environmental regulatory agencies have been encouraged by the scientific community to develop logical and orderly sediment assessment frameworks and to ensure that assessments are comprehensive, transparent, and consistent. A sediment assessment framework should be structured to ensure that any evaluation that follows the steps of the framework is comprehensive and complete in its consideration and analysis of present and future exposures, effects, and human and ecological risks at the site of concern. **All routes of exposure and types of effects will not occur at every site; however, a comprehensive assessment framework, if followed, should require consideration of the likelihood for all possible routes of exposure and the potential for adverse biological effects to ensure that required or**
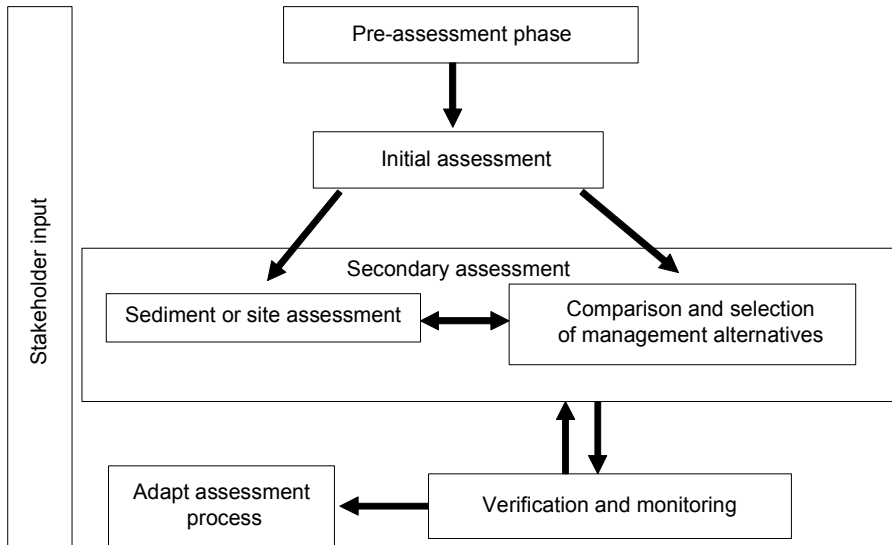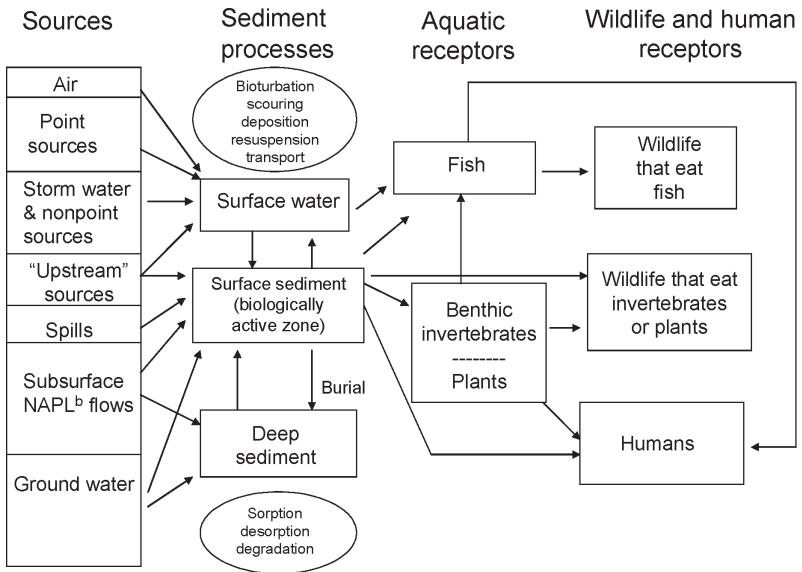
Figure 3 A basic risk-based sediment assessment framework

important site-specific environmental factors are not omitted from the evaluation process. Assessment frameworks should also provide a measure of transparency to sediment investigations and management, as well as facilitate meaningful participation in the assessment and decision-making process by scientists, regulatory agencies, and representatives of affected communities. Active stakeholder involvement throughout the assessment process is essential to ensuring that the results of the assessment can be successfully applied within the decision making process. Finally, development and application of an assessment framework will facilitate consistent application of the assessment and management process at different sites.

The sediment assessment framework proposed in Figure 3 incorporates the key elements typically specified in various international frameworks and provides a structured, defined role for SQGs within a risk-based evaluation. The framework is composed of 5 major phases of activity: pre-assessment; initial assessment; secondary assessment; verification and monitoring; and process adaptation. Assessment frameworks are commonly structured with tiers. Because the amount of information required for management decisions will vary from site to site, assessments conducted in a tiered fashion permit time and cost savings when initial assessment methods are sufficient to reach decisions. A tiered structure that allows for iteration also facilitates more focused analysis through subsequent tiers as new information becomes available that necessitates refinements to either the conceptual model, management objectives, or both.

During the pre-assessment phase of an evaluation, programmatic and/or regulatory goals or objectives must be clearly defined and procedural and technical constraints, if any, understood to ensure that the evaluation results in an assessment that is sufficient

Sources
- Air
- Point sources
- Storm water & nonpoint sources
- "Upstream" sources
- Spills
- Subsurface NAPL[b] flows
- Ground water

Sediment processes
- Bioturbation scouring deposition resuspension transport
- Surface water
- Surface sediment (biologically active zone)
- Deep sediment
- Burial
- Sorption desorption degradation

Aquatic receptors
- Fish
- Benthic invertebrates --------- Plants

Wildlife and human receptors
- Wildlife that eat fish
- Wildlife that eat invertebrates or plants
- Humans

Reminder: all of the above have specific spatial and temporal scales

A related fundamental requirement of sediment assessments concerns identification of meaningful sediment and site-specific questions a priori and the selection of specific LOE and assessment tools. Some assessment questions cannot be addressed with the current suite of available SQGs. For example, a common assessment question "Does sediment contain bioaccumulative chemicals that pose an unacceptable risk to upper trophic levels?" cannot be addressed by comparing bulk sediment chemistry measurements to the most widely used effects-based SQGs. **Considerable effort should be devoted to formulating and refining specific and detailed questions that must be answered to reach conclusions about the presence and magnitude of risk.**

There is strong merit for using SQGs and other sources of information in the initial assessment phase to identify sediments requiring no further evaluation because the sediments pose little potential for adverse biological risks (Table 2). In cases where comparisons of bulk chemistry data to SQGs results in ambiguous answers to the assessment questions concerning the presence of unacceptable risk, the assessment should proceed to a secondary sediment or site assessment after revising, as necessary, the list of contaminants of potential concern, the conceptual model, and the assessment questions.

Within the context of a recommended sediment assessment framework (Figure 3), proceeding to a secondary sediment or site assessment includes: 1) defining measurement and assessment endpoints; 2) selecting LOE within 3 general categories (assessing direct exposure or effects in the water column, assessing direct exposure or effects to the benthos, and assessing indirect exposure and effects through contaminant trophic transfer); 3) selecting and applying assessment tools within the chosen LOE; 4) analyzing the collected information to reach conclusions based on a WOE; and 5) revising the conceptual model to identify remaining data gaps or to communicate conclusions about risks. When assessment questions have been satisfactorily addressed and conclusions reached concerning the extent and magnitude of risks, the framework transitions to comparison and selection of management alternatives. This phase involves 1) listing the practical management alternatives, 2) comparing the risks associated with implementing those alternatives, 3) comparing the costs of implementing the alternatives, and 4) apportioning the sediment at a site among the selected alternatives.

It is evident from the available scientific literature that there are no zero-risk options for managing contaminated sediments. **Given that there are no zero-risk options for managing contaminated sediments, comparative risk analysis methods should be developed and used to evaluate and select management alternatives.** Effective risk management requires comparing the risks and costs associated with the full spectrum of available management alternatives. This comparative approach requires interaction and iteration between sediment or site assessment activities and analyses conducted as part of the comparison and selection of management alternatives. Effective sediment management requires developing monitoring strategies to verify the accuracy of risk predictions made during sediment assessment and the appropriateness of management decisions based on these

Table 2 Management reasons for performing a sediment assessment and the role of SQGs in different sediment management scenarios.

Reason for sediment assessment[a]	Role for SQGs?[b]	Specific role and comments
Mapping spatial patterns	Primary	SQGs can be used to address relative patterns of contamination, including probable no effect and possible effect concentrations in sediments
Measuring temporal trends	Primary	
Determining condition of populations and communities	Secondary	As part of an ERA and/or in a tiered assessment scheme, SQGs are appropriate when used in conjunction with other tools
Estimating ecological risks, including bioaccumulation	Secondary	
Screening the suitability of proposed use or development	Secondary	
Assessing impacts of sediment dredging and/or management	Secondary	
Estimating human health risks and evaluation of biomagnification	None	SQGS have not been developed for this purpose
Determining sediment stability and transport	None	SQGs are not relevant
Remediation and restoration objectives	Primary	In cases of simple contamination where adverse biological effects are likely, 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 investigation
	Secondary	As part of an ERA and/or in a tiered assessment scheme, SQGS are appropriate when used in conjunction with other tools
Long-term post-remediation monitoring	Secondary	As part of an ERA and/or in a tiered assessment scheme, SQGs are appropriate when used in conjunction with other tools. However, SQGs alone do not address all possible monitoring needs such as human health, exposure routes, or functional aspects of the ecosystem.

[a] Adapted and modified from GIPME (2000)
[b] "Primary" can be used alone for management purposes; "Secondary" should be used with other assessment tools. In most cases, site-specific information should be generated to supplement the use of SQGs in sediment assessments.

predictions. Such a verification step also provides an opportunity for strengthening the assessment process and framework.

Apportioning contaminated sediment among more than one management alternative requires developing a logic or metric for using the information derived from different LOE developed during the assessment to rank the risks associated with contaminant conditions in the sediment. Successful sediment assessments will make use of a variety of chemical, biological, and physical data and risk assessment tools as a basis for management decisions. **Making scientifically credible decisions about managing risks posed by contaminated sediments requires using multiple LOE. Generally, no single LOE will provide sufficient information for effective decision-making.**

Successful implementation of a sediment assessment framework includes provisions for frequent communication with different stakeholder groups. It is particularly important to solicit stakeholder input when the questions that will ultimately drive sediment assessment activities are being formulated and management alternatives are being evaluated.

Workgroup 5
Addressing sediments in different aquatic environments

There are physical, chemical, and biological factors in the environment that complicate and introduce uncertainty into the derivation and application of SQGs and other sediment assessment tools. Sediments are heterogeneous and dynamic. Important physical and chemical properties, such as grain size, sulfide levels, organic carbon type, and content, may vary at small scales (millimeters) or large scales (estuaries) within one assessment area. Organisms' exposures to sediment-associated contaminants can occur by different routes, such as via sediment-water interface, pore water, direct contact, or ingestion. Biological factors reflecting species-specific differences in physiology, biochemistry, and behavior result in varying tolerances, acclimation, or adaptation, that result in different levels of adverse biological effects of contaminants. In addition, sediments tend to be contaminated by mixtures of chemicals whose potential interactions are not well characterized and whose bioavailability can be variable and challenging to predict. None of these factors are "fatal" to the derivation or application of SQG approaches or to other sediment assessment tools.

In general, several LOE are needed to properly evaluate contaminated aquatic environments. The absence of information or inadequate appreciation of the variation in any one of the following areas will detract from a complete understanding of the aquatic system relative to the occurrence and potential effects of contamination:
- Nature and extent of contamination;
- Expected or acceptable diversity and abundance of benthic biota in the absence of contamination;
- Bioavailability, bioaccumulation, and effects of contamination (the potential for chronic, as well as acute effects) on aquatic organisms;
- Stability of sediments and contaminants (fate and transport); and

- Risk of contamination to aquatic biota and associated resources.

There are a number of tools (i.e., not including specialized studies such as toxicity identification evaluations [TIEs], biodegradation studies, exposure surrogates, or modeling) available to obtain some or all of this information (Table 1); each tool has its own inherent strengths and weaknesses:

- Numeric SQGs;
- Sediment toxicity tests (chronic as well as acute);
- Resident exposed communities (i.e., not necessarily restricted to the benthos);
- Bioaccumulation; and
- Biomarkers and/or histopathology.

These tools should be applied as needed to meet the objectives of the sediment assessment and as appropriate to the specific environment. In this regard, assessment frameworks (Figure 3) and conceptual diagrams (Figure 4) are needed to help appropriately apply these tools and to determine that appropriate exposure routes and site-related taxa are considered.

At present, chemical analysis of whole sediment is an adequate estimate of exposure; however adjustments to account for bioavailability or chemical speciation can improve exposure estimates. The potential for adverse biological effects to aquatic organisms are reasonably measured using benthic community analysis (e.g., diversity, abundance, and presence or absence of key species), analysis of contaminant residues in tissue, and toxicity testing of appropriate, representative taxa.

SQGs alone are sometimes, but not always, sufficient for management decision-making. As detailed in Table 2, there are 10 management reasons for sediment assessments. In 5 cases, SQGs should be used in a WOE approach with other tools; in 2 cases, SQGs can be used alone; in 2 other cases, SQGs should not be used at all; and, in one case, SQGs can be used either alone or in a WOE approach with other tools, depending on the circumstances.

Different tools are needed to characterize sediment quality in different environments. The characteristics of the 5 basic sediment assessment tools described above, along with 3 general types of aquatic habitats that are typically encountered were reviewed (Table 3). It is apparent that, with the possible exception of biomarkers, sediment assessment tools are not equally applicable to evaluations of different aquatic habitats. Also evident is the need for additional standardized methods and procedures to further validate SQGs in estuarine and stream habitats.

Table 4 summarizes the issues that differentiate various aquatic environments and provides general suggestions for conducting sediment assessments in different depositional habitats in addition to estuarine and stream or other highly erosional systems. Cases where there is overlap between habitats (e.g., wetlands in estuaries; watershed systems combining erosional and depositional ecosystems) require consideration of combined issues with consequent possible modification of the suggestions for the different environments. There is a complex interplay between physicochemical and biological components that will dictate some level of uncertainty regarding contami-

Table 3 Characteristics of different[a] sediment assessment tools for describing sediment quality in depositional marine, freshwater, estuarine, stream and other erosional aquatic environments.

Tools	Depositional marine and freshwater	Estuarine	Streams and other erosional ecosystems
Sediment chemistry including numeric SQGs	Contaminant analyses based on sediments; current SQGs usually relevant	Contaminant analyses based on sediments; current SQGs should be further validated	Contaminant analyses should include biofilm; current SQGs may not be relevant for riffle or erosional environments
Toxicity tests	Generally conducted in containers with sediments—standardized methods are available	Generally conducted in containers with sediments—additional tests required—standardized methods available, but may not be applicable to all situations	Conducted with biofilm; may be necessary to provide water current in test chambers during exposure.
Bioaccumulation tests	Organisms exposed to contaminants in sediments, tissue levels measured in higher trophic levels		Organisms exposed to contaminants in biofilm
Biomarkers	Dependent on the exposure; can involve field or laboratory measurements; similar tests can be applied to all habitats		
Resident aquatic community structure	Samples collected directly from benthic habitats		Samples collected from riffles

[a] Does not include specialized tools such as TIE, biodegradation studies, exposure surrogates or modeling. Nor does it include frameworks such as ERA.

nant bioavailability and effects, so sensitive diagnostic tools may be especially valuable in validating predictive models. Thus, it is critical that the right assessment tools be selected to match the ecological system being evaluated.

Research Needs

The development of a uniformly accepted set of SQGs presents the major scientific challenge that most likely will be impossible to achieve. Contemporary understanding of several fundamental scientific principles such as the definition of bioaccumulation potential and sediment toxicity indicative of unacceptable, or significant, adverse biological effects underscores the complexity of any detailed assessment of contaminated sediments. Nevertheless, it may be possible to define a protocol, based primarily on biological testing and ERA, that could be consistently used to evaluate

Table 4 Major issues and suggested approaches for conducting sediment assessments and using SQGs in different aquatic habitats.

Habitats	Issues	Suggestions
Depositional: Lakes and ponds	SQGs generally appropriate. Thermal stratification, dynamic bioavailability, exotic species, seasonal anoxia.	• Design sampling based on seasonal differences • Do not exclude the influence of exotics species if present
Depositional: Low gradient rivers and streams	SQGs generally appropriate. Highly modified habitat, dynamic sediment-water interface, high total organic carbon (TOC), complex mixtures in urban systems, multiple diffuse sources in watershed, high recreational and commercial use.	• Further verify SQGs with site-specific data, develop site-specific SQGs, if needed and if practical • Define different ecoregions if present • Design sampling based on upstream and downstream comparisons if appropriate
Depositional: Wetlands	SQGs may not be specific for this environment. High organic matter and sulfides, high benthos patchiness.	• Focus on WOE evaluation and not individual LOE because of uncertain bioavailability • Assess bioaccumulation and biomagnification
Depositional: Ports and harbors	SQGs generally appropriate. Assessments typically address complex mixtures, high TOC, highly modified habitat, multi-jurisdictional issues, high resource demand, and involve multiple contaminant sources.	• Further verify SQGs with site-specific data, develop site-specific SQGs, if needed and if practical • Rely heavily on ground-truthing using WOE • Relate reference or background comparisons to assessment goals • Account for physical disturbances that confound benthic assessments
Depositional: Open ocean disposal	SQGs may not be appropriate. Highly modified sites, can attract biota, exposure difficult to quantify, bioavailability likely to change between dredging and placement.	• Evaluate exposure based on residence time in the affected area • Develop appropriate site-specific SQGs (coastally derived SQGs may not be appropriate) • Use WOE to assess effects
Depositional: Oil and gas production environments	SQGs may not be appropriate. Altered physical habitat, increased biological activity (hydrocarbon food source), modified benthos (nutrient, oxygen changes), potential ecotoxicity.	• Use before and after comparisons, temporal trends • Distinguish between petrogenic and pyrogenic PAH and hydrocarbons for source identification • Use a WOE approach

Table 4 continued

Habitats	Issues	Suggestions
Depositional: Highly modified systems	SQGs may not be appropriate. Altered physical habitat, multiple sources, and no obvious reference condition. Elevated TOC, legacy contamination; multiple contaminants make ecotoxicological significance of single contaminants unclear.	• Identify comparison sites based on physical similarities (no true reference sites) • Conduct initial benthos surveys • Conduct pattern analyses to find sources and develop mass loadings
Estuaries	May require site-specific SQGs and additional bioassays. Dynamic areas dominated by salinity, grain size and other gradients; contaminants, sediments and benthic populations move bi-directional.	• Define salinity and grain size zones both temporally and spatially • Use salinity appropriate bioassays and site-specific SQGs and/or normalized background comparisons • Design sampling based on salinity and grain size zones
Non-depositional and erosional systems	SQGs are not appropriate. Lack of fine grain sediments; longitudinal variation in particle size, TOC, benthos.	• Measure other sources of exposure (e.g., biofilm) • Quantify longitudinal variation • Use sampling designs that quantify longitudinal variation

sediment-bound contaminants and define acceptable concentrations for waterway-specific, ecosystem-specific, regional, and possibly national applications.

The path forward to achieve this goal involves additional research. As illustrated by the general concentration–response paradigm for sediment-bound contaminants (Figure 2), one of the reasons for the transition zone between TE and PE guidelines is an incomplete understanding of the factors controlling the bioavailability of contaminants in sediment and the biological and ecological factors that alter an organism's response to contaminant exposures. Factors that create the transition zone are several; some reflect intrinsic variability (measurement uncertainty, and chemical, biological, or ecological variability), some reflect errors in design or interpretation of experiments, and some reflect an incomplete scientific understanding of microenvironments, biogeochemistry, bioavailability, nonchemical stressors, and biological activity. The ability to interpret the importance of these and other factors is further limited by an incomplete understanding of mixtures and the potential of unmeasured contaminants.

A related area of research is needed to improve the understanding of the variation in site-specific bioavailability of both sediment-bound organic contaminants and metals. A better understanding of the factors controlling bioaccumulation of metals and the

importance of metal tissue residues is needed to confirm the relationships established by mechanistic SQG approaches such as EqP and SEM–AVS.

Consequently, perhaps the highest research priority in the field of sediment quality assessment is to further develop or refine the current approaches for estimating chemically based numeric SQGs from field effects of sediment-bound contaminants on benthic ecosystems. Certainly, the currently available sets of SQGs can be improved; but, any new or revised sets of SQGs must be demonstrated with empirical independent observations to be more predictive of effects and more protective of valued biological resources than are those currently available. The discrimination of adverse biological effects associated with exposure to single or mixtures of contaminants from those responses attributable to other noncontaminant stressors (e.g., grain size, organic enrichment, salinity, habitat modification, and exotic species) is essential in this research.

In addition, research is needed to support the development of new SQGs for characterizing exposure and effects from bioaccumulative contaminants. None of the currently available SQGs adequately address trophic transfer mechanisms and the potential for some contaminants to move within aquatic food webs. Progress in this area will provide an improvement in sediment assessments and management decisions given the increasingly ubiquitous occurrence of bioaccumulative contaminants in the environment.

Research is also needed to develop single contaminant SQGs based on spiked sediments, or evaluation of field samples where there are a limited number of contaminants. Future analyses of the predictive ability of SQGs should include an evaluation of the sensitivity and efficiency of SQGs (to date, sensitivity and efficiency analyses have been performed primarily for AETs). The number of sediment samples needed to conduct a site-specific evaluation of the utility of SQGs can be estimated from the variance associated with established concentration–response relationships for SQGs. Future evaluations of the predictive ability of SQGs should also include controlled benthic community colonization studies, mesocosm studies, and chronic laboratory tests to better account for abiotic and habitat factors influencing the response of benthic invertebrates in the field. Further research is also needed to determine the conditions where site-specific SQGs may differ from SQGs derived from a range of toxicity studies reported in the literature.

Finally, training in the role of SQGs as one tool among several to evaluate sediment quality is highly recommended. Short courses could be given that address the development and application of SQGs and other measures of sediment quality. Once developed, these short courses should then be presented at annual regional, national, and international scientific meetings. Aside from the various chemical and biological areas of research needed to further elucidate the significance of contaminants in sediment, there is an urgent need to provide training to professionals charged with sediment management. Among the many issues currently debated regarding the use of SQGs, the interpretation of contaminants in sediment using SQGs has not often been consistent with the narrative intent of the different SQG approaches.

Acknowledgments

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Multiple Stressors in Ecological Risk and Impact Assessment
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SETAC

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