Development and Applications of Sediment Quality Criteria for Managing Contaminated Sediment in British Columbia

Submitted to:

Mike Macfarlane
British Columbia Ministry of Water, Land and Air Protection
Environmental Management Branch
PO Box 9342 Stn Prov Govt
Victoria, British Columbia V8W 9M1

Submitted – November 2003 – by:

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ENVIRONMENTAL SCIENCES LTD.

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List of Acronyms

2,3,7,8-TCDD TEQ	2,3,7,8-tetrachlordibenzo-*p*-dioxin toxic equivalent
AETA	Apparent Effects Threshold Approach
ANOVA	analysis of variance
AVS	Acid volatile sulfide
BEDS	Biological Effects Database for Sediment
BIEAP	Burrard Inlet Environmental Action Program
BSAF	sediment-to-biota bioaccumulation factor
CA	Consensus Approach
CCME	Canadian Council of Ministers of the Environment
CEPA	*Canadian Environmental Protection Act*
COPC	chemical of potential concern
CSR	Contaminated Sites Regulation
EC_{50}	median effective concentration affecting 50 percent of the test organisms
ELA	Effects Level Approach
EqPA	Equilibrium Partitioning Approach
ERA	Effects Range Approach
ERL	effects range-low
ERM	effects range-median
ESG	equilibrium-based sediment guideline
FA	*Fisheries Act*
FCV	final chronic value
foc	fraction organic carbon
FREMP	Fraser River Estuary Management Program
K_{oc}	sediment organic carbon
K_{ow}	octanol-water partition coefficient
Kp	sediment/water partition coefficients
LEL	lowest effect level
LRMA	Logistic Regression Modelling Approach
MESL	MacDonald Environmental Sciences Ltd.
MET	minimal effect threshold
"Mean-MPP (or)"	a procedure for calculating mean SedQC-Qs that involves utilization of sediment chemistry for metals, PAHs and/or PCBs
NCSRP	National Contaminated Sites Remediation Program
NEC	no effect concentration
NSTP	National Status and Trends Program
OC pesticide	organochlorine pesticide
P_{20}	20% probability of observing toxicity
PAETs	probable AET
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin

PCDF	polychlorinated dibenzofuran
PEC	probable effect concentration
PEC-Q	probable effect concentration-quotient
PEL	probable effect level
PSQS	preliminary sediment quality standard
RAP	remedial action plan
SEC	sediment effect concentrations
SedQC	sediment quality criteria
SedQC-Q	sediment quality criteria-quotients
$SedQC_{SCS}$	sediment quality criteria for sensitive contaminated sites
$SedQC_{TCS}$	sediment quality criteria for typical contaminated sites
SedQGs	sediment quality guidelines
SedTox	sediment toxicity database
SEL	severe effect level
SEM	simultaneously extracted metal
SLC	screening level concentration
SLCA	Screening Level Concentration Approach
SMO	sediment management objective
SQG	sediment quality guideline
SQG-Q	sediment quality guideline-quotient
SQS	sediment quality standard
SSLC	species screening level concentration
SVOCs	semi-volatile organic chemicals
SWR	Special Waste Regulation
T_{10}	10% probability of observing toxicity
TECs	threshold effect concentrations
TEL	threshold effect level
TET	toxic effect threshold
TOC	total organic carbon
TRA	tissue residue approach
TRC	tissue residue criteria
TRG	tissue residue guideline
WMA	*Waste Management Act*

Chapter 1 Interests and Needs Related to the Assessment of Contaminated Sediments

1.0 Introduction

In British Columbia, the federal and provincial governments share authority for assessing and managing contaminated sediments (i.e., under the *Fisheries Act - FA*, the *Waste Management Act - WMA*, and, to a lesser extent, the *Canadian Environmental Protection Act - CEPA*). Currently, standard procedures for assessing contaminated sediments have not been established by either level of government. In addition, numerical sediment quality criteria (SedQC) have not been formally established for assessing or managing contaminated sites. As such, decisions regarding the selection of assessment procedures and the establishment of remedial targets are being made on a site-by-site basis. For this reason, the establishment of SedQC and the harmonization of the federal and provincial requirements of managing sites with contaminated sediment have been identified as priorities by both levels of government.

1.1 Role of Sediments in Freshwater Ecosystems

The particulate materials that lie below the water in ponds, lakes, springs, streams, rivers, and other aquatic systems are called sediments (ASTM 2003a). Sediments represent essential elements of aquatic ecosystems because they support both autotrophic and heterotrophic organisms. Autotrophic (which means self-nourishing) organisms are those that are able to synthesize food from simple inorganic substances (e.g., carbon dioxide, nitrogen, and phosphorus) and the sun's energy. Green plants, such as algae, bryophytes (e.g., mosses and liverworts), and aquatic macrophytes (e.g., sedges, reeds, and pond weed), are the main autotrophic organisms in freshwater ecosystems. In contrast, heterotrophic (which means other-nourishing) organisms utilize, transform, and decompose the materials that are synthesized by autotrophic organisms (i.e., by consuming or decomposing autotrophic and other heterotrophic organisms). Some of the important heterotrophic organisms that can be present in aquatic ecosystems include bacteria, epibenthic and infaunal invertebrates, fish, amphibians, and reptiles. Birds and mammals can also represent important heterotrophic components of aquatic and aquatic-dependent food webs (i.e., through the consumption of aquatic organisms).

Sediments support the production of food organisms in several ways. For example, hard-bottom sediments, which are characteristic of faster-flowing streams and are comprised largely of gravel, cobbles, and boulders, provide stable substrates to which periphyton (i.e., the algae that grows on rocks) can attach and grow. Soft sediments, which are common in ponds, lakes, and the slower-flowing sections of rivers and streams, are comprised largely of sand, silt, and clay. Such sediments provide substrates in which aquatic macrophytes can root and grow. The nutrients that are present in such sediments can also nourish aquatic macrophytes. By providing habitats and nutrients for aquatic plants, sediments support autotrophic production (i.e., the production of green plants) in aquatic systems. Sediments can also support prolific bacterial and meiobenthic communities, the latter including protozoans, nematodes, rotifers, benthic cladocerans, copepods, and other organisms.

Bacteria represent important elements of aquatic ecosystems because they decompose organic matter (e.g., the organisms that die and accumulate on the surface of the sediment, as well as anthropogenically-derived organic chemicals) and, in so doing, release nutrients to the water column and increase bacterial biomass. Bacteria represent the primary heterotrophic producers in aquatic ecosystems, upon which many meiobenthic organisms depend. The role that sediments play in supporting primary productivity (both autotrophic and heterotrophic) is essential because green plants and bacteria represent the foundation of food webs upon which all other aquatic organisms depend (i.e., they are consumed by many other aquatic species).

In addition to their role in supporting primary productivity, sediments also provide essential habitats for many sediment-dwelling invertebrates and benthic fish. Some of these invertebrate species live on the sediments (termed epibenthic species), while others live in the sediments (termed infaunal species). Both epibenthic and infaunal invertebrate species consume plants, bacteria, and other organisms that are associated with the sediments. Invertebrates represent important elements of aquatic ecosystems because they are consumed by a wide range of wildlife species, including amphibians, reptiles, fish, birds, and mammals. For example, virtually all fish species consume aquatic invertebrates during all or a portion of their life cycle. In addition, many birds (e.g., dippers, sand pipers, and swallows) consume aquatic invertebrates. Similarly, aquatic invertebrates represent important food sources for both amphibians (e.g., frogs and salamanders) and reptiles (e.g., turtles and snakes). Therefore, sediments are of critical importance to many wildlife species due to the role that they play in terms of the production of aquatic invertebrates.

Importantly, sediments can also provide habitats for many wildlife species during portions of their life cycle. For example, a variety of fish species utilize sediments for spawning and incubation of their eggs and larvae. In addition, juvenile fish often find refuge from predators in sediments and/or in the aquatic vegetation that is supported by the sediments. Furthermore, many amphibian species burrow into the sediments in the fall and remain there throughout the winter months, such that sediments provide important overwintering habitats. Therefore, sediments play a variety of essential roles in terms of maintaining the structure (i.e., assemblage of organisms in the system) and function (i.e., the processes that occur in the system) of aquatic ecosystems.

1.2 Sediment Quality Issues and Concerns

Considering the important roles that they play, it is apparent that sediments represent essential elements of freshwater ecosystems. Yet, the available information on sediment quality conditions indicate that sediments in many water bodies are contaminated by a wide range of toxic and bioaccumulative substances, including metals, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), organochlorine pesticides (OC pesticides), a variety of semi-volatile organic chemicals (SVOCs), and polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs; FRAP 1997; MESL 1997; USEPA 1997a). The nature and extent of such sediment contamination depend on a variety of factors, such as the types of contaminant sources that are present in the system under investigation, the loadings of contaminants from the various sources, proximity to sources, and the fate of the contaminants once they are released into the aquatic system.

Contaminated sediments represent an important environmental concern for several reasons. First, contaminated sediments have been demonstrated to be toxic to sediment-dwelling organisms and fish (Ingersoll *et al.* 1997). As such, exposure to contaminated sediments can result in decreased survival, reduced growth and/or impaired reproduction in benthic invertebrates and fish. Additionally, some contaminants in the sediments are taken up by benthic organisms through a process called bioaccumulation (Ingersoll *et al.* 1997). When larger animals feed on these contaminated prey species, the pollutants are taken into their bodies and are passed along to other animals in the food web in a process called biomagnification. As a result of the effects of toxic and bioaccumulative substances, benthic organisms, fish, birds, and mammals can be adversely affected by contaminated sediments (MacDonald *et al.* 2002).

Contaminated sediments can also adversely affect human health and the human uses of aquatic ecosystems. First, human health can be adversely affected due to direct exposure to contaminated sediments during wading or swimming in affected waterbodies. Consumption of contaminated fish and shellfish also poses a risk to human health. Human use of aquatic ecosystems can be compromised by the presence of contaminated sediments through reductions in the abundance of food or sportfish species or due to the imposition of fish consumption advisories (i.e., when fish or shellfish tissues are found to contain unacceptable levels of bioaccumulative substances). As such, contaminated sediments in freshwater ecosystems pose potential hazards to sediment-dwelling organisms (i.e., epibenthic and infaunal invertebrate species), aquatic-dependent wildlife species (i.e., fish, amphibians, reptiles, birds, and mammals), and human health.

1.3 Chemicals of Potential Concern

Identification of chemicals of potential concern (COPCs) represents an essential element of the overall SedQC derivation process. In the context of this report, COPCs are defined as those substances that are released into freshwater, estuarine, or marine ecosystems as a result of human activities (including those originating from both point and non-point sources) and have the potential to adversely affect the uses of aquatic ecosystems (e.g., aquatic life, recreation and aesthetics). It is important to identify the COPCs in British Columbia because such information, when considered in conjunction with data on the environmental fate and persistence of these chemicals, provides a basis for determining which substances are likely to partition into sediments (i.e., the sediment-associated COPCs). The toxic and bioaccumulative COPCs that are likely to occur in sediments within the province are considered to be the highest priority for establishing numerical SedQC.

In general, COPCs are identified using information on the land and water uses within the waterbody under consideration. More specifically, information on existing and historic land and water uses is utilized to identify the probable sources of environmental contaminants within the waterbody. In turn, data on the chemical characteristics of point and non-point source discharges from these sources, the results of historic and ongoing environmental monitoring programs, and information on the environmental fate and persistent of the substances that have been or are likely to have been released into surface waters can be used to identify the substances that are likely to partition into sediments (i.e., sediment-associated COPCs).

The results of several investigations conducted over the past decade suggest that numerous COPCs have been released into surface waters in British Columbia (FRAP 1997; MESL 1997; Golder Associates 1999). To expedite the identification of COPCs, the members of the Federal/Provincial Technical Steering Committee were asked to draw on their extensive experience and identify the substances that have the potential to accumulate in bedded sediments in the province. Based on the input that was provided by committee members, the following substances were identified as sediment-associated COPCs in British Columbia:

Toxic Substances that Partition into Sediments:

- Metals (arsenic, cadmium, chromium, copper, lead, mercury, and zinc);
- PAHs (acenaphthene, acenaphthylene, anthracene, fluorene, 2-methylnaphthalene, naphthalene, phenanthrene, benz(a)anthracene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene, fluoranthene, pyrene, total PAHs, and other PAHs);
- PCBs (total PCBs);
- Chlorinated phenols (pentachlorophenol); and,

- OC pesticides (chlordane; dieldrin, DDTs, endrin, heptachlor, heptachlor epoxide, and lindane).

Bioaccumulative Substances that Partition into Sediments:

- Metals (lead and mercury);
- PAHs (acenaphthene, acenaphthylene, anthracene, fluorene, 2-methylnaphthalene, naphthalene, phenanthrene, benz(a)anthracene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene, fluoranthene, pyrene, total PAHs, and other PAHs);
- PCBs;
- PCDDs and PCDFs; and,
- OC pesticides (chlordane; dieldrin, DDTs, endrin, heptachlor, heptachlor epoxide, and lindane).

The toxic substances that partition into sediments were identified as the highest priority for establishing effects-based SedQC, while the bioaccumulative substances that partition into sediments were targeted for establishing bioaccumulation-based SedQC and/or tissue residue criteria (TRC). Such TRC are presented in Macfarlane *et al.* (2003). The SedQC for bioaccumulative substances will be established at a later date.

1.4 Purpose of this Report

This document was prepared to describe the process that was used to establish and evaluate the effects-based SedQC for managing contaminated sediments in British Columbia. More specifically, this report describes the existing framework for managing contaminated sites in British Columbia (Chapter 2). In addition, a review of the various approaches that could be used for establishing numerical SedQC for freshwater, estuarine, and marine ecosystems is presented (Chapter 3). Furthermore, the procedures that were used to derive numerical

SedQC for assessing and managing sediment contaminated sites in British Columbia are described (Chapter 4). The report also presents the results of evaluations conducted to assess the reliability of the SedQCs (Chapter 5). Finally, the applications of the SedQC are discussed (Chapter 6).

Chapter 2 Existing Framework for Managing Contaminated Sites in British Columbia

2.0 Introduction

The procedures for assessing and managing contaminated sites that fall under provincial jurisdiction are specified in two components of the *WMA*, including the Contaminated Sites Regulation (CSR) and the Special Waste Regulation (SWR). The site management process proposed under the CSR is intended to provide a consistent basis for assessing and remediating contaminated sites in the province. The process consists of five main elements, including site identification and assessment; site investigation; planning; remediation; and, evaluation and monitoring. However, every site need not proceed through each component of the process (Figure 1 and 2). The following summary of the framework is intended to provide an overview of the existing contaminated site management process. More detailed information on the elements of this framework is included in the CSR (BC 1997) and in a series of Fact Sheets that have been published by the Ministry.

2.1 Site Identification and Assessment

In British Columbia, sediment contaminated sites are identified in much the same way that other contaminated sites are identified. That is, through the preparation, submission, and evaluation of a site profile. Site profiles must be submitted to the responsible government agency when an application for subdivision, zoning, development, demolition of a structure, or removal of soil is received by a local government or when ordered by a regional manager.

Following its submission, the site profile is assessed by provincial or local government official and a determination is made regarding the need for further investigations at the site. The completed site profile is intended to provide the responsible agency(ies) with sufficient information for determining if the site under consideration is a potential contaminated site.

Therefore, the information provided in the site profile should provide sufficient information to assess the potential for sediment contamination. For example, the site map which is provided with the site profile provides a basis for identifying waterfront properties, while the historic land use information should provide a basis for assessing the potential for releases of toxic and/or bioaccumulative substances into receiving water systems. Sediment contamination should be suspected at any waterfront site which is suspected to have soil, surface water, or groundwater contamination and at any site located nearby effluent or stormwater discharges. Further site investigations to assess sediment contamination must be undertaken at sites are suspected to have contaminated sediments. However, no further action is required at sites that considered to not be potentially contaminated.

2.2 Site Investigation

Information from the site profile or from other sources may indicate that a site may have contaminated sediments. In this situation, preliminary and/or detailed site investigations may be required to determine if the site is contaminated, as defined under the CSR. Preliminary site investigations are intended to determine the probability that a site is contaminated, based on archival records, site visits, and knowledge of the historical activities that were conducted on site. In addition, field sampling is commonly undertake at this stage of the investigation to evaluate the nature, location, and magnitude of sediment contamination. The available data on the concentrations of sediment-associated COPCs are usually compared to established sediment quality benchmarks (i.e., SedQC) to determine if it is a sediment contaminated site, as defined in the CSR (the reader is directed to Chapter 6 of this report for more information on the application of SedQC for identifying sediment contaminated sites).

At sites that are designated as contaminated sites, more comprehensive investigations need to be conducted to evaluate the magnitude, severity, and areal extent of sediment contamination at the site. In addition to generating additional sediment chemistry data, information is often collected on the physical and biological characteristics of the site. The detailed site investigations are intended to provide the information needed to confirm or refute the potential for site contamination. The legislation provides for the use of numerical and risk-based standards for determining if a site is contaminated, if remedial measures are required, and if they have been satisfactorily completed.

2.3 Site Management Planning

Following the completion of the detailed site investigation and the designation of a site as contaminated, a planning process is initiated to support the management of the site. The first priority in the planning stage of the site management process is to determine who is potentially responsible for the contamination and who is potentially liable for clean-up costs. In addition, the need for and relative priority for remediation is assessed at this stage of the process. Other important planning steps include evaluating various remediation options (i.e., the feasability study) and initiating the approvals process.

2.4 Site Remediation

With the completion of the feasability study and the selection of the preferred remedial alternative(s), remedial actions can be initiated at the site. The remediation step in the process covers all of the activities that are associated with cleaning-up or securing a contaminated site. The legislation defines two broad types of remediation, including removal of contaminants by excavation or treatment and management of contaminants on-site. The legislation also provides environmental quality standards that are used to determine when the cleanup is complete. Alternatively, risk-based procedures may be used to determine the level of contamination that can remain on-site. In such situations, additional protective measures may have to be taken to assure that the uses of the site are not impaired.

2.5 Monitoring and Evaluation

Following their implementation, confirmatory sampling and analysis are normally conducted to determine if remedial measures have reduced the level of contamination or risk to tolerable levels. If the numerical or risk-based standards of the Contaminated Sites Regulation have been satisfied, then a certificate can be issued by the Ministry. When the contamination is managed on-site, conditions must be met by the site manager to ensure protection of the environment and human health.

Chapter 3 Approaches for Establishing Numerical Sediment Quality Criteria for Freshwater, Estuarine, and Marine Ecosystems

3.0 Introduction

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

A number of jurisdictions throughout North America have developed numerical SQGs for freshwater, estuarine, and/or marine ecosystems. The SQGs that are currently being used in North America have been developed using a variety of approaches, including both empirical and theoretical approaches. Both empirical and theoretical approaches were considered to support the derivation of numerical SedQCs for assessing and managing sediment contaminated sites in BC, including:

- Screening Level Concentration Approach (SLCA);
- Effects Range Approach (ERA);
- Effects Level Approach (ELA);

- Apparent Effects Threshold Approach (AETA);
- Equilibrium Partitioning Approach (EqPA);
- Logistic Regression Modelling Approach (LRMA); and,
- Consensus Approach (CA).

The tissue residue approach (TRA) was considered to be the primary method for deriving numerical sediment quality objectives for the protection of wildlife and human health (i.e., for substances that bioaccumulate in the food web).

This chapter of the report is intended to provide the information needed to support the selection of the most relevant approach or approaches for establishing numerical SedQC for managing contaminated sediment in British Columbia. To that end, the existing approaches to the derivation of numerical SQGs and their uses are described. Additionally, each of these approaches are critically evaluated to determine their strengths and limitations (Table 1).

3.1 Screening Level Concentration Approach

The SLCA is a biological effects-based approach for deriving SQGs for the protection of benthic organisms. This approach utilizes matching biological and chemical data collected in field surveys to calculate a screening level concentration (SLC; Neff *et al*. 1986). The SLC is an estimate of the highest concentration of a COPC that can be tolerated by a pre-defined proportion of benthic infaunal species.

The SLC is calculated using a database that contains information on the concentrations of specific COPCs in sediments and on the co-occurrence of benthic organisms in the same sediments. For each benthic organism for which adequate data are available, a species screening level concentration (SSLC) is calculated. The SSLC is determined by plotting the frequency distribution of the COPC concentrations over all of the sites at which the species occurs (information from at least ten sites is required to calculate a SSLC). The 90th percentile of this distribution is taken as the SSLC for the species being investigated. The SSLCs for all of the species for which adequate data are available are then compiled as a frequency distribution to determine the concentration that can be tolerated by a specific proportion of the species (i.e., the 5th percentile of the distribution would provide an SLC that should be tolerated by 95% of the species). This concentration is termed the screening level concentration of the COPC.

A number of jurisdictions have used the SLCA to derive numerical SQGs. For example, Neff *et al.* (1986) developed freshwater SLCs for a variety of chemical substances, primarily using data from the Great Lakes. Similarly, the Quebec Ministry of the Environment used the SLCA to derive two SQGs for each COPC in the St. Lawrence River, including a minimal effect threshold (MET) and a toxic effect threshold (TET; EC and MENVIQ 1992). The MET was calculated as the 15th percentile of the SSLCs, while the TET was calculated as the 90th percentile of the SSLC distribution for each substance. Therefore, the MET and TET are considered to provide protection for 85% and 10% of the species represented in the database, respectively. Furthermore, Environment Ontario developed a lowest effect level (LEL) and severe effect level (SEL) for various chemical substances using this approach (Persaud *et al.* 1993).

3.2 Effects Range Approach

The ERA to the derivation of SQGs was formulated to provide informal tools for assessing the potential for various COPCs tested in the National Status and Trends Program (NSTP) to be associated with adverse effects on sediment-dwelling organisms (Long and Morgan 1991). The SQGs derivation process involves several steps, including acquisition of candidate data sets, review and evaluation of data sets, compilation of acceptable data in a project database, and data analysis (including guideline derivation).

In the first step of the process, candidate data sets were identified using bibliographic database searches and communications with investigators active in the sediment assessment field. Following their retrieval, candidate data sets were reviewed and evaluated to determine their applicability for incorporation into the database (MacDonald *et al.* 1996). This evaluation was designed to determine the overall applicability of the data set, the methods that were used, the endpoints that were measured, and the degree of concordance between the chemical and biological data. The data which met the evaluation criteria were incorporated into the project database.

Information from several types of investigations were incorporated into the project database, including spiked-sediment toxicity tests, field studies conducted in North America, and initiatives directed at the formulation of numerical SQGs. All of the information contained in the database was weighted equally, regardless of the methods that were used in the investigation. Individual entries in the database consisted of the concentration of the COPC, the location of the study, the species tested and endpoint measured, and an indication of whether or not there was concordance between the observed effect and the concentrations of a specific chemical (i.e., no effect, no or small gradient, no concordance, or a "hit", which indicated that an effect was measured in association with elevated sediment chemistry). Data from non-toxic or unaffected samples were assumed to represent background conditions. Data which showed no concordance between chemical and biological variables were included in the database, but were not used to calculate the SQGs.

Simple analytical procedures were used to derive numerical SQGs using the information that was compiled in the database. First, the data for which a biological effect was observed in association with elevated chemical concentrations (i.e., hits) were sorted in ascending order of concentration. Next, the 10th and 50th percentile concentrations for each compound were determined. The effects range-low (ERL; 10th percentile value) was considered to represent a lower threshold value, below which adverse effects on sensitive life stages and/or species occurred only infrequently. The effects range-median (ERM; 50th percentile value) was considered to represent a second threshold value, above which adverse effects were frequently observed.

Using the ERA, Long and Morgan (1991) and Long *et al.* (1995a) derived two types of informal SQGs (i.e., ERL and ERM) for use in the NSTP. The database that was used by Long and Morgan (1991) to derive the SQGs consisted of data from freshwater, estuarine, and marine ecosystems. Ingersoll *et al.* (1996) used a similar approach to derive ERLs (15th percentile of the effects data set) and ERMs (50th percentile of the effects data set) for assessing sediments from various freshwater locations in the United States. Similarly, MacDonald (1997) applied the ERA to regionally-collected field data to derive site-specific sediment effect concentrations (SECs) for PCBs and DDTs in the Southern California Bight.

3.3 Effects Level Approach

The ELA is closely related to the ERA described above. However, the ELA is supported by an expanded version of the database that was used to derive the effects levels (Long and Morgan 1991). The expanded database contains matching sediment chemistry and biological effects data from spiked-sediment toxicity tests and from field studies conducted throughout North America (including both effects and no effects data). The expanded database also contains sediment quality guidelines derived using various approaches. The information contained in the expanded database was evaluated and classified in the same manner that was used to compile the original NSTP database.

In the ELA, the underlying information in the database was used to derive two types of sediment quality guidelines, including threshold effect levels (TELs) and probable effect levels (PELs). The TEL, which is calculated as the geometric mean of the 15^{th} percentile of the effects data set and the 50^{th} percentile of the no effects data set, represents the chemical concentration below which adverse effects are expected to occur only infrequently. The PEL represents a second threshold value, above which adverse effects are expected to be frequently observed. The PEL is calculated as the geometric mean of the 50^{th} percentile of the effects data set and the 85^{th} percentile of the no effects data set.

The ELA was applied to the expanded database (i.e., Biological Effects Database for Sediments; BEDS) to derive numerical SQGs (i.e., TELs and PELs) for Florida coastal waters (MacDonald *et al.* 1996). Similarly, Ingersoll *et al.* (1996) applied this approach to the results of freshwater toxicity tests on amphipods and midges to derive SQGs for assessing sediment quality conditions in freshwater systems. Furthermore, Smith *et al.* (1996) and CCME (1999) used the ELA to derive TELs and PELs for freshwater and marine systems in Canada.

3.4 Apparent Effects Threshold Approach

The AETA to the development of SQGs was developed for use in the Puget Sound area of Washington State (Tetra Tech Inc. 1986). The AETA is based on empirically-defined relationships between measured concentrations of a COPC in sediments and observed biological effects. This approach is intended to define the concentration of a COPC in sediment above which significant ($p \leq 0.05$) biological effects are *always* observed. These biological effects include, but are not limited to, toxicity to benthic and/or water column species (as measured using sediment toxicity tests), changes in the abundance of various benthic species, and changes in benthic community structure. The AET values can be based on dry weight-normalized COPC concentrations or total organic carbon-normalized concentrations for organic substances (Barrick *et al*. 1988; WDOE 1990).

The state of Washington has used AET values to establish sediment quality standards (SQSs) and minimum clean-up levels for contaminants of concern in the state (WDOE 1990). Cubbage *et al.* (1997) refined this approach to support the development of probable AETs (PAETs) using matching sediment chemistry and toxicity data for freshwater sediments from the state of Washington. Ingersoll *et al.* (1996) utilized a similar approach to develop freshwater AETs (termed no effect concentrations or NECs in that study) using the results of toxicity tests and chemical analyses conducted on sediments from various freshwater locations in the United States.

3.5 Equilibrium Partitioning Approach

The water-sediment EqPA is based on the premise that the distribution of COPCs among the two principal compartments in the sediment matrix (i.e., sediment solids and interstitial water) is predictable based on their physical and chemical properties, assuming that continuous equilibrium exchange between sediment and interstitial water occurs. This approach has been supported by the results of spiked-sediment toxicity tests, which indicate that positive correlations exist between the biological effects observed and the concentrations of COPCs measured in the interstitial water (Di Toro *et al.* 1991; Berry *et al.* 1996; Hansen *et al.* 1996).

In the EqPA, water quality criteria developed for the protection of freshwater or marine organisms are used to support the SQGs derivation process. As such, the water quality criteria formulated for the protection of water column species are assumed to be applicable to benthic organisms (Di Toro *et al*. 1991). The SQGs are calculated using the appropriate water quality criteria [usually the final chronic values (FCVs) or equivalent values; USEPA 1998] in conjunction with the sediment/water partition coefficients (Kp) for the specific COPCs. The FCV is derived from the species mean chronic values that have been calculated from published toxicity data and is intended to protect 95% of aquatic species. The calculation procedure for non-ionic organic COPCs is as follows:

$$\mathbf{SQG = Kp \times FCV}$$

where:

SQG = Sediment quality guideline (in µg/kg);
Kp = Partition coefficient for the chemical (in L/kg); and,
FCV = Final chronic value (in µg/L).

The Kp is a function of the partition coefficient for sediment organic carbon (K_{oc}) of the substance under consideration and the amount of organic carbon in the sediment under investigation (f_{oc}; where Kp = K_{oc} x f_{oc}; Di Toro *et al.* 1991). The K_{oc} for non-ionic substances can be calculated from its octanol-water partition coefficient (K_{ow}; Di Toro *et al.* 1991). The f_{oc} is the decimal equivalent of the percent organic carbon in the sediment (i.e., f_{oc} = 0.01 if total organic carbon = 1%).

The EqPA has been used to derive numerical SQGs in several jurisdictions. For example, USEPA (1997a) reported organic carbon-normalized SQGs (termed equilibrium-based sediment guidelines; ESGs) for a variety of non-polar organic substances. In addition, draft ESGs have been developed for endrin, dieldrin, and metal mixtures (S. Ireland, United States Environmental Protection Agency, personal communication). The SQGs for divalent cationic metals (i.e., simultaneously extracted metals; SEM) are applied using data on the levels of acid volatile sulfide (AVS) in sediments (i.e., metals are thought to contribute to sediment toxicity only when SEM concentrations exceed AVS concentrations by a factor of five or more; Hansen *et al.* 1996; USEPA 1997a). New York State Department of Environmental Conservation also developed SQGs for the protection of aquatic life using the EqPA (NYSDEC 1999).

3.6 Logistic Regression Modelling Approach

In the LRMA, numerical SQGs are derived from the results of field studies conducted to assess sediment quality conditions. The sediment samples collected in such investigations typically contained complex mixtures of contaminants. The first step of the SQGs derivation process involves the collection, evaluation, and compilation of matching sediment chemistry and toxicity data from a wide variety of sites in North America. Next, the information compiled in the project database is retrieved on a substance-by-substance basis, with the data from individual sediment samples sorted in order of ascending concentration. For each sediment sample, the ascending data table provides information on the concentration of the COPC under consideration (on either a dry weight- or organic carbon-normalized basis) and the results of the toxicity test (i.e., toxic or not toxic) for each endpoint (e.g., 10-d survival of amphipods; Field *et al.* 1999).

In the next step of the process, the data contained in the ascending data tables are screened to minimize the potential for including samples in which the selected COPC did not contribute substantially to the observed toxicity. In this analysis, the chemical concentration in each toxic sample is compared to the mean concentration in the non-toxic samples from the same study and geographic area. The toxic samples with concentrations of the selected COPC that are less than or equal to the average concentration of that chemical in the non-toxic samples are not used in further analyses of the data (i.e., it was highly unlikely that the contaminant substantially contributed to sediment toxicity in such samples; Field *et al.* 2002).

In the final step of the analysis, the screened data are used to develop logistic regression models, which express the relationship between the concentration of the selected COPC and the probability of observing toxicity. In its simplest form, logistic models can be described using the following equation (Field *et al.* 1999):

$$\boldsymbol{p} = (e^{B0 + B1(x)}) \;/\; (1 + e^{B0 + B1(x)})$$

where:

p	=	probability of observing a toxic effect;
B0	=	intercept parameter;
B1	=	slope parameter; and,
x	=	concentration or log concentration of the chemical.

Using a preliminary database consisting of the results of 10-d toxicity tests with marine and estuarine amphipods, Field *et al*. (1999) derived logistic regression models for seven chemical substances to illustrate the methodology. More specifically, these investigators calculated T_{10}, T_{50}, and T_{90} values for four metals, two PAHs, and total PCBs. These values represent the chemical concentrations that correspond to a 10%, 50%, and 90% probability of observing sediment toxicity. In addition to supporting the derivation of specific T-values, this method can be used to determine the concentration of a COPC that corresponds to any probability of observing toxicity. Therefore, a sediment manager can identify an acceptable probability of observing sediment toxicity at a site (e.g., 25%) and determine the corresponding chemical concentrations (e.g., T_{25} value). The calculated value can then be used as the sediment quality guidelines (SQG) for the site. While the existing data from 10-d toxicity tests with marine and estuarine amphipods (endpoint: survival) support the development of logistic models for 37 substances (Field *et al.* 2002), insufficient data were available to derive reliable logistic models for any freshwater invertebrate species or toxicity test endpoint (Crane *et al.* 2000).

3.7 Consensus Approach

In the CA, consensus-based SQGs are derived from the existing SQGs that have been established for the protection of sediment-dwelling organisms. Derivation of numerical SQGs using the CA involves a four-step process. In a first step, the SQGs that have been derived by various investigators for assessing the quality of freshwater sediments are collected and collated. Next, the SQGs obtained from all sources are evaluated to determine their applicability to the derivation of consensus-based SQGs. The selection criteria that are applied are intended to evaluate the transparency of the derivation methods, the degree to which the SQGs are effects-based, and the uniqueness of the SQGs.

The effects-based SQGs that meet these selection criteria are then grouped to facilitate the derivation of consensus-based SECs (Swartz 1999). Specifically, the SQGs for the protection of sediment-dwelling organisms are grouped into two categories according to their original narrative intent, including threshold effect concentrations (TECs) and probable effect concentrations (PECs). The TECs are intended to identify COPC concentrations below which harmful effects on sediment-dwelling organisms are unlikely to be observed. Examples of TECs include TELs (Smith *et al.* 1996; Ingersoll *et al.* 1996), effect range low values (ERLs; Long and Morgan 1991; Ingersoll *et al.* 1996), and LELs (Persaud *et al.* 1993). The PECs are intended to identify COPC concentrations above which harmful effects on sediment-dwelling organisms are likely to be frequently or always observed (MacDonald *et al.* 1996; Swartz 1999). Examples of PECs include PELs (Smith *et al.* 1996; Ingersoll *et al.* 1996), effect range median values (ERMs; Long and Morgan 1991; Ingersoll *et al.* 1996); and SELs (Persaud *et al.* 1993).

Following classification of the existing SQGs, consensus-based TECs are calculated by determining the geometric mean of the SQGs that are included in this category. Likewise, consensus-based PECs are calculated by determining the geometric mean of the PEC-type values. The geometric mean, rather than the arithmetic mean, is calculated because it provides an estimate of central tendency that is not unduly affected by outliers and because the SQGs may not be normally distributed. Consensus-based TECs or PECs are calculated only if three or more published SQGs are available for a chemical substance or group of substances (MacDonald *et al.* 2000a).

The CA has been used to derive numerical SQGs for a variety of chemical substances and media types. For example, Swartz (1999) derived consensus-based SQGs for PAHs in marine ecosystems. Using a similar approach, MacDonald *et al.* (2000b) derived SQGs for total PCBs in freshwater and in marine and estuarine sediments. Ingersoll and MacDonald (1999) and MacDonald *et al.* (2000a) developed consensus-based SQGs for metals, PAHs, PCBs, and several pesticides in freshwater sediments. As the term implies, consensus-based SECs are intended to reflect the agreement among the various SQGs by providing an estimate of their central tendency. Consensus-based SECs are, therefore, considered to provide a unifying synthesis of the existing SQGs, reflect causal rather than correlative effects, and account for the effects of contaminant mixtures in sediment (Swartz 1999; MacDonald *et al.* 2000a; MacDonald *et al.* 2000b). The predictive ability of the consensus-based SECs were evaluated by MacDonald *et al.* (2000a; 2000b); Kemble *et al.* (2000), USEPA (2000), and Ingersoll *et al.* (2001; 2002; Crane *et al.* 2002).

3.8 Tissue Residue Approach

The TRA (which is also known as the biota-water-sediment EqPA) for deriving numerical SQGs was developed to address concerns regarding the bioaccumulation of sediment-associated COPCs in aquatic and aquatic-dependent food webs. The TRA is used to estimate the levels of individual chemicals or classes of chemicals in sediments that are unlikely to result in unacceptable tissue residues (i.e., levels in excess of the concentrations recommended to protect aquatic-dependent wildlife and/or human health).

Derivation of numerical SQGs using the TRA involves several steps. As a first step, the COPCs for which SQGs are to be derived are selected based on their potential to accumulate in aquatic food webs (e.g., based on their K_{ow}). Next, numerical tissue residue guidelines (TRGs) are identified for these COPCs. While most of the available TRGs are intended to provide protection for human health (e.g., Food and Drug Administration Action Levels; USEPA 1989), it is also important to obtain TRGs that are explicitly designed to protect piscivorus wildlife species. Following the selection of TRGs, sediment-to-biota bioaccumulation factors (BSAFs) are determined for each COPC. Such BSAFs can be determined from the results of bioaccumulation assessments, from matching sediment chemistry and tissue residue data (i.e., from the results of field studies), or from the results of bioaccumulation models. Numerical SQGs are subsequently derived using the equation:

$$\mathbf{SQG = TRG \;/\; BSAF}$$

The applicability of the TRA is supported by data which demonstrate that declines in DDT residues in fish and birds (since its use was banned) are strongly correlated with declining concentrations of this substance in surficial sediments in the Great Lakes and Southern California Bight. This approach has been used in Lake Ontario to derive numerical SQGs for 2,3,7,8 tetrachlorodibenzo-*p*-dioxin on the basis of fish tissue residues (Endicott *et al*. 1989; Cook *et al*. 1989). In addition, the New York State Department of Environmental Conservation has developed numerical SQGs for the protection of wildlife and human health using this approach (NYSDEC 1999). Health-based sediment quality guidelines have also been established in Washington State by the Washington State Department of Health (WDOH 1995; 1996).

Chapter 4 Derivation of Numerical Sediment Quality Criteria for Managing Sediment Contaminated Sites in British Columbia

4.0 Introduction

Under the former National Contaminated Sites Remediation Program (NCSRP), the Canadian Council of Ministers of the Environment (CCME) issued interim Canadian environmental quality criteria for soil and groundwater at contaminated sites. These criteria were adopted for use in managing contaminated sites in British Columbia in 1988 (BC 1988). In keeping with the policy of using current and comprehensive criteria to manage contaminated sites, the 1988 criteria were updated in 1995 and many were incorporated into the British Columbia CSR in 1997 (BC 1997).

The CSR provides detailed guidance on a range of issues related to the assessment and management of contaminated sites, including numerical standards for soil and water. While no specific guidance on the management of contaminated sediments was established in the CSR, the need for such guidance is indicated in the *WMA*. Specifically, Section 26(1) of the *Act* states that:

> *"A contaminated site means an area of land in which the soil or any groundwater lying beneath it, or the water or the underlying sediment, contains: a hazardous waste; or, another prescribed substance in quantities or concentrations exceeding prescribed risk-based or numerical criteria, or standards, or conditions."*

Therefore, criteria are required for assessing and remediating freshwater, estuarine, and marine sediments to support the management of contaminated sites in British Columbia. The Federal/Provincial Technical Steering Committee initiated the process of developing such SedQC by reviewing the existing approaches to the derivation of numerical SQGs (Chapter 2). The results of this review indicated that several approaches could, potentially, be used to support the development of SedQC for assessing and managing contaminated sediment sites in British Columbia. However, no single approach was likely to meet all of the needs for numerical SedQC.

This chapter of the report describes the steps that were taken by the Federal/Provincial Technical Steering Committee to develop SedQC for assessing and managing contaminated sediment sites in British Columbia. More specifically, the narrative intent of the SedQC is described (i.e., the sediment management objectives for sediment contaminated sites). In addition, the procedures that were used to identify preliminary benchmarks that generally met the narrative intent of the SedQC are presented. Furthermore, the methods that were used to generate concentration-response relationships for COPC mixtures in freshwater, estuarine, and marine sediments are described. Finally, the steps that were taken to refine the preliminary benchmarks for individual substances, based on the concentration-response relationships that were established for COPC mixtures, are presented.

4.1 Establishment of Sediment Management Objectives for Sediment Contaminated Sites

Description of the narrative intent of the SedQC is the first and most important step of the SedQC development process. In recognition that the uses of benthic habitats in freshwater, estuarine, and marine ecosystems differ among sites, the Federal/Provincial Technical Steering Committee established a two-tiered system for assessing and managing contaminated sediment sites in the province. More specifically, sediment contaminated sites were classified into two categories, sensitive sites and typical sites, based on the level of protection that is needed to support the designated uses of the aquatic ecosystem.

A number of criteria have been established to support the classification of contaminated sites, based on their designated uses (Table 2). Sensitive contaminated sites are those that are known to support red or blue listed plant and animal species, or nests designated under the *Wildlife Act*. In addition, such sites may be identified based on the use of aquatic habitats by threatened or endangered species or by species of special concern, as designated under the *Species at Risk Act*. Furthermore, sites with habitats that are important for the preservation of fish and wildlife, sites that encompass or border habitat compensation or restoration sites, and unique habitats or environmentally-sensitive areas as identified on provincial or municipal land use maps are considered to be sensitive sites under the Contaminated Sites Program. Finally, reaches of the aquatic environment that exist within provincial marine parks, provincial parks, ecological reserves or provincial wildlife management areas are considered to be sensitive for the purpose of assessing sediment quality conditions. All other sites are considered to be typical contaminated sites.

The Federal/Provincial Technical Steering Committee has established sediment management objectives (SMOs) for sensitive contaminated sites. These SMOs articulate the narrative intent of the SedQCs that are to be established for this type of site. At sites with sensitive habitats, the principal SMOs are to restore sediments to a state that will facilitate restoration of productive and diverse benthic macroinvertebrate communities in the ***near-term*** and to minimize the risks to organisms at higher trophic levels in the food web. For this reason, the effects-based criteria for sensitive sites ($SedQC_{SCS}$) need to be established at levels that provide a relatively high level of protection for sediment-dwelling organisms. That is, the $SedQC_{SCS}$ need to define concentrations of COPCs below which there is a relatively low probability of observing significant adverse effects in standardized toxicity tests with sensitive benthic species and life stages (i.e., 20% probability of observing an EC_{20}).

Sediment management objectives have also been established for typical contaminated sites to guide the development of numerical SedQC. At typical contaminated sites, the principal SMOs are to restore sediments to a state that will facilitate restoration of productive and diverse benthic macroinvertebrate communities in the ***longer-term*** and to minimize the risks to organisms at higher trophic levels in the food web. For this reason, the numerical criteria for typical sites ($SedQC_{TCS}$) need to be established at levels that provide a moderate level of protection for sediment-dwelling organisms. That is, the $SedQC_{TCS}$ are intended to define the concentrations of COPCs above which there is a moderate probability of observing significant adverse effects in standardized toxicity tests with sensitive benthic species and life stages (i.e., 50% probability of observing an EC_{20}).

4.2 Identification of Preliminary Benchmarks for Sediment Chemistry

Of the approaches to the derivation of SQGs that were reviewed, the LRMA provides the most direct means of identifying SedQC that meet the narrative intent expressed in the SMOs. However, logistic regression models have been established based on the results of 10-d toxicity tests with marine and estuarine amphipods only (Field *et al.* 1999; 2002). Therefore, it would be possible to derive SedQC using the LRM approach for marine and estuarine ecosystems only. For this reason, the Federal/Provincial Technical Steering Committee adopted an alternate approach to the establishment of numerical SedQCs.

The approach that was used to establish numerical SedQCs for assessing and managing contaminated sediment sites in British Columbia consisted of three main steps. Initially, preliminary benchmarks for sediment chemistry were identified. Then, the preliminary benchmarks were used together with matching sediment chemistry and toxicity data to develop concentration-response relationships for COPC mixtures. Then, the concentration-response models were used to identify the mean SedQC-quotients (SedQC-Qs) that corresponded to a 20% (i.e., for sensitive sites) or 50% (i.e., for typical sites) probability of observing significant toxicity (e.g., EC_{20}) to marine and estuarine or freshwater amphipods. These results were subsequently used to refine the preliminary benchmarks such that they were more consistent with the narrative intent expressed in the SMOs.

Because they have been developed for use throughout Canada and because they have been extensively evaluated, the Canadian sediment quality guidelines were used to establish the preliminary benchmarks for sediment chemistry (CCME 1999). The Canadian sediment quality guidelines report numerical values for each substance, including a TEL and a PEL (CCME 1999). The TEL is intended to define the concentration of a substance below which adverse biological effects are unlikely to occur. By comparison, the PEL is intended to define the concentration of a substance above which adverse biological effects are likely to occur frequently. The narrative intent of the PELs was considered to be generally consistent with the SMOs for sensitive and typical sites and, hence, were adopted directly as the preliminary benchmarks (Table 3).

4.3 Development of Concentration-Response Relationships for COPC Mixtures in Freshwater and in Marine and Estuarine Sediments

Development of concentration-response relationships for COPC mixtures provides a means of refining the preliminary benchmarks such that they are more consistent with the SMOs that have been established for sensitive and typical contaminated sites. Development of such concentration-response relationships involved the following steps:

- Acquiring matching chemistry and toxicity data for freshwater, estuarine, and marine sediments;
- Evaluating the data relative to project data quality objectives (DQOs);
- Compiling the data of acceptable quality on a per sample basis in a sediment toxicity database;
- Calculating the concentrations of key COPCs;
- Verifying and auditing the information in the SedTox database to assure data quality (i.e., following data entry and/or data translation); and,
- Development of concentration-response relationships for COPC mixtures.

Each of these steps is described in the following sections of this report.

4.3.1 Acquisition of Matching Sediment Chemistry and Toxicity Data

An extensive search of the scientific literature was conducted to acquire matching sediment chemistry and sediment toxicity data for developing concentration-response relationships for COPC mixtures in freshwater and in marine or estuarine sediments. More specifically, an effort was made to acquire all of the relevant information on the concentrations of COPCs in sediments and associated data on the effects of those sediments to sediment-dwelling organisms. The process that was used to identify and acquire candidate data sets included:

- Accessing the information contained in MacDonald Environmental Sciences Ltd.'s (MESL's) database on the effects of sediment-sorbed contaminants on aquatic organisms (i.e., BEDS);
- Conducting on-line searches of a number of bibliographic databases (e.g., Biosis, Aquaref, ChemAbstracts) to obtain recently published articles from peer-reviewed journals;
- Reviewing recent volumes of peer-reviewed journals that routinely publish papers on the effects of sediment-associated contaminants to access recently published data (e.g., *Chemosphere*, *Environmental Toxicology and Chemistry*; *Water, Air, and Soil Pollution*; *Toxicology*; *Archives of Environmental Contamination and Toxicology*; *Environmental Science and Technology*; *Ecotoxicology*); and,
- Contacting various experts in the sediment quality assessment field, by either letter or phone, to obtain published and unpublished data sets relevant to this project.

Hard copies of any candidate data sets identified using these procedures were retrieved for subsequent review and evaluation, and incorporated into the MESL library.

4.3.2 Review and Evaluation of Candidate Data Sets

All of the data sets and associated documents that were retrieved during the course of this study were critically evaluated to determine their scientific and technical validity. To support this evaluation, a set of selection criteria were developed in cooperation with the Science Advisory Group on Sediment Quality Assessment (Appendix 1). These selection criteria provided a means of consistently evaluating the methods that were used in each study, including the procedures that were used to collect, handle, and transport sediment samples, the protocols that were applied to conduct sediment toxicity tests, the methods that were used to determine the concentrations of COPCs in sediments, and the statistical tests that were applied to the study results. In many cases, additional communications with investigators and professional judgement were needed to determine if the selection criteria had been satisfied.

4.3.3 Development of a Sediment Toxicity Database

All of the matching sediment chemistry and toxicity data that met the selection criteria were incorporated into the project database on a per sample basis. Each record in the resulting database included the citation, a brief description of the study area (i.e., by waterbody and reach), a description of the sampling locations (including georeferencing data, if available), information on the toxicity tests that were conducted (including species tested, endpoint measured, test duration, etc.), type of material tested (i.e., whole sediment, pore water, or elutriate), total organic carbon (TOC) concentrations (if reported), and the chemical concentrations (expressed on a dry weight basis). Other supporting data, such as simultaneously extracted metals (SEM) concentrations, acid volatile sulfide (AVS) concentrations, and particle size distributions, were also included as available.

The freshwater data sets that were assembled provided information on the toxicity of whole sediment samples to a variety of sediment-dwelling organisms. More specifically, the freshwater database includes information on the effects associated with exposure to contaminated sediments on the following species: the amphipod, *Hyalella azteca* (endpoints: survival, growth, and maturation); the midges, *Chironomus tentans* or *Chironomus riparius* (endpoints: survival, growth, and emergence); the cladocerans (i.e., water fleas), *Daphnia magna*, *Daphnia pulex*, or *Ceriodaphnia dubia* (endpoints: survival and reproduction); the mayfly, *Hexagenia limbata* (endpoint: survival); steelhead trout, *Oncorhynchus mykiss* (endpoints: survival and growth); and the bacterium, *Vibrio fisheri* (Microtox; endpoint: bioluminescence). Additionally, the results of pore-water toxicity tests on the following species were incorporated into the regional database: the amphipod, *Hyalella azteca* (endpoint: survival); the cladoceran, *Daphnia magna* (endpoint: survival); steelhead trout, *Oncorhynchus mykiss* (endpoint: survival); and the bacterium, *Vibrio fisheri* (endpoint: bioluminescence).

Information of the toxicity of sediment-associated COPCs was also assembled for a variety of marine and estuarine species. More specifically, the results of whole sediment toxicity tests were compiled for the following marine and estuarine species: amphipods (*Rhepoxynius abronius, Ampelisca abdita, A. verrilli, Eohaustorius estuarius, Corophium acherusicum, C. volutator, Lepidactylus dytiscus, Grandidierella japonica*, and *Leptocheirus plumulosus* - endpoint: survival); bivalves (*Macoma nasuta, Mercenaria mercenaria*, and

Panope generosa - endpoint: survival and/or reburial), echinoderms (*Lytechinus pictus* - endpoint: survival); crustaceans (*Mysidopsis bahia* and *Palaemontes pugio* - endpoint: survival); and, polychaetes *(Armandia brevis, Neanthes arenaceodentata, N. spp., Nebalia pugettensis*, and, *Nereis virens* - endpoint: survival and growth). Additionally, the results of pore-water toxicity tests on the following species have also been incorporated into the database: microorganisms (*Vibrio fisheri* - endpoint: bioluminescence); bivalves (*Mytilus edulis* - endpoint: survival and development); echinoderms (*Arbacia punctulata, Dendraster ecentricus*, and *Strongylocentrotus purpuratus* - endpoint: survival and development); and, gastropods (*Haliotis rufescens* - endpoint: survival and development).

4.3.4 Calculation of the Total Concentrations of Key COPCs

To support subsequent interpretation of the sediment chemistry data, the total concentrations of several chemical classes were determined for each sediment sample. Specifically, the concentrations of total PAHs were calculated by summing the concentrations of up to 13 individual PAHs, including acenaphthene, acenaphthylene, anthracene, fluorene, 2-methylnaphthalene, naphthalene, phenanthrene, benz(a)anthracene, dibenz(a,h)anthracene, benzo(a)pyrene, chrysene, fluoranthene, and pyrene. For PCBs, the concentrations of total PCBs were determined using various procedures, depending on how the data were reported in the original study. If only the concentrations of total PCBs were reported in the study, then those values were used directly. If the concentrations of various Aroclors (e.g., Aroclor1242, Aroclor 1248) were reported, then the concentrations of the various Aroclors were summed to determine the concentration of total PCBs. If the concentrations of individual congeners were reported, these values were summed to determine total PCB concentrations. For DDTs, the concentrations of p,p’-DDD and o,p’-DDD, p,p’-DDE and o,p’-DDE, and p,p’-DDT and o,p’-DDT were summed to calculate the concentrations of sum DDD, sum DDE, and sum DDT, respectively. Total DDTs was calculated by summing the concentrations of sum DDD, sum DDE, and, sum DDT. Finally, the concentrations of chlordane were determined by summing the concentrations of alpha- and gamma-chlordane isomers. If only the concentrations of total chlordane were reported in the study, then those values were used directly.

In calculating the total concentrations of the various chemical classes, less than detection limit values for individual substances were assigned a value of one-half of the detection limit, except when the detection limit was greater than the consensus-based PEC (or an alternate sediment quality guideline if a PEC was not available; MacDonald *et al.* 2000a). In this latter case, the less than detection limit value was not used in the calculation of the total concentration of the substance.

4.3.5 Verification and Auditing of the Sediment Toxicity Database

A number of procedures were implemented to assure the quality of the matching sediment chemistry and toxicity data contained in the sediment toxicity database (i.e., SedTox). First, all of the data that were hand entered in the database were verified against the original data source on a number for number basis (i.e., 100% data verification). In addition, 10% of the data (i.e., 10% of the samples and 10% of the COPCs) that were received electronically were verified on a number for number basis to assure that data translation was accurate. Furthermore, a series of outlier checks (e.g., maximum and minimum analyses) and spot checks of the data were implemented to further ensure that project data quality objectives were met. Finally, a quality assurance review of the database development procedures was conducted by the project manager. Application of these quality assurance procedures was intended to ensure that only high quality and fully verified data were incorporated into the project database.

4.3.6 Development of Concentration-Response Relationships for COPC Mixtures

The development of concentration-response relationships for COPCs mixtures represents a key component of the overall SedQC derivation process. To facilitate this step of the process, the Federal-Provincial Technical Steering Committee examined several methods for assessing the effects of COPC mixtures on sediment-dwelling organisms. For example, Long *et al.* (1998) developed a procedure for evaluating the biological significance of contaminant mixtures in marine and estuarine sediments through the application of mean SQG-quotients (SQG-Qs). These mean SQG-Qs were calculated as the arithmetic mean of the SQG-Q that was determined for each measured substances, where SQG-Q = concentration of a substance divided by the SQG for that substance. Subsequently, USEPA (2000) and Ingersoll *et al.* (2001) evaluated 11 different procedures for calculating mean SQG-Qs and concluded that the *"Mean-MPP (or)"* procedure yielded the most robust (i.e., included the largest number of samples) and reliable (i.e., concordance between sediment chemistry and toxicity) results for freshwater sediments (see Macfarlane *et al.* 2003, Appendix 1 for an example calculation).

In this investigation, concentration-response relationships for COPC mixtures in freshwater and in marine or estuarine sediments were developed using the matching sediment chemistry and toxicity data compiled in the SedTox database. For both freshwater and for marine or estuarine sediments, the measured concentrations of COPCs were used together with the preliminary benchmarks to calculate mean SedQC-Qs for each sediment sample represented in the database. More specifically, mean SedQC-Qs were calculated by determining the arithmetic mean of the average SedQC-Q for metals, the SQG-Q for tPAHs, and the SQG-Q for tPCBs (i.e., the "*Mean-PPP (or)"* procedure that was established by USEPA 2000). For freshwater sediments, the response of sediment-dwelling organisms to exposures to COPC mixtures was evaluated using the results of 28-d to 42-d whole sediment toxicity tests with the amphipod, *Hyalella azteca* (endpoint: survival and growth; Table 6) and for marine and estuarine sediments, 10-d whole sediment toxicity tests with the amphipods, *Rhepoxynius abronius* and *Ampelisca abdita* (endpoint: survival; Table 7). In both cases, sediment samples were designated as toxic if the measured response of amphipods exposed to field-collected sediments was significantly greater than the response measured for amphipods exposed to negative control or reference sediments. All of the available data in the sediment toxicity (SedTox) database on the responses of these test organisms to contaminant challenges were used to generate the concentration-response relationships for COPC mixtures (i.e., data from throughout North America was utilized in the analysis).

Development of the concentration-response relationships from the matching sediment chemistry and toxicity data involved several steps. First, all of the data for a sediment type (i.e., for freshwater or for marine and estuarine) were sorted in ascending order according to the mean SedQC-Q. Sediment samples were then grouped into a number of concentration intervals (i.e., groups of samples with similar concentrations of COPCs) that contained 15 samples for freshwater sediments and 25 for marine and estuarine sediments. For each group of sediment samples, the geometric mean of the SedQC-Q and incidence of toxicity (i.e., percent of samples designated as toxic) was determined. Subsequently, the relationship between mean PEC-Qs (concentration) and incidence of toxicity (response) was evaluated by developing three parameter logistic regression models using the data for each concentration interval.

The relationship between the concentration of COPCs in freshwater sediments and the response of the amphipod, *Hyalella azteca*, is presented in Figure 3. These results demonstrate that the incidence of toxicity to freshwater amphipods increases markedly with increasing concentrations of COPCs (r^2=0.99; $p<0.001$; n=303). The resultant logistic regression model (i.e., regression equation) was used to calculate point estimates (i.e., P-values) of adverse effects thresholds, including a P_{20}-value (i.e., the mean SQG-Q that corresponds to a 20% probability of observing toxicity) and a P_{50}-value (i.e., the mean SQG-Q that corresponds to a 50% probability of observing toxicity). Application of the logistic regression model indicated that mean PEL-Qs of 0.6 (i.e., P_{20} value) and 1.3 (i.e., P_{50} value) were associated with a 20% and 50% probability, respectively, of observing significant toxicity to freshwater amphipods (i.e., about an EC_{20} effect concentration).

Figure 4 shows the relationship between the concentration of COPCs in marine and estuarine sediments and the response of the amphipods, *Ampelisca abdita* and *Rhepoxynius abronius*. As was the case for freshwater amphipods, the incidence of toxicity to marine and estuarine amphipods increases markedly with increasing mean PEL-Qs. Although it was not possible to generate a P_{20} value from the resultant concentration-response relationship, a P_{50} value of 1.15 was calculated using the resultant logistic model.

4.4 Refinement of the Preliminary Benchmarks for Sediment Chemistry

The preliminary benchmarks for sediment chemistry were refined using the results of logistic regression modelling of matching sediment chemistry and toxicity data. More specifically, the freshwater $SedQC_{SCS}$ were derived by multiplying the freshwater PEL for each COPC by the mean PEL-Q that corresponded to a 20% probability of observing toxicity to freshwater amphipods (i.e., 0.62). By comparison, the SedQC for typical freshwater sites were derived by multiplying the PEL for each COPC by the average of the P_{50} values for freshwater and for marine and estuarine amphipods (i.e., 1.2). The P_{50} values for freshwater and for marine or estuarine were averaged because they were not considered to be statistically different from one another (i.e., 1.15 vs. 1.30).

The preliminary benchmarks for marine and estuarine sediments were also refined using the toxicity thresholds that were developed from the concentration-response relationships. For sensitive sites, the SedQC were derived by multiplying the marine and estuarine PEL by the P_{20} for freshwater amphipods (i.e., 0.62). The freshwater P_{20} values was used in this application because it was not possible to calculate a marine and estuarine P_{20} value and because the freshwater, estuarine, and marine P_{50} values were nearly the same. The SedQC for typical marine or estuarine sites were calculated by multiplying the PEL for each COPC by the average of the P_{50} values for freshwater and marine or estuarine amphipods (i.e., 1.2). The rationale for this decision is the same as that for SedQC for typical freshwater sites.

The SedQC for assessing and managing contaminated sediments at sensitive sites in British Columbia (i.e., $SedQC_{SCS}$) are presented in Table 4. The corresponding $SedQC_{TCS}$ are presented in Table 5.

Chapter 5 Evaluation of the Numerical Sediment Quality Criteria for Assessing and Managing Sediment Contaminated Sites in British Columbia

5.0 Introduction

Effects-based SedQCs are required to support the assessment and management of sediment contaminated sites in British Columbia. The approach that was used to establish and refine the preliminary benchmarks for assessing and managing contaminated sediments is described in Chapter 4 of this document. While such SedQCs are considered to be generally consistent with the SMOs for contaminated sites that have been established by the Federal-Provincial Technical Steering Committee, the relevance of these SedQC needs to be demonstrated to provide stakeholders with an understanding of the confidence that can be placed in these tools. In this way, stakeholders can make informed decisions regarding the application of the criteria-based or risk-based approaches at contaminated sediment sites in the province.

A variety of approaches have been used previously to evaluate sediment quality benchmarks. In general, these approaches fall into three main categories, including evaluations of comparability, evaluations of reliability, and evaluations of predictive ability (MacDonald *et al.* 1996). More specifically, comparability describes the extent to which the SedQC are similar in value to other sediment quality benchmarks with similar narrative intent. By comparison, reliability describes the extent to which the SedQC meet their narrative intent (i.e., as described in the SMOs), based on the information that was used to derive the SedQC. Finally, predictive ability describes the extent to which the SedQC meet their narrative intent, based on the information contained in an independent database.

This chapter describes the strategy that was used to evaluate the reliability of the SedQC. More specifically, this chapter describes the efforts that were made to acquire matching sediment chemistry and toxicity data from British Columbia and elsewhere in the Pacific

Northwest and from sites located throughout North America. In addition, the methods that were used to review and evaluate each of the candidate data sets are described. Furthermore, the procedures that were used to compile the highest quality data sets in a regional sediment toxicity database and a North American sediment toxicity database are described. Finally, the methods that were used to evaluate the reliability of the SedQC and the results of those evaluations are presented.

5.1 Acquisition of Candidate Data Sets

The procedures that were use to acquire candidate data sets for evaluating the numerical SedQC are described in Section 4.3.1 of this document. Briefly, this process involved accessing the data sets with matching sediment chemistry and toxicity data from the BEDS (MacDonald *et al.* 1996), accessing the papers that have been published in the peer-reviewed literature, and contacting various experts in the field to obtain recently available data. Hard copies of all candidate data sets were retrieved from the applicable source to support subsequent review and evaluation of the information.

5.2 Review and Evaluation of Candidate Data Sets

The procedures that were use to review and evaluate candidate data sets for evaluating the numerical SedQC are described in Section 4.3.2 of this document. Briefly, the metadata obtained with each candidate data set were reviewed to determine its scientific and technical validity. The selection criteria presented in Appendix 1 were used to support the evaluation of candidate data sets. These criteria provided a means of consistently evaluating the methods that were used in each study, including the procedures that were used to collect, handle, and transport sediment samples, the protocols that were applied to conduct sediment toxicity tests, the methods that were used to determine the concentrations of chemicals of concern in sediments, and the statistical tests that were applied to the study results.

5.3 Development of the Freshwater and the Marine and Estuarine Sediment Toxicity Databases

All of the matching sediment chemistry and toxicity data that met the selection criteria were incorporated into the project database on a per sample basis. Each record in the resulting database included the citation, a brief description of the study area (i.e., by waterbody and reach), a description of the sampling locations (including georeferencing data, if available), information on the toxicity tests that were conducted (including species tested, endpoint measured, test duration, etc.), type of material tested (i.e., whole sediment, pore water, or elutriate), TOC concentrations (if reported), and the chemical concentrations (expressed on a dry weight basis). Other supporting data, such as SEM concentrations, AVS concentrations, and particle size distributions, were also included as available.

Individual sediment samples were designated as toxic or not toxic based on comparison of the measured response for that sample to the response for the control or reference samples. More specifically, the sediment samples tested with *Ampelisca abdita* or *Rhepoxynius abronius* were designated as toxic if survival was significantly different from the control (based on analysis of variance; ANOVA) and control-adjusted survival was <80% (Thursby *et al.* 1997). For *Hyalella azteca* survival, sediment samples were designated as toxic if there was a significant reduction in survival relative to a control (based on ANOVA) and the control-adjusted survival was <80% (Long and MacDonald 1998). For *Hyalella azteca* growth, sediment samples were designated as toxic if there was a significant reduction in amphipod length relative to a control (based on ANOVA) and the control-adjusted length was <90% (USEPA 2001). If the results for the control treatment were unavailable, then the responses for sediment samples from the study area were compared to those for appropriately selected sediment samples from reference areas (i.e., reference sediments; ASTM 2003b).

To support subsequent interpretation of the sediment chemistry data, the total concentrations of several chemical classes were determined for each sediment sample (see Section 4.3.2). In calculating the total concentrations of the various chemical classes, less than detection limit values were assigned a value of one-half of the detection, except when the detection limit was greater than the PEL; MacDonald *et al.* 1996; Smith *et al.* 1996). In this latter case, the less than detection limit value was not used in the calculation of the total concentration of the substance or in the calculation of mean SedQC-Qs.

In total, two project databases were developed, including a North American freshwater database and a North American marine and estuarine database. The North American freshwater database included all of the matching sediment chemistry and toxicity from anywhere in Canada or the United States (Table 6). The North American marine and estuarine database included all of the matching sediment chemistry and toxicity from any nearshore area in Canada or the United States (Table 7). These two databases provided a robust basis for evaluating the numerical SedQC.

5.4 Evaluation of the Reliability of the Numerical Sediment Quality Criteria

In this study, reliability was defined as the ability of SedQC to correctly predict toxicity to selected sediment-dwelling organisms. The reliability of the SedQC was evaluated using the matching sediment chemistry and toxicity data that were compiled in the project databases. Because the relationship between the concentration of an individual COPC and toxicity in field-collected sediments is frequently complicated by the presence of multiple contaminants, the data on samples that were designated as toxic were further screened before they were used in the reliability analyses. This screening process was conducted to minimize the potential for including samples in which the selected COPC did not contribute substantially to the observed toxicity. Following the screening approach used by Ingersoll *et al.* (1996) and Field *et al.* (1999; 2002), the concentration of the selected COPC in each toxic sample was compared to the mean concentration of the substance in the nontoxic samples collected in the same study and geographic area. If the concentration of the COPC in an individual toxic sample was less than or equal to the mean concentration of that COPC in the nontoxic samples, it was considered to be highly unlikely that the observed toxicity could be attributed to that substance. Therefore, these toxic samples were not included in the screened data set used to evaluate the reliability of that substance. All nontoxic samples were included in the analysis, however.

The assessment of the numerical criteria for the protection of sediment-dwelling organisms focussed on SedQC for seven trace metals, 13 individual PAHs, total PAHs, total PCBs, nine OC pesticides, and 2,3,7,8-tetrachlordibenzo-*p*-dioxin toxic equivalents (TCDD TEQs).

Using the matching sediment chemistry and toxicity data, reliability was evaluated by calculating the percent incidence of toxicity within the ranges of COPC concentrations defined by the SedQC. The $SedQC_{SCS}$ for a specific COPC was considered to be highly reliable if the incidence of toxicity was $\leq$20% at concentrations below the $SedQC_{SCS}$ (i.e., if the probability of observing an EC_{20} or greater was less than or equal to 20%). By comparison, the $SedQC_{TCS}$ were considered to be highly reliable if the incidence of toxicity was $\geq$50% at concentrations below the $SedQC_{TCS}$ (i.e., if the probability of observing an EC_{20} or greater was greater than or equal to 50%).

An SedQC was considered to be moderately reliable if the incidence of toxicity was within 10% of the narrative objective articulated for that SedQC, while a larger deviation from the narrative objective rendered the SedQC to be less reliable. The reliability of a SedQC was determined only if $\geq$10 samples were available for a specific concentration range (e.g., below the $SedQC_{SCS}$).

In freshwater sediments, the $SedQC_{SCS}$ were generally found to provide a reliable basis for identifying COPC concentrations below which there is a low probability of observing toxicity to amphipods (i.e., in 28 to 42-d toxicity tests; Table 8). For metals, the incidence of sediment toxicity ranged from 13% (lead; n=203) to 32% (chromium; n=72) at concentrations below the $SedQC_{SCS}$. The incidence of toxicity to freshwater amphipods was also low (8 to 21%; n=145 to 230) when concentrations of individual PAHs or total PAHs were below the $SedQC_{SCS}$. For total PCBs, the incidence of toxicity was 7% (n=123) at concentrations below the $SedQC_{SCS}$. The incidence of sediment toxicity was also less than 20% (n=27 to 34) at concentrations below the $SedQC_{SCS}$ for seven of nine OC pesticides, with the exceptions being endrin (25%; n=178) and lindane (47%; n=45). By comparison, the incidence of sediment toxicity was generally much higher (i.e., 50 to 100%; n=1 to 80) at COPC concentrations above the $SedQC_{SCS}$, which indicates that adverse effects are likely to occur when the $SedQC_{SCS}$ is exceeded. Collectively, these results indicate that the $SedQC_{SCS}$ are generally consistent with the SMO that were established for sensitive contaminated sites. The freshwater $SedQC_{SCS}$ were considered to be moderately or highly reliable for 30 of the 33 COPCs evaluated (Table 9).

The $SedQC_{TCS}$ for freshwater sediments were found to provide a reliable basis for identifying COPC concentrations above which there is a relatively high probability of observing toxicity to amphipods (Table 8). For metals, the incidence of sediment toxicity ranged from 62% (zinc; n=45) to 89% (cadmium; n=36) at concentrations above the $SedQC_{TCS}$. The incidence of toxicity to freshwater amphipods was also elevated (44% to 89%; n=20 to 45) when concentrations of individual PAHs or total PAHs were above the $SedQC_{TCS}$. For total PCBs, the incidence of toxicity above the $SedQC_{TCS}$ was 54% (n=36). The incidence of sediment toxicity ranged from 69 to 100% (n=1 to 36) at concentrations above the $SedQC_{TCS}$ for the nine OC pesticides considered in this evaluation. Collectively, these results indicate that the $SedQC_{TCS}$ are generally consistent with the SMOs that were established for typical contaminated sites and indicate that there is a high probability of observing sediment toxicity at COPC concentrations above the $SedQC_{TCS}$. The freshwater $SedQC_{TCS}$ were considered to be moderately or highly reliable for 27 of the 33 COPCs evaluated (Table 9). The probability of observing sediment toxicity at COPC concentrations below the freshwater $SedQC_{TCS}$ was typically less than 50%.

In marine and estuarine sediments, the $SedQC_{SCS}$ were also found to generally provide a reliable basis for identifying COPC concentrations below which there is a low probability of observing toxicity to amphipods (i.e., based on the results of 10-d toxicity tests; Table 10). For metals, the incidence of sediment toxicity was $\leq$20% at concentrations below the $SedQC_{SCS}$ for four of the seven metals considered, with the exceptions being arsenic (27%; n=1780), cadmium (22%; n=1718), and chromium (22%; n=1516). The incidence of toxicity to marine and estuarine amphipods was also low (12 to 19%; n=1163 to 1467) when concentrations of individual PAHs or total PAHs were below the $SedQC_{SCS}$. For total PCBs, the incidence of toxicity was 11% (n=1207) at concentrations below the $SedQC_{SCS}$. The incidence of sediment toxicity was also less than 20% at concentrations below the $SedQC_{SCS}$ for eight of nine OC pesticides (n=927 to 1225), with the exception being Sum DDE (22%; n=1546). Finally, the incidence of toxicity was 20% (n=20) in marine and estuarine sediments with concentrations of 2,3,7,8-TCDD TEQs below the $SedQC_{SCS}$. By comparison, the incidence of sediment toxicity was greater than 50% for 31 of 33 at COPC concentrations above the $SedQC_{SCS}$. The incidence of toxicity was somewhat lower above the $SedQC_{SCS}$ for two substances, including lindane (39%; n=103) and Sum DDE (43%; n=60). Collectively, these results indicate that the marine and estuarine $SedQC_{SCS}$ are generally consistent with the SMOs that were established for sensitive contaminated sites (i.e., the $SedQC_{SCS}$ were moderately or highly reliable for all 33 COPCs; Table 11).

The marine and estuarine $SedQC_{TCS}$ were found to provide a reliable basis for identifying COPC concentrations above which there is a relatively high probability of observing sediment toxicity (Table 10). For metals, the incidence of sediment toxicity ranged from 33% (arsenic) to 78% (cadmium) at concentrations above the $SedQC_{TCS}$ for marine and estuarine sediments. The incidence of toxicity to marine and estuarine amphipods was also elevated (61 to 79%) when concentrations of individual PAHs or total PAHs were above the $SedQC_{TCS}$. For total PCBs, the incidence of toxicity above the $SedQC_{TCS}$ was 69% for marine and estuarine sediments. The incidence of sediment toxicity was greater than 50% at concentrations above the $SedQC_{TCS}$ for 7 of 9 OC pesticides in marine and estuarine sediments. Collectively, these results indicate that the $SedQC_{TCS}$ are generally consistent with the SMOs that were established for typical contaminated sites. The marine and estuarine $SedQCT_{SCS}$ were considered to be moderately or highly reliable for 29 of the 33 COPCs evaluated (Table 11). The probability of observing sediment toxicity at COPC concentrations below the marine and estuarine $SedQC_{TCS}$ was typically less than 50%.

The highly reliable and moderately reliable SedQC should be used directly at contaminated sites in the province. In addition, those SedQC with lower reliability can also be used to assess and manage sediment quality conditions. However, a responsible person may wish to derive site-specific SedQC in such cases to reduce uncertainty in the assessment.

Chapter 6 Applications of the Sediment Quality Criteria for Assessing and Managing Sediment Contaminated Sites in British Columbia

6.0 Introduction

Sediment quality criteria represent the benchmarks against which sediment quality conditions are measured at contaminated sites in British Columbia. Such SedQC provide essential tools for assessing potentially contaminated sediments and establishing clean-up targets for remedial actions. This chapter of the report is intended to provide guidance on the application of numerical SedQC for assessing and managing sediment contaminated sites in the province. Accordingly, the recommended uses of the SedQC are identified. In addition, the procedures for determining if a site is contaminated are described. Furthermore, the methods that can be used to establish sediment quality standards (SQSs; i.e., remedial action targets or preliminary remediation goals) are discussed.

6.1 Uses of Numerical Sediment Quality Criteria

Numerical SedQC are intended to serve as benchmarks which define the conditions needed to protect sediment-dwelling organisms, wildlife, and human health at sites with contaminated sediments. These benchmarks may be used in a variety of ways, including:

- As indicators of sediment quality at a site (i.e., during site screening);
- For identifying the COPCs (i.e., during site investigation);
- To support the design of sampling programs (i.e., during site investigation);
- For interpreting sediment chemistry data (i.e., during site investigation);

- For identifying potentially unacceptable levels of risk to the environment at a site (i.e., during site investigation);
- For determining if a site is contaminated (i.e., during site investigation);
- For determining the factors that are most likely associated with measured or potential effects (i.e., to assist with the interpretation of sediment toxicity data);
- For determining if site remediation, risk assessment, or risk management are necessary (i.e., following detailed site investigation);
- As a basis for establishing site management goals and remediation targets (i.e., during remedial action planning);
- As a basis for developing legally-enforceable standards (i.e., during remediation planning);
- For evaluating the adequacy of site remediation (i.e., confirming that site remediation has been successfully completed); and,
- For the purposes of issuing certification of satisfactory site remediation.

The criteria are not intended to be applied or interpreted as thresholds to pollute up to. Nor should they be interpreted as acceptable thresholds for ambient environmental quality outside of the boundaries of a contaminated site.

6.2 Determining if a Site is Contaminated

One of the most important uses of the SedQC is for determining if a site is contaminated, as defined under the CSR. In this application, the SedQC are used during Stage 1 or Stage II of the preliminary site investigation (MacDonald and Ingersoll 2003b). In the Stage I PSI, the existing sediment chemistry data for the site are collected, collated, and evaluated to determine if they are sufficient for making the determination. Some of the factors that need to be considered when evaluating the existing data include: the age of the data, the geographic coverage of the data, the analytes measured (as compared to the COPCs for the site), the quality of the data (i.e., accuracy, precision, detection limits), sampling depth, and the sampling design utilized. In the event that insufficient data are available, then a Stage

II PSI needs to be conducted to acquire the sediment chemistry data required to complete the determination.

Following acquisition of the necessary and sufficient information on the chemical characteristics of whole sediments, the sediment chemistry data for the site is compared to the numerical SedQC. The $SedQC_{SCS}$ are used at sites that are considered to contain sensitive habitats, while the $SedQC_{TCS}$ are employed at typical sites. To ensure the proper application of the SedQC, administrative rules have been established to guide determinations of sites as contaminated or uncontaminated. These administrative rules state that:

1. A sensitive site or a typical site is a contaminated site if any of the following conditions exist:
   - The 90th percentile concentration of one or more COPCs equals or exceeds their respective SedQC (i.e., 9 of 10 measurements must be below the SedQC to designate a site as uncontaminated) and exceeds upper limit of background for that substance (i.e., mean + 2SD);
   - The concentration of one or more analytes exceeds their respective SedQC by a factor of two or more in any sediment sample and exceeds upper limit of background for that substance (i.e., mean + 2SD);
   - The 90th percentile mean SedQC-Q for the contaminant mixture equals or exceeds 1.0; or,
   - The mean SedQC-Q for the contaminant mixture in any sediment sample equals or exceeds 2.0.

2. The $SedQC_{SCS}$ are to be applied to a depth of 100 cm (i.e., 0-100 cm) in areas where the sediment bed has been demonstrated to be stable (i.e., non-erosional, not subject to navigational dredging, etc.).

3. The $SedQC_{SCS}$ will apply to depths of greater than 100 cm in areas where the sediment bed has been demonstrated to be unstable (i.e., erosional, subject to navigational dredging, etc.) or the stability of the bed is unknown; or it is demonstrated that there is on-going transport of contaminants at depth into the

shallower portions of the sediment bed at rates capable of contaminating sediments in the top 100 cm to levels exceeding the $SedQC_{SCS}$.

4. The $SedQC_{SCS}$ must be used during the site investigation process to determine if a sensitive site contains contaminated sediments.

5. The $SedQC_{SCS}$ will apply at contaminated sites that have sediments that border or include habitat protection or conservation zones, or where biological habitat mapping (e.g., such as has been conducted by Fraser River Estuary Management Program; FREMP or Burrard Inlet Environmental Action Program; BIEAP) has designated the area as a high productivity zone. Table 2 provides a checklist of factors to be considered in applications to the Ministry in support of the selection of SedQC values.

6. The $SedQC_{SCS}$ should be used to determine if remedial measures are needed at a sensitive site and to establish target clean-up goals for contaminated sediments.

7. The $SedQC_{TCS}$ are to be applied to a depth of 100 cm (i.e., 0-100 cm) in areas where the sediment bed has been demonstrated to be stable (i.e., non-erosional, not subject to navigational dredging).

8. The $SedQC_{TCS}$ will apply to depths of greater than 100 cm in areas where the sediment bed has been demonstrated to be unstable (i.e., erosional, subject to navigational dredging) or the stability of the bed is unknown; or it is demonstrated that there is on-going transport of contaminants at depth into the shallower portions of the sediment bed at rates capable of contaminating sediments in the top 100 cm to levels exceeding the $SedQC_{TCS}$.

9. The $SedQC_{TCS}$ must be used during the site investigation process to determine if a typical site contains contaminated sediments.

10. The $SedQC_{TCS}$ should be used to determine if remedial measures are needed at a typical site and to establish target clean-up goals for contaminated sediments.

11. The presence of sediments containing contaminant concentrations qualifying as Special Wastes, as defined under the SWR, necessitates the imposition of limitations on potential remedial actions. Where Special Waste is present, remedial measures should focus on the removal of these wastes, to the extent feasible. The handling, treatment and disposal of these materials is to be conducted in accordance with the provisions of the SWR.

Application of these administrative rules provides a consistent basis for determining if a site is contaminated, as defined under the CSR.

6.3 Development of Sediment Quality Standards

Further action is required at sites that are deemed to be contaminated (Figure 2). First, a Stage II PSI or DSI is conducted to acquire the information needed to confirm that the site is contaminated and to evaluate the nature, severity, and areal extent of such contamination (MacDonald and Ingersoll 2003b). Next, the person or parties that are responsible and liable for the contamination are identified. Subsequently, a feasibility study is conducted to assess the need and priority for remedial action. A voluntary remediation agreement can then be established or a remediation order is issued to activate the remediation process. A remedial action plan (RAP) is then developed and submitted to the Ministry for approval. Following approval of the RAP, the responsible parties can conduct remedial measures at the site. Finally, monitoring activities are conducted at the site to determine if the remedial measures have reduced COPC concentrations or risks to tolerable levels.

A key element of the remedial action planning process involves the establishment of SQSs to guide remedial activities. Such SQSs (which are also termed remedial action targets or preliminary remediation goals) identify the concentrations of sediment-associated COPCs that need to be achieved to meet the SMOs (i.e., remedial action objectives) for the site. Alternatively, risk-based criteria can be established to support evaluations of the extent to which the SMOs are being met at the site. The procedures that can be used to establish generic criteria-based SQS, site-specific criteria-based SQSs, and risk-based SQSs are described in the following sections of this report.

6.3.1 Generic Remedial Action Targets

Under the CSR, generic numerical SedQC have been established to support the assessment and management of sediment contaminated sites in the province. Such generic SedQC are intended to protect human health and the environment at any site, without consideration of site-specific features other than land and water use. The generic SedQC that have been established for sensitive sites and typical sites in British Columbia are presented in Table 4 and 5, respectively. Under many circumstances, the generic SedQC can be used directly to assess sediment quality conditions and establish SQSs for the site. However, it is prudent to evaluate the relevance of the generic SedQC before adopting them as SQSs at a site.

Determination of the range of natural background concentrations of metals and certain organic contaminants (e.g., petroleum hydrocarbons) at the site under consideration is essential to ascertain if the generic SedQC are realistic for application at a site. Two procedures may be used to establish natural background concentrations of COPCs in bedded sediments, including:

- For organic contaminants, nearby, uncontaminated reference sites should be selected on the basis of its similarity to the contaminated site. Sediments should then be sampled to characterize background conditions (i.e., sediment chemistry) at the reference sites. The generic SedQCs can then be compared to the upper limit of background concentrations at the reference sites (i.e., mean plus two standard deviations).

- For metals in freshwater, estuarine, and marine, relationships between metal concentrations and the levels of reference elements (e.g., aluminum, lithium, etc.) in uncontaminated sediments should be used to estimate natural background conditions (using the methods described by Schropp *et al.* 1990; Loring 1991; Carvalho and Schropp 2002). Specifically, the plots of metal to reference element concentrations at reference sites should be prepared. These plots should include the regression equation and the 95% prediction limits. The upper limit of background for each metal may then be established as the upper 95% prediction limit. Data from the contaminated site is also represented on this plot to facilitate comparison with the background data.

If the generic SedQC is greater that the upper limit of background concentrations, then this provisional value would be further evaluated to determine its applicability to the site under consideration. Conversely, if the generic SedQC is lower than the estimated background level of a contaminant, then the generic criterion would not be directly applicable to the contaminated site. In this situation, site-specific SQSs should be developed for the site.

6.3.2 Site-Specific Sediment Quality Standards

As an alternative adopting the generic SedQC directly as SQSs, a responsible party may choose to derive site-specific SQSs for a site. A number of procedures may be used to modify the generic SedQC to reflect site-specific conditions. Each of the recommended procedures for modifying the generic SedQC will result in the derivation of a preliminary SQS (PSQS), which must be evaluated to assess its applicability to the site under consideration. If this PSQS satisfies all of the evaluation criteria, then it is adopted as the recommended SQS. However, the PSQS may require further modification if one or more of the evaluation criteria are not satisfied.

Under certain circumstances, it may be determined that the data used to derive the PSQS are not directly applicable to the site under consideration. For example, matrix SQS for the protection of aquatic life may have been derived using the BEDS, which generally considers data from throughout North America and encompasses a diverse array of species and endpoints. However, the results of regional sediment sampling may indicate that only a limited number of species occur or are expected to occur at the contaminated site. Under these circumstances, the PSQS may be recalculated using only the information that is relevant to the water body under consideration. The administrative rules presented in Appendix 2 provide a basis for assessing the applicability of the available toxicological data to a specific site.

The recalculation procedure for modifying the PSQS to account for the sensitivity range of species that occur or are expected to occur involves three steps. The first step in this process is to compile the toxicological data for those species that occur or are likely to occur at the site, in the absence of contamination. Specifically, data on species of sediment-dwelling organisms representing orders that do not occur within the system under consideration may be excluded from the database. Next, the CCME (1995) protocol should be used to derive the site-specific PEL, if all of the necessary conditions identified in that document are met (i.e., minimum data requirements). Finally, the SedQC is calculated by multiplying the PEL by the appropriate factor (i.e., 0.62 for sensitive sites and 1.2 for typical sites). As indicated previously, evaluation of the PSQS so derived will provide a means of assessing its relevance to the contaminated site.

The PSQS for non-polar organic contaminants (i.e., PCBs, PAHs, certain pesticides) in marine and estuarine ecosystems may be modified if the site under consideration has atypical levels of TOC. The Canadian SQGs for marine and estuarine ecosystems are considered to apply directly to sediments with relatively low levels of TOC (roughly 1.2 $\pm$ 1.8% TOC). If median level of TOC in the sediments at the site falls outside the 95% confidence interval (i.e., 0.1 to 4.7%), then the PSQS may be modified to account for the predicted bioavailability of the substance under the conditions at the site. Likewise, the PSQS for freshwater sediments may be modified if TOC levels fall outside the typical range (0.4 to 10.1%).

A number of specific procedures could be used to adapt the PSQS to reflect site-specific sediment characteristics. For example, SAIC (1991) recommended that the lowest level of TOC measured at the site be used to establish the site-specific SedQC. However, it is likely that this procedure would yield overly conservative values under many circumstances (i.e., when there is high variability in the levels of TOC or low levels of TOC occur only infrequently at the site). Therefore, an alternate procedure is recommended for modifying the PSQS to account for atypical levels of TOC at the site. Specifically, it is recommended that the 10th percentile TOC values for the site (TOC_{site}; expressed as a percentage) and for BEDS (TOC_{BEDS}) be used as a basis for modifying the PSQSs, as follows:

$$\mathbf{PSQS_{new} = TOC_{site} \ / \ TOC_{BEDS} \ x \ PSQS}$$

This procedure is likely to support the derivation of SQGs that are generally applicable to the site. However, it should be noted that carbon-based contaminants (oil, grease, PAHs, etc.) may comprise a significant proportion of the total TOC at contaminated sites. Rather than mitigating toxicity, this contaminant-dominated TOC may actually contribute to toxicity. For example, sediment-associated TOC was significantly positively correlated with toxicity in sea urchin pore-water tests conducted in Tampa Bay (Long *et al.* 1995a). Therefore, care should be exercised when the PSQS is modified to account for high levels of TOC, with only the non-contaminant TOC used in the calculation (i.e., by subtracting the concentrations of organic contaminants).

Acid volatile sulfide has been identified as an important factor influencing the bioavailability of divalent metals (see Di Toro *et al.* 1990; 1992; Ankley *et al.* 1996). Specifically, the results of several investigations have indicated that metals are unlikely to cause toxicity when SEM concentrations are lower than the concentrations of AVS (when each are expressed on a molar basis; i.e., SEM - AVS < 0.00; e.g., Hansen *et al.* 1996; Berry *et al.* 1996). However, the SEM-AVS tool did not predict the absence of toxic effects more accurately than dry-weight normalized SQGs (Long *et al.* 1998b). For this reason, adjustment of SQSs for metals for AVS concentrations may be of limited utility.

6.4 Risk-Based Sediment Quality Standards

Risk-based procedures provide another option for establishing SQS at contaminated sites in British Columbia. The risk-based approach cannot be used to determine if a site is contaminated, but may be used to establish remediation standards for the site. In contrast to numerical standards, risk-based standards do not identify the levels of sediment-associated contaminants that are needed to restore the designated uses of the sediment resource. Rather, risk-based procedures provide a means of determining the risks to human health and the environment that are posed by ambient concentrations of contaminants. These procedures can also be used to identify the concentrations of sediment-associated contaminants that pose tolerable risks to human health and the environment. Therefore, the risk-based approach tends to focus primarily on risk management (i.e., reducing exposures).

From a regulatory perspective, application of risk-based standards may introduce more complexity and uncertainty into the contaminated site remediation process. For this reason, regulatory agencies may apply a number of institutional controls at these sites to assure that human health and the environment are adequately protected in the long-term. For example, long-term monitoring may be required after the risk management actions have been completed at the site. Other conditions may also have to be met to be in compliance with the provisions of the *WMA*. The Certificate of Compliance, which is issued to the site owner/operator following successful completion of the remedial measures, provides a list of specific conditions that must be met at the site to remain in compliance with the *Act*.

This document does not provide specific guidance on the development or implementation of risk-based SQSs. The reader is directed to Recommended Guidance and Checklist for Tier 1 Ecological Risk Assessment of Contaminated Sites in British Columbia for more information on the application of ecological risk assessment at contaminated sites in British Columbia (Landis *et al*. 1997). A list of policy decisions regarding the use of ecological risk assessment procedures is available from the Ministry.

6.5 Establishment of the Final Sediment Standards

The recommended procedures for deriving SQSs are designed to provide practitioners in this field with general guidance on the technical aspects of the contaminated site assessment and remediation process. However, these procedures are not intended to provide the sole basis for establishing SQSs at contaminated sites. Instead, they are designed to support the derivation of recommended SQSs for these sites, which are science-based. The final SQSs which are ultimately adopted at a site may consider other factors as well.

An evaluation of the technical feasibility and costs associated with site cleanup is required to assess the practicality of adopting the recommended SQSs at a contaminated site. This step in the process is designed to determine if the use protection goals that were originally identified for the site were realistic and achievable. If the available information suggests that the existing technology would not be adequate to facilitate cleanup to the SQS or that the cost-benefit ratio associated with remediation to the proposed SQS would not be favourable (i.e., beyond the point of diminishing returns), then it will not be feasible to achieve the recommended SQSs. Under this scenario, a management decision might be made to sacrifice one or more of the use protection goals for the aquatic ecosystem at the contaminated site. However, it is absolutely essential to maintain transparency in this decision-making process and effectively communicate such decisions to the public.

Chapter 7 References Cited

Adams, D.A., J.S. O'Connor, and S.B. Weisberg. 1998. Sediment quality of the NY/NJ Harbor System: An investigation under the Regional Environmental Monitoring and Assessment Program (R-EMAP). EPA/902-R-98-001. Division of Environmental Sciences Assessment. United States Environmental Protection Agency Region II. Edison, New Jersey.

Adams, W.J., R.A. Kimerle, and J.W. Barnett, Jr. 1992. Sediment quality and aquatic life assessment. Environmental Science and Technology 26:1863-1876.

Anderson, B., J. Hunt, S. Tudor, J. Newman, R. Tjeerdema, R. Fairey, J. Oakden, C. Bretz, C.J. Wilson, F. LaCaro, G. Kapahi, M. Stephenson, M. Puckett, J. Anderson, E.R. Long, T. Flemming, and K. Summers. 1997. Chemistry, toxicity and benthic community conditions in sediments of selected southern California bays and estuaries. Sacramento, CA. State Water Resources Control Board, State of California. Sacramento, California.

Ankley, G.T., D.M. Di Toro, D.J. Hansen, and W.J. Berry. 1996. Technical basis and proposal for deriving sediment quality criteria for metals. Environmental Toxicology and Chemistry 15:2056-2066.

ASTM (American Society for Testing and Materials). 2003a. Standard guide for collection, storage, characterization, and manipulation of sediments for toxicological testing and for selection of samples used to collected benthic invertebrates. E1391-03. ASTM 2003 Annual Book of Standards Volume 11.05. West Conshohocken, Pennsylvania.

ASTM (American Society for Testing and Materials). 2003b. Standard test methods for measuring the toxicity of sediment-associated contaminants with freshwater invertebrates. E1706-00. ASTM 2003 Annual Book of Standards Volume 11.05. West Conshohocken, Pennsylvania.

Barrick, R., S. Becker, R. Pastorok, L. Brown, and H. Beller. 1988. Sediment quality values refinement: 1988 update and evaluation of Puget Sound AET. Prepared by PTI Environmental Services for Environmental Protection Agency. Bellevue, Washington

Bay, S., D.Greenstein, J.Brown, and A. Jirik. 1994. Investigation of toxicity in Palos Verdes sediments Final Report 1994. Prepared for Santa Monica Bay Restoration Project. Southern California Coastal Water Research Project. Westminster, California.

BC (British Columbia Environment). 1988. *Waste Management Act*: Special Waste Regulation. B.C. Reg. 63/88. Victoria, British Columbia.

BC (British Columbia Environment). 1997. *Waste Management Act*: Contaminated Sites Regulation. Updated to February 4, 2002. B.C. Reg. 375/96. Victoria, British Columbia.

Berry, W.J., D.J. Hansen, J.D. Mahony, D.L. Robson, D.M. DiToro, B.P. Shipley, B. Rogers, J.M. Corbin, and W.S. Boothman. 1996. Predicting the toxicity of metals-spiked laboratory sediments using acid volatile sulfide and interstitial water normalizations. Environmental Toxicology and Chemistry 15(12):2067-2079.

Carvalho, A. and S.J. Schropp. 2002. Development of an interpretive tool for assessment of metal enrichment in Florida freshwater sediments. Prepared by Taylor Engineering, Inc. Prepared for Florida Department of Environmental Protection. Tallahassee, Florida. 38 pp.

CCME (Canadian Council of Ministers of the Environment). 1995. Protocol for the derivation of Canadian sediment quality guidelines for the protection of aquatic life. CCME EPC-98E. Canadian Council of Ministers of the Environment Task Group on water quality guidelines. Ottawa, Ontario.

CCME (Canadian Council of Ministers of the Environment). 1999. Canadian environmental quality guidelines. Guidelines and Standards Division. Environment Canada. Winnipeg, Manitoba.

Chapman, P.M., R.N. Dexter, and E.R. Long. 1987. Synoptic measures of sediment contamination, toxicity and infaunal community composition (the Sediment Quality Triad) in San Francisco Bay. Marine Ecology - Progress Series 37:75-96.

Chapman, P.M., E.A. Power, R.N. Dexter and H.B. Andersen. 1991. Evaluation of effects associated with an oil platform, using Sediment Quality Triad. Environmental Toxicology and Chemistry 10:407-424.

Cook, P.M., A.R. Batterman, and K.R. Lodge. 1989. Laboratory measurement of lake trout bioaccumulation of 2,3,7,8-TCDD from Lake Ontario sediment. 32nd Conference on Great Lakes Research at the University of Wisconsin, Madison, Wisconsin. United States Environmental Protection Agency. Environmental Research Laboratory. Duluth, Minnesota.

Crane, J.L., D.D. MacDonald, C.G. Ingersoll, D.E. Smorong, R.A. Lindskoog, C.G. Severn, T.A. Berger, L.J. Field. 2000. Development of a framework for evaluating numerical sediment quality targets and sediment contamination in the St. Louis Area of Concern. EPA 905-R-00-008. Great Lakes National Program Office. Chicago, Illinois.

Crane, J.L., D.D. MacDonald, C.G. Ingersoll, D.E. Smorong, R.A. Lindskoog, C.G. Severn, T.A. Berger, and L.J. Field. 2002. Evaluation of numerical sediment quality targets for the St. Louis River area of concern. Archives of Environmental Contamination and Toxicology 43:1-10.

Crocker, P.A., G.J. Guillen, R.D. Seiler, E. Petrocelli, M. Redmond, W. Lane, T.A. Hollister, D.W. Neleigh, and G. Morrison. 1991. Water quality, ambient toxicity and biological investigations in the Houston Ship Channel and tidal San Jacinto River. United States Environmental Protection Agency Region VI. Dallas, Texas.

Cubbage, J., D. Batts, and S. Briedenbach. 1997. Creation and analysis of freshwater sediment quality values in Washington State. Environmental Investigations and Laboratory Services Program. Washington Department of Ecology. Olympia, Washington.

Di Toro, D.M., J. Mahony, D.J. Hansen, K.J. Scott, M.B. Hicks, S.M. Mayr, and M. Redmond. 1990. Toxicity of cadmium in sediments: The role of acid volatile sulfides. Environmental Toxicology and Chemistry 9:1487-1502.

Di Toro, D.M., C.S. Zarba, D.J. Hansen, W.J. Berry, R.C. Swartz, C.E. Cowan, S.P. Pavlou, H.E. Allen, N.A. Thomas, and P.R. Paquin. 1991. Technical basis for establishing sediment quality criteria for nonionic organic chemicals using equilibrium partitioning. Environmental Toxicology and Chemistry 10:1541-1583.

Di Toro, J.D. Mahony, D.J. Hansen, K.J. Scott, A.R. Carlson, and G.T. Ankley. 1992. Acid volatile sulfide predicts the acute toxicity of cadmium and nickel in sediments. Environmental Science and Technology 26(1):96-101.

Downing, J., R. Fairey, C. Roberts, E. Landrau, R. Clark, J. Hunt, B. Anderson, B. Phillips, C. Wilson, F. La Caro, G. Kapahi, K. Worcester, M. Stephenson, and M. Puckett. 1998. Chemical and biological measures of sediment quality in the Central Coast Region. Final Report. California State Water Resources Control Board. Sacramento, California.

EA Engineering, Science, and Technology, Inc. 1994. Results of dredged material testing of samples from the proposed Bayou Casotte Turning Basin, Pascagoula, Mississippi. Prepared for United States Army Engineer District. Mobile, Alabama. Prepared by EA Mid-Atlantic, EA Engineering, Science, and Technology, Inc. Sparks, Maryland.

EC and MENVIQ (Environment Canada and Ministere de l'Envionnement du Quebec). 1992. Interim criteria for quality assessment of St. Lawrence River sediment. ISBN 0-662-19849-2. Environment Canada. Ottawa, Ontario.

Endicott, D., W. Richardson and D.M. Di Toro. 1989. Lake Ontario TCDD modelling report. Large Lakes Research Station. Environmental Research Laboratory-Duluth. United States Environmental Protection Agency. Grosse Ile, Minnesota. 94 pp.

Engle, V. and L. Harwell. 1996. Catalog documentation EMAP-Estuaries Province Level Database Louisianian Province 1991-1993. Station location data and sediment chemistry data and sediment toxicity data. United States Environmental Protection Agency and United States Geological Survey.

Fairey, R., C. Baretz, S. Lamerdin, J. Hunt, B. Anderson, S. Tudor, C. Wilson, F. Lacaro, M. Stephenson, M. Puckett, and E. Long. 1996. Chemistry, toxicity and benthic community conditions in sediments of the San Diego Bay Region. Moss Landing Marine Laboratories. University of California Santa Cruz. California

Ferraro, S.P., R.C. Swartz, F.A. Cole, and D.W. Schults. 1991. Temporal changes in the benthos along a pollution gradient: Discriminating the effects of natural phenomena from sewage-industrial wastewater effects. Estuaries Coastal Shelf Science 33(4):383-407.

Field, L.J., D.D. MacDonald, S.B. Norton, C.G. Severn, and C.G. Ingersoll. 1999. Evaluating sediment chemistry and toxicity data using logistic regression modeling. Environmental Toxicology and Chemistry 18(6):1311-1322.

Field, L.J., D.D. MacDonald, S.B. Norton, C.G. Ingersoll, C.G. Severn, D.E. Smorong, and R.A. Lindskoog. 2002. Predicting amphipod toxicity from sediment chemistry using logistic regression models. Environmental Toxicology and Chemistry 21:1993-2005.

Flegal, A.R., R.W. Risebrough, B. Anderson, J. Hunt, S. Anderson, J. Oliver, M. Stephenson, and R. Packard. 1996. San Francisco estuary pilot regional monitoring program: Sediment studies. Water Quality Control Board. San Francisco Bay Regional Water Quality Control Board State Water Resources Control Board. San Francisco, California.

FRAP (Fraser River Action Plan). 1997. Contaminants in bed sediments from 15 reaches of the Fraser River Basin. DOE-FRAP 1997-37. Aquatic and Atmospheric Division. Environment Canada. Vancouver, British Columbia.

Golder Associates. 1999. Aquatic risk assessment for 8335 Meadow Avenue, Burnaby, B.C. Prepared for Canadian National Railway Company, Beazer East Inc., and Atlantic Industries, Ltd. Burnaby, British Columbia.

Hansen, D.J., W.J. Berry, J.D. Mahoney, W.S. Boothman, D.M. DiToro, D.L. Robson, G.T. Ankley, D. Ma, Q. Yan, and C.E. Pesch. 1996. Predicting the toxicity of metal-contaminated field sediments using interstitial concentration of metals and acid-volatile sulfide normalizations. Environmental Toxicology and Chemistry 15:2080-2094.

Hart Crowser Inc. 1986. Results of chemical and biological analyses of sediment samples, Everett Marina, and Jetty Island Pier, Everett, Washington. Prepared by Hart Crowser and Associates Inc. Seattle, Washington.

Hunt, J.B., B. Anderson, B. Phillips, J. Newman, R. Tjeerdema, K. Taberski, C. Wilson, M. Stephenson, M. Puckett, R. Fairey and J. Oakden. 1998. Sediment quality and biological effects in San Francisco Bay. Final Technical Report. Bay Protection and Toxic Cleanup Program. California State Water Resources Control Board. Sacramento, California.

Hyland, J.L., T.J. Herrlinger, T.R. Snoots, A.H. Ringwood, R.F. Van Dolah, C.T. Hackney, G.A. Nelson, J.S. Rosen, and S.A. Kokkinakis. 1996. Environmental quality of estuaries of the Carolinian Province: 1994. Annual statistical summary for the 1994 EMAP-Estuaries Demonstration Project in the Carolinian Province. NOAA Technical Memorandum NOS ORCA 97. Office of Ocean Resources Conservation and Assessment. Silver Spring, Maryland

Hyland, J.L., L. Balthis, C.T. Hackney, G. McRae, A.H. Ringwood, T.R. Snoots, R.F. Van Dolah, and T.L. Wade. 1998. Environmental quality of estuaries of the Carolinian Province: 1995. Annual statistical summary for the 1995 EMAP - Estuaries demonstration project in the Carolinian Province. NOAA Technical Memorandum NOS ORCA 123. 143 pp. National Oceanic and Atmospheric Administration. Silver Springs, Maryland..

Ingersoll, C.G. and D.D. MacDonald. 1999. An assessment of sediment injury in the West Branch of the Grand Calumet River. Volume I. Prepared for Environmental Enforcement Section. Environment and Natural Resources Division. United States Department of Justice. Washington, District of Columbia. 161 pp.

Ingersoll, C.G., D.R. Buckler, E.A. Crecelius, and T.W. LaPoint. 1993a. Assessment and remediation of contaminated sediments (ARCS) program. Biological and chemical assessment of contaminated Great Lakes sediment. EPA 905/R-93/006. United States Environmental Protection Agency. Chicago, Illinois.

Ingersoll, C.G., W.G. Brumbaugh, A.M. Farag, T.W. La Point, and D.F. Woodward. 1993b. Effects of metal-contaminated sediment, water, and diet on aquatic organisms. NFCRC-UW Final Report for the USEPA Milltown Endangerment Assessment Project. National Technical Information Service PB93-21592. Springfield, Virginia.

Ingersoll, C.G., P.S. Haverland, E.L. Brunson, T.J. Canfield, F.J. Dwyer, C.E. Henke, and N.E. Kemble. 1996. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the midge *Chironomus riparius*. Journal Great Lakes Research 22:602-623.

Ingersoll, C.G., T. Dillon, and R.G. Biddinger (Eds.). 1997. Methodological uncertainty in sediment ecological risk assessment. *In*: Ecological Risk Assessments of Contaminated Sediment. SETAC Press. Pensacola, Florida. 389 pp.

Ingersoll, C.G., E.L. Brunson, F.J. Dwyer, D.K. Hardesty, and N. Kemble. 1998. Use of sublethal endpoints in sediment toxicity tests with the amphipod *Hyalella azteca*. Environmental Toxicology and Chemistry 17:1508-1523.

Ingersoll, C.G., D.D. MacDonald, N. Wang, J.L. Crane, L.J. Field, P.S. Haverland, N.E. Kemble, R.A. Lindskoog, C.G. Severn, and D.E. Smorong. 2001. Predictions of sediment toxicity using consensus-based freshwater sediment quality guidelines. Archives of Environmental Contamination and Toxicology 41:8-21.

Ingersoll, C.G., D.D. MacDonald, W.G. Brumbaugh, B.T. Johnson, N.E. Kemble, J.L. Kunz, T.W. May, N. Wang, J.R. Smith, D.W. Sparks, and S.D. Ireland. 2002. Toxicity assessment of sediments from the Grand Calumet River and Indiana Harbor Canal in northwestern Indiana. Archives of Environmental Contamination and Toxicology 43:156-167.

Jacobi, M., R. Fairey, C. Roberts, E. Landrau, J. Hunt, B. Anderson, B. Phillips, C.J. Wilson, G. Kapahi, F. LaCaro, B. Gwynne, M. Stephenson and M. Puckett. 1998. Chemical and biological measures of sediment quality and tissue bioaccumulation in the North Coast Region. Bay Protection and Toxic Cleanup Program. Final Report. California State Water Resources Control Board. Sacramento, California

Johnson, A. and D. Norton. 2001. Chemical analysis and toxicity testing of Spokane River sediments collected in October 2000. Publication number 01-03-019. Environmental Assessment Program. Washington State Department of Ecology. Olympia, Washington.

Kemble, N.E, D.K. Hardesty, C.G. Ingersoll, B.T. Johnson, F.J. Dwyer, and D.D. MacDonald. 2000. An evaluation of the toxicity of contaminated sediments from Waukegan Habor, Illinois, following remediation. Archives of Environmental Contamination and Toxicology 39:452-461.

Landis, W.G., A.J. Markiewicz, V. Wilson, A. Fairbrother, G. Mann. 1997. Recommended guidance and checklist for Tier 1 ecological risk assessment of contaminated sites in British Columbia: Review Draft. Pollution Prevention and Remediation Branch. Environmental and Resource Management Branch. Ministry of Environment Land and Parks. Victoria, British Columbia.

Lasier, P.J., P.V. Winger, J.L. Shelton Jr., and K.J., Bogenrieder. 2001. Contaminant impacts to early life stages of the robust redhorse (*Moxostoma robustum*) in the Lower Oconee River. United States Geological Survey. Patuxent Wildlife Research Center. Laurel, Maryland. Submitted to: Species at Risk Program. Biological Resources Division. United States Geological Survey.

Long, E.R. and L.G. Morgan. 1991. The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 52. National Oceanic and Atmospheric Administration. Seattle, Washington. pp. 175 + appendices.

Long, E.R. and D.D. MacDonald. 1998. Recommended uses of empirically derived, sediment quality guidelines for marine and estuarine ecosystems. Human and Ecological Risk Assessment 4:1019-1039.

Long, E.R., M.F. Buchman, S.M. Bay, R.J. Breteler, R.S. Carr, P.M. Chapman, J.E. Hose, A.L. Lissner, J. Scott, and D.A. Wolfe. 1990. Comparative evaluation of five toxicity tests with sediments from San Francisco Bay and Tomales Bay, California. Environmental Toxicology and Chemistry 9:1193-1214.

Long, E.R., D.A. Wolfe, R.S. Carr, K.J. Scott, G.B. Thursby, H.L. Windom, R. Lee, F.D. Calder, G.M. Sloane, and T. Seal. 1994. Magnitude and extent of sediment toxicity in Tampa Bay, Florida. NOAA Technical Memorandum NOS ORCA 78. National Oceanic and Atmospheric Administration. Silver Spring, Maryland.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. 1995a. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environmental Management 19:81-97.

Long, E.R., D.A. Wolfe, K.J. Scott, G.B. Thursby, E.A. Stern, C. Peven, and T. Schwartz. 1995b. Magnitude and extent of sediment toxicity in the Hudson-Raritan Estuary. NOAA Technical Memorandum NOS ORCA 88. National Oceanic and Atmospheric Administration. Silver Spring, Maryland.

Long, E.R., G.M. Sloane, R.S. Carr, K.J. Scott, G.B. Thursby, and T.L. Wade. 1996. Sediment toxicity in Boston Harbor: Magnitude, extent, and relationships with chemical toxicants. NOAA Technical Memorandum NOS ORCA 96. National Oceanic and Atmospheric Administration. Silver Spring, Maryland.

Long, E.R., G.M. Sloane, S.R. Carr, S.R., T. Johnson, J. Biedenbach, J., J.K. Scott, G.B. Thursby, E. Crecelius, C. Peven, H.L. Windom, R.D. Smith Loganathon B. 1997. Magnitude and extent of sediment toxicity in four bays of the Florida Panhandle: Pensacola, Choctawatchee, St. Andrew and Apalachicola. United States Department of Commerce. National Oceanic and Atmospheric Administration. Silver Spring, Maryland.

Long, E.R., D.D. MacDonald, J.C. Cubbage, and C.G. Ingersoll. 1998a. Predicting the toxicity of sediment-associated trace metals with SEM-AVS concentrations and dry weight-normalized concentrations - A critical comparison. Environmental Toxicology and Chemistry 17:972-974.

Long, E.R., L.J. Field, and D.D. MacDonald. 1998b. Predicting toxicity in marine sediments with numerical sediment quality guidelines. Environmental Toxicology and Chemistry 17:714-727.

Long, E., G. Scott, J. Kucklick, M. Fulton, B. Thompson, R.S. Carr, J. Biedenbach, K.J. Scott, G. Thursby, G. Chandler, J. Anderson, G. Sloane. 1998c. Magnitude and extent of sediment toxicity in selected estuaries of South Carolina and Georgia. NOAA Technical Memorandum NOS ORCA 128. National Oceanic and Atmospheric Administration. Silver Spring, Maryland

Long, E., G. Sloane, G. Scott, B. Thompson, R. Carr, J. Biendenbach, T. Wade, R. Presley, K. Scott, C. Mueller, G. Brecken-Fols, B. Albrecht, J. Anderson and G. Chandler. 1999. Magnitude and extent of chemical contamination and toxicity in sediments of Biscayne Bay and vicinity - final draft. NOAA Technical Memorandum NOS NCCOS CCMA 141. National Status and Trends Program. National Oceanic and Atmospheric Administration. Silver Spring, Maryland.

Loring, D.H. 1991. Normalization of heavy-metal data from estuarine and coastal sediments. ICES Journal of Marine Science 48:101-115.

MacDonald, D.D. 1997. Sediment injury in the Southern California Bight: Review of the toxic effects of DDTs and PCBs in sediments. Prepared for National Oceanic and Atmospheric Administration. United States Department of Commerce. Long Beach, California.

MacDonald, D.D., S.L. Smith, M.P. Wong, and P. Mudroch. 1992. The development of Canadian marine environmental quality guidelines. Report prepared for the Canadian Council of Resource and Environment Ministers. Ottawa, Canada.

MacDonald, D.D., R.S. Carr, F.D. Calder, E.R. Long, and C.G. Ingersoll. 1996. Development and evaluation of sediment quality guidelines for Florida coastal waters. Ecotoxicology 5:253-278.

MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000a. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Archives of Environmental Contamination and Toxicology 39:20-31.

MacDonald, D.D., L.M. DiPinto, J. Field, C.G. Ingersoll, E.R. Long, and R.C. Swartz. 2000b. Development and evaluation of consensus-based sediment effect concentrations for polychlorinated biphenyls (PCBs). Environmental Toxicology and Chemistry 19:1403-1413.

MacDonald, D.D., C.G. Ingersoll, D.R.J. Moore, M. Bonnell, R.L. Breton, R.A. Lindskoog, D.B. MacDonald, Y.K. Muirhead, A.V. Pawlitz, D.E. Sims, D.E. Smorong, R.S. Teed, R.P. Thompson, and N. Wang. 2002. Calcasieu Estuary remedial investigation/feasability study (RI/FS): Baseline ecological risk assessment (BERA). Technical report plus appendices. Contract No. 68-W5-0022. Prepared for CDM Federal Programs Corporation and United States Environmental Protection Agency. Dallas, Texas.

McGee, B.L., C.E. Schlekat, D.M. Boward, and T.L. Wade. 1995. Sediment contamination and biological effects in a Chesapeake Bay marina. Ecotoxicology 4:39-59

McLeay, D., S. Yee, K. Doe, and S. Wade. 1991. Final Report: Phase II and phase III studies by EC Laboratories of 10-day tests for sediment toxicity using marine or estuarine infaunal amphipods. McLeay Associates Ltd. West Vancouver, British Columbia. Prepared for Environment Canada.

Neff, J.M., D.J. Bean, B.W. Cornaby, R.M. Vaga, T.C. Gulbransen, and J.A. Scanlon. 1986. Sediment quality criteria methodology validation: Calculation of screening level concentrations from field data. Prepared for Environmental Protection Agency Region V. Washington, District of Columbia. 225 pp.

Nelson, W.G., B.J. Bergen, S.J. Benyi, G.Morrison, R.A. Voyer, C.J. Strobel, S. Rego, G. Thursby, and C.E. Pesch. 1996. New Bedford Harbor Long - Term Monitoring Assessment Report: Baseline Sampling. United States Environmental Protection Agency. Research Report, October 1996. National Health and Environmental Effects Research Laboratory. Atlantic Ecology Division. Narragansett, Rhode Island.

NYSDEC (New York State Department of Environmental Conservation). 1999. Technical guidance for screening contaminated sediments. Division of Fish, Wildlife and Marine Resources. Albany, New York. 39 pp.

Pastorok, R.A. and D.S. Becker. 1990. Comparative sensitivity of sediment toxicity bioassays at three Superfund sites in Puget Sound. American Society for Testing and Materials ASTM STP 1096. pp 123-139. PTI Environmental Services. Bellevue, Washington.

Persaud, D., R. Jaagumagi, and A. Hayton. 1993. Guidelines for the protection and management of aquatic sediment quality in Ontario. Standards Development Branch. Ontario Ministry of Environment and Energy. Toronto, Ontario. 27 pp.

Rice, C.A., P.D. Plesha, E. Casillas, D.A. Misitano, and J.P. Meador. 1995. Growth and survival of three marine invertebrate species in sediments from the Hudson-Raritan Estuary, New York. Environmental Toxicology and Chemistry 14(11):1931-1940.

SAIC (Science Application International Corporation). 1991. Draft compilation of sediment quality guidelines for EPA Region 5 inventory of contaminated sediment sites. Prepared by Science Application International Corporation. Chicago, Illinois. 48 pp.

Sapudar, R.A., C.J. Wilson, M.L. Reid, E.R. Long, M. Stephenson, M. Puckett, R. Fairey, J. Hunt, B. Anderson, D. Holstad, J. Newman, S. Birosik, and H. Smythe. 1994. Sediment chemistry and toxicity in the vicinity of the Los Angeles and Long Beach Harbors. Draft Final Report. Prepared jointly by California State Water Resources Control Board, National Oceanic and Atmospheric Administration, and California Department of Fish and Game.

Schropp, S.J., F.G. Lewis, H.L. Windom, J.D. Ryan, F.D. Calder, and L.C. Burney. 1990. Interpretation of metal concentrations in estuarine sediments of Florida using aluminium as a reference element. Estuaries 13(3): 227-235.

Smith, S.L., D.D. MacDonald, K.A. Keenleyside, C.G. Ingersoll, and L.J. Field. 1996. A preliminary evaluation of sediment quality assessment values for freshwater ecosystems. Journal of Great Lakes Research 22(3):624-638.

Striplin, P., P. Sparks-McConkey, D. Davis, F. Svendsen. 1991. Puget Sound ambient monitoring program: Marine sediment monitoring task. Appendices. Washington State Department of Ecology. Environmental Investigations and Laboratory Services Program. Olympia, Washington

Striplin, P., P. Sparks-McConkey, D. Davis, F. Svendsen. 1992. Puget Sound Ambient Monitoring Program, Marine Sediment monitoring task, Annual Report 1990. Washington State Department of Ecology. Environmental Investigations and Laboratory Services Program. Olympia, Washington

Swartz, R.C. 1999. Consensus sediment quality guidelines for PAH mixtures. Environmental Toxicology and Chemistry 18:780-787.

Swartz, R.C., D.W. Schults, G.R. Ditsworth, W.A. DeBen, F.A. Cole. 1985. Sediment toxicity, contamination, and macrobenthic communities near a large sewage outfall. Validation and Predictability of Laboratory Methods for Assessing the Fate and Effects of Contaminants in Aquatic Ecosystems, ASTM STP 865. American Society for Testing and Materials. Philadelphia, Pennsylvania.

Swartz, R.C., F.A. Cole, D.W. Schults and W.A. DeBen. 1986. Ecological changes in the Southern California Bight near a large sewage outfall: benthic conditions in 1980 and 1983. Marine Ecology Progress Series. United States Environmental Protection Agency. Newport, Oregon.

Swartz, R.C., D.W. Schults, J.O. Lamberson, R.J. Ozretich, and J.K. Stull. 1991. Vertical profiles of toxicity, organic carbon, and chemical contaminants in sediment cores from the Palos Verdes shelf and Santa Monica Bay, California. Marine and Environmental Research 31(3):215-225.

Tay, Kok-Leng, Ken G. Doe, Suzanne J. Wade, David A. Vaughan, Rita E. Berrigan and Michael J. Moore. 1990. Sediment assessment in Halifax Harbour. *In:* Aquatic Toxicity Workshop Proceedings. Environment Canada. Dartmouth, Nova Scotia.

Tetra Tech, Inc. 1986. Development of sediment quality values for Puget Sound. Volume 1. Puget Sound Dredged Disposal Analysis Report. Seattle, Washington. 129 pp.

Tetra Tech, Inc. 1990. Puget Sound Ambient Monitoring Program 1989: Marine Sediment Monitoring - Final Report and Appendices. Tetra Tech, Inc. 262. Washington Department of Ecology, Ambient Monitoring Section. Bellevue, Washington.

Thursby, G.B., Heltshe, J., and K.J. Scott. 1997. Revised approach to toxicity test acceptability criteria using a statistical performance assessment. Environmental Toxicology and Chemistry 16:1322-1329.

USEPA (United States Environmental Protection Agency). 1989. Assessing human health risks from chemically contaminated fish and shellfish: A guidance manual. EPA-503/8-89-002. Office of Water Regulations and Standards. Washington, District of Columbia. 91 pp + app.

USEPA (United States Environmental Protection Agency). 1992. Sediment classification methods compendium. EPA 813/R-92/006. Washington, District of Columbia.

USEPA (United States Environmental Protection Agency). 1996. Assessment and Remediation of Contaminated Sediments (ARCS) Program: Assessment of Sediment in the Indiana Harbor Area of Concern. EPA 905-R96-009. Great Lakes National Program Office. Chicago, Illinois.

USEPA (United States Environmental Protection Agency). 1997a. The incidence and severity of sediment contamination in surface waters of the United States: Volume 1: National Sediment Quality Survey. EPA/823/R-97/006. Washington, District of Columbia.

USEPA (United States Environmental Protection Agency). 1997b. An assessment of sediments from the Upper Mississippi River, Final Report. EPA 823-R-97-005. United States Department of the Interior. Office of Water.

USEPA (United States Environmental Protection Agency). 1998. EPA's sediment management strategy. EPA/823/R-98/001. Office of Water. Washington, District of Columbia

USEPA (United States Environmental Protection Agency). 2000. Prediction of sediment toxicity using consensus-based freshwater sediment quality guidelines. EPA 905/R-00/007. Great Lakes National Program Office. Chicago, Illinois.

USEPA (United States Environmental Protection Agency). 2001. Methods for collection, storage and manipulation of sediments for chemical and toxicological analyses: Technical manual. EPA-823-B-01-002. Office of Water. Washington, District of Columbia.

Velinsky, D.J., T.J. Wade, C.E. Schlekat; B.L. McGee, B.J. Presley; Wade, T.L., D.J. Velinsky, E. Reinharz, and C.E. Schlekat; C.E. Schlekat, B.L. McGee, D.M. Boward, E. Reinharz, D.J. Velinsky, and T.L. Wade. 1994. Tidal River Sediments in the Washington, D.C. Area. - I, II, III. Estuarine Research Federation. Estuaries 17(2):305-344.

Ward, J.A., M.R. Pinza, M.E. Barrows, and J.Q. Word. 1992. Ecological evaluation of proposed dredged material from Wilmington Harbor, North Carolina. Draft report. Prepared for the United States Army Corps of Engineers with the United States Department of Energy. Contract DE-AC06-76RLO 1830. Pacific Northwest Laboratory. Richland, Washington.

WDOE (Washington State Department of Ecology). 1990. Sediment management standards: Chapter 173-204 WAC. Olympia, Washington. 106 pp.

WDOE (Washington State Department of Ecology). 1994. Puget Sound Ambient Monitoring Program Marine Sediment Monitoring Program. Publication #94-93. Environmental Investigations and Laboratory Services Program. Ambient Monitoring Section. Washington State Department of Ecology. Olympia, Washington

WDOH (Washington State Department of Health). 1995. Development of sediment quality criteria for the protection of human health. Tier I report. Environmental Health Programs. Office of Toxic Substances. Olympia, Washington.

WDOH (Washington State Department of Health). 1996. Development of sediment quality criteria for the protection of human health. Tier II report Environmental Health Programs. Office of Toxic Substances. Olympia, Washington.

Windom, H. 1995. An evaluation of ambient sediment quality in the Brunswick Harbor and Savannah River entrance channels, Georgia. Chemical data. Excerpts from an unpublished report. Skidaway Institute of Oceanography. Savannah, Georgia.

Winger, P.V., P.J. Lasier, and K.J. Bogenrieder. 2000. Bioassessment of Hollis Creek, Oktibbeha County, Mississippi. United States Geological Survey. Patuxent Wildlife Research Center. Prepared for United States Fish and Wildlife Service.

Wolfe, D.A., S.B. Bricker, E.R. Long, K.J. Scott and G. B. Thursby. 1994. Biological Effects of Toxic Contaminants in Sediments from Long Island Sound and Environs. NOAA Technical Memorandum NOS ORCA 80. National Oceanic and Atmospheric Administration. Silver Spring, Maryland.

Tables

Table 1. Summary of the strengths and limitations of existing approaches for deriving numerical sediment quality assessment guidelines (adapted from Crane *et al.* 2000).

Approach	Strengths	Limitations
Screening Level Concentration Approach	* Based on biological effects data. * Sufficient data to derive SQGs are generally available for many chemicals. * Suitable for all classes of chemicals and most types of sediments. * Accounts for the effects of mixtures of contaminants.	* Not possible to establish cause and effect relationships. * Large database of matching sediment chemistry and benthic data is required. * Chemistry and benthic data are rarely strictly matching (i.e., generated from splits of a homogenized sediment sample). * Bioavailability is not considered.
Effects Range Approach	* Based on biological effects data. * Many types of biological effects data are considered. * Suitable for all classes of chemicals and most types of sediments. * Provides a weight of evidence. * Provides data summaries for evaluating sediment quality. * Accounts for the effects of mixtures of contaminants.	* Large database of matching sediment chemistry and biological effects data is required. * Not possible to establish cause and effect relationships. * Bioavailability is not considered. * Does not consider the potential for bioaccumulation.
Effects Level Approach	* Based on biological effects data. * Many types of biological effects data are considered. * Suitable for all classes of chemicals and most types of sediments. * Provides a weight of evidence. * Provides data summaries for evaluating sediment quality. * Accounts for the effects of mixtures of contaminants.	* Large database of matching sediment chemistry and biological effects data is required. * Not possible to establish cause and effect relationships. * Bioavailability is not considered. * Does not consider the potential for bioaccumulation.

Table 1. Summary of the strengths and limitations of existing approaches for deriving numerical sediment quality assessment guidelines (adapted from Crane *et al.* 2000).

Approach	Strengths	Limitations
Apparent Effects Threshold Approach	* Based on biological effects data. * Several types of biological effects data are considered. * Considers effects on benthic invertebrate community structure. * Suitable for all classes of chemicals and most types of sediments. * Accounts for the effects of mixtures of contaminants.	* Extensive site-specific database is required. * Not possible to establish cause and effect relationships. * Risk of under-protection of resource. * Bioavailability is not considered. * Does not consider the potential for bioaccumulation.
Equilibrium Partitioning Approach	* Based on biological effects. * Suitable for many classes of chemicals and most types of sediments. * Bioavailability is considered. * Supports cause and effect evaluations.	* Water quality criteria are not available for certain substances. * *In situ* sediments are rarely at equilibrium. * Further field validation is needed. * Guidelines for single chemicals do not account for effects. of mixtures of contaminants. * Risk of under-protection of resource. * Does not consider the potential for bioaccumulation.
Logistic Regression Modelling Approach	* Based on sediment toxicity test results. * Suitable for all classes of chemicals and most types of sediments. * Accounts for the effects of mixtures of contaminants. * Provides SQGs that are associated with a specific probability of observing sediment toxicity. * Provides SQGs that are species and endpoint specific. * Factors that influence bioavailability can be considered. * SQGs can be derived that correspond to specific management goals (e.g., 20% probability of observing sediment toxicity).	* Large database of matching sediment chemistry and biological effects data is required. * Insufficient data are available for most freshwater receptors. * Not possible to establish cause and effect relationships. * Bioavailability is not considered. * Does not consider the potential for bioaccumulation.

Table 1. Summary of the strengths and limitations of existing approaches for deriving numerical sediment quality assessment guidelines (adapted from Crane *et al.* 2000).

Approach	Strengths	Limitations
Consensus-Based Sediment Quality Guidelines Approach	* Provides a unifying synthesis of the existing sediment quality guidelines. * Reflects causal rather than correlative effects. * Accounts for the effects of contaminant mixtures in sediments. * Predictive ability in freshwater sediments has been demonstrated.	* Bioavailability is not considered. * Does not consider the potential for bioaccumulation.
Tissue Residue Approach	* Bioaccumulation is considered. * A protocol for the derivation of tissue residue guidelines is available. * Numerical SQGs can be derived if biota-sediment accumulation factors are available.	* Tissue residue guidelines for wildlife are not yet available for most chemicals. * Wildlife may be exposed to contaminants from multiple sites.

SQGs = sediment quality guidelines.

Table 2. Factors for consideration in the application of the criteria for sensitive contaminated sites ($SedQC_{SCS}$).

Areas to Which the $SedQC_{SCS}$ Should be Applied Include:

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* Areas, sites or reaches which support red and blue listed plants and animal species, or nests designated under the *Wildlife Act.*

* Habitats used by endangered or threatened species, or Species of Special Concern under the *Species at Risk Act.*

* Watercourses, wetlands, forested riparian areas, mudflats and intertidal zones that are important to preservation of fish and wildlife.

* Reaches of aquatic habitats that are important to fish spawning or serve as important rearing habitat for fish.

* Reaches of aquatic environments encompassing, and/or bordering habitat compensation or restoration sites, or other areas that are intended or designed to create, restore or enhance biological or habitat features.

* Areas of unique habitat that are identified in provincial or municipal landuse plans.

* Reaches of the aquatic environment that exist within provincial marine parks, provincial parks, or ecological reserves.

* Areas and aquatic habitat included within provincial Wildlife Management Areas.

* Areas covered under conservation agreements and areas designated as "Environmentally Sensitive" in municipal landuse plans or strategies.

Marinas, docks, wharves and associated infrastructure located within these areas may be assessed making use of the $SedQC_{TCS}$ criteria limits. To make use of the $SedQC_{TCS}$ in these circumstances, the proponents must present information to support their proposal to the appropriate agencies. This information should include:

* The identification of existing resources in the area;

* The identification of offsite contaminant sources; and,

* The measures taken to eliminate on-site sources of contamination.

$SedQC_{TCS}$ = sediment quality criteria for typical sites.

Table 3. Preliminary benchmarks for sediment chemistry (CCME 1999).

Chemicals of Potential Concern (COPCs)	Marine and Estuarine Sediments	Freshwater Sediments
Metals (µg/kg DW)		
Arsenic	41 600	17 000
Cadmium	4 200	3 500
Chromium	160 000	90 000
Copper	108 000	197 000
Lead	112 000	91 300
Mercury	700	486
Zinc	271 000	315 000
Polycyclic Aromatic Hydrocarbons (PAHs; µg/kg DW)		
2-Methylnaphthalene	201	201
Acenaphthene	88.9	88.9
Acenaphthylene	128	128
Anthracene	245	245
Fluorene	144	144
Naphthalene	391	391
Phenanthrene	544	515
Benz(a)anthracene	693	385
Benzo(a)pyrene	763	782
Chrysene	846	862
Dibenz(a,h)anthracene	135	135
Fluoranthene	1 494	2 355
Pyrene	1 398	875
Total PAHs	16 770	16 770
Polychlorinated Biphenyls (PCBs; µg/kg DW)		
Aroclor 1254	709	340
Total PCBs	189	277
Organochlorine Pesticides (µg/kg DW)		
Chlordane	4.79	8.87
Dieldrin	4.3	6.67
Endrin	62.4	62.4
Heptachlor	2.74	2.74
Heptachlor epoxide	2.74	2.74
Lindane	0.99	1.38
Sum DDD	7.81	8.51
Sum DDE	374	6.75
Sum DDT	4.77	4.77
PCDD/PCDFs (µg TEQ/kg)		
2,3,7,8-TCDD TEQ	0.215	0.215

DW = dry weight; PCDDs = polychlorinated dibenzo-*p*- dioxins; PCDFs = polychlorinated dibenzofurans; TEQ = toxic equivalents; TCDD = tetrachlorodibenzo-*p*- dioxin.

Table 4. Sediment quality criteria for assessing and managing contaminated sediments at sensitive sites ($SedQC_{SCS}$)[1].

Chemicals of Potential Concern (COPCs)	Marine and Estuarine Sediments	Freshwater Sediments
Metals (µg/kg DW)		
Arsenic	26 000	11 000
Cadmium	2 600	2 200
Chromium	99 000	56 000
Copper	67 000	120 000
Lead	69 000	57 000
Mercury	430	300
Zinc	170 000	200 000
Polycyclic Aromatic Hydrocarbons (PAHs; µg/kg DW)		
2-Methylnaphthalene	120	120
Acenaphthene	55	55
Acenaphthylene	79	80
Anthracene	150	150
Fluorene	89	89
Naphthalene	240	240
Phenanthrene	340	320
Benz(a)anthracene	430	240
Benzo(a)pyrene	470	480
Chrysene	520	530
Dibenz(a,h)anthracene	84	84
Fluoranthene	930	1 500
Pyrene	870	540
Total PAHs	10 000	10 000
Polychlorinated Biphenyls (PCBs; µg/kg DW)		
Aroclor 1254	440	210
Total PCBs	120	170
Organochlorine Pesticides (µg/kg DW)		
Chlordane	3.0	5.5
Dieldrin	2.7	4.1
Endrin	39	39
Heptachlor	1.7	1.7
Heptachlor epoxide	1.7	1.7
Lindane	0.61	0.86
Sum DDD	4.8	5.3
Sum DDE	230	4.2
Sum DDT	3.0	3.0
PCDD/PCDFs (µg TEQ/kg)		
2,3,7,8-TCDD TEQ	0.13	0.13

[1] The SedQC for sites with sensitive habitats were established by multiplying the PEL (CCME 1999) by 0.62.

DW = dry weight; PCDDs = polychlorinated dibenzo-*p*- dioxins; PCDFs = polychlorinated dibenzofurans; TEQ = toxic equivalents; TCDD = tetrachlorodibenzo-*p*- dioxin; PEL = probable effect level.

Table 5. Sediment quality criteria for assessing and managing contaminated sediments at typical sites ($SedQC_{TCS}$)[1].

Chemicals of Potential Concern (COPCs)	Marine and Estuarine Sediments	Freshwater Sediments
Metals (µg/kg DW)		
Arsenic	50 000	20 000
Cadmium	5 000	4 200
Chromium	190 000	110 000
Copper	130 000	240 000
Lead	130 000	110 000
Mercury	840	580
Zinc	330 000	380 000
Polycyclic Aromatic Hydrocarbons (PAHs; µg/kg DW)		
2-Methylnaphthalene	240	240
Acenaphthene	110	110
Acenaphthylene	150	150
Anthracene	290	290
Fluorene	170	170
Naphthalene	470	470
Phenanthrene	650	620
Benz(a)anthracene	830	460
Benzo(a)pyrene	920	940
Chrysene	1 000	1 000
Dibenz(a,h)anthracene	160	160
Fluoranthene	1 800	2 800
Pyrene	1 700	1 100
Total PAHs	20 000	20 000
Polychlorinated Biphenyls (PCBs; µg/kg DW)		
Aroclor 1254	850	410
Total PCBs	230	330
Organochlorine Pesticides (µg/kg DW)		
Chlordane	5.7	11
Dieldrin	5.2	8.0
Endrin	75	75
Heptachlor	3.3	3.3
Heptachlor epoxide	3.3	3.3
Lindane	1.2	1.7
Sum DDD	9.4	10
Sum DDE	450	8.1
Sum DDT	5.7	5.7
PCDD/PCDFs (µg TEQ/kg)		
2,3,7,8-TCDD TEQ	0.26	0.26

[1] The SedQC for typical sites were established at 1.2 times the PEL (CCME 1999).

DW = dry weight; PCDDs = polychlorinated dibenzo-*p*- dioxins; PCDFs = polychlorinated dibenzofurans; TEQ = toxic equivalents; TCDD = tetrachlorodibenzo-*p*- dioxin; PEL = probable effect level.

Table 6. Listing of matching sediment chemistry and toxicity data sets compiled in the national freshwater database used to assess the reliability of the sediment quality criteria for sensitive ($SedQC_{SCS}$) and typical ($SedQC_{TCS}$) contaminated sediments.

Location	Sample Date	n	Number of Toxic Samples (%)[1]	Reference
Upper Mississippi River, MN	1987[2]	4	0 (0%)	Ingersoll *et al.* (1996)
Waukegan Harbor Area of Concern, IL	1987[2]	4	2 (50%)	Ingersoll *et al.* (1996)
Trinity River, TX	1988[2]	5	0 (0%)	Ingersoll *et al.* (1996)
Mobile Bay, AL	1988[2]	5	0 (0%)	Ingersoll *et al.* (1996)
Buffalo River (NY) and Saginaw River (MI) Areas of Concern	1989-90	18	7 (39%)	Ingersoll *et al.* (1993a)
Indiana Harbour Area of Concern, IN	1989	4	4 (100%)	USEPA (1996)
Tabbs Bay, TX	1990[2]	5	3 (60%)	Ingersoll *et al.* (1996)
Anacostia River, Kingman Lake, and Potomac River, DC	1991	14	5 (36%)	Velinsky *et al.* (1994)
Upper Clark Fork River, MT	1991[2]	15	8 (53%)	Ingersoll *et al.* (1993b)
Bohemia River, MD	1991	10	3 (30%)	McGee *et al.* (1995)
Columbia River Basin, WA	2000	8	8 (100%)	Johnson and Norton (2001)
Waukegan Harbor Area of Concern, IL	1996	20	20 (100%)	Kemble *et al.* (2000)
Upper Mississippi River and St. Croix River (MN, WI, IL, IA, MO)	1994	49	2 (4%)	USEPA (1997b)

Table 6. Listing of matching sediment chemistry and toxicity data sets compiled in the national freshwater database used to assess the reliability of the sediment quality criteria for sensitive ($SedQC_{SCS}$) and typical ($SedQC_{TCS}$) contaminated sediments.

Location	Sample Date	n	Number of Toxic Samples (%)[1]	Reference
Canal Creek, MD; Eliza Pool and Rio Grande River, TX; Kennebec River, ME	1995-96[2]	17	4 (24%)	Ingersoll *et al.* (1998)
Oconee River, GA	1998	12	6 (50%)	Lasier *et al.* (2001)
Hollis Creek, MS	1999	5	3 (60%)	Winger *et al.* (2000)
Barton Creek and Wells Branch Creek, TX	2000	9	0 (0%)	Ingersoll *et al.* (2001)
Calcasieu River, LA	2000	99	23 (23%)	MacDonald *et al.* (2002)
Overall		***303***	***98 (32%)***	

[1]Toxicity to the marine amphipods, *Hyalella azteca,* in 28-42 day toxicity tests (endpoint: survival or growth). Individual samples were designated as toxic based on a statistically significant difference from the control or reference.

[2]Sampling date unknown, date of sample analysis used.

$SedQC_{TCS}$ = sediment quality criteria for typical sites; $SedQC_{SCS}$ = sediment quality criteria for sensitive sites.
AL = Alabama; DC = District of Columbia; GA = Georgia; IA = Iowa; IL = Illinois; IN = Indiana; LA = Louisiana; MD = Maryland; ME = Maine; MI = Michigan
MN = Minnesota; MO = Missouri; MS = Mississippi; MT = Montana; NY = New York; TX = Texas; WA = Washington; WI = Wisconsin.

Table 7. Listing of matching sediment chemistry and toxicity data sets compiled in the national marine and estuarine database used to assess the reliability of the sediment quality criteria for sensitive ($SedQC_{SCS}$) and typical ($SedQC_{TCS}$) contaminated sediments.

Location	Sampling Date	n	Number of Toxic Samples (%)[1]	Reference
Palos Verdes and Santa Monica Bay, CA	1980	7	3 (43%)	Swartz *et al.* (1985)
Palos Verdes and Santa Monica Bay, CA	1980, 1983	5	0 (0%)	Swartz *et al.* (1986)
Palos Verdes and Santa Monica Bay, CA	1985	31	17 (55%)	Swartz *et al.* (1991)
Everett, WA	1986	6	0 (0%)	Hart Crowser Inc. (1986)
Palos Verdes, CA	1986	9	0 (0%)	Ferraro *et al.* (1991)
Gulf of Mexico, TX	1987	10	3 (30%)	Chapman *et al.* (1991)
San Francisco Bay, CA	1987	15	12 (80%)	Long *et al.* (1990)
San Francisco Bay, CA	1987[2]	9	2 (22%)	Chapman *et al.* (1987)
Houston Ship Channel/San Jacinto River, TX	1988-1989	18	2 (11%)	Crocker *et al.* (1991)
Howe Sound, BC	1989	9	1 (11%)	McLeay *et al.* (1991)
Puget Sound, WA	1989	48	21 (44%)	Tetra Tech, Inc. (1990)
Halifax Harbour, NS	1990	12	6 (50%)	Tay *et al.* (1990)
Puget Sound, WA	1990	64	55 (86%)	Striplin *et al.* (1991; 1992)
Puget Sound, WA	1990[2]	18	6 (33%)	Pastorok *et al.* (1990)
San Francisco Estuary, CA	1991-1992	7	2 (29%)	Flegal *et al.* (1996)
Tampa Bay, FL	1991-1992	96	9 (9%)	Long *et al.* (1994)

Table 7. Listing of matching sediment chemistry and toxicity data sets compiled in the national marine and estuarine database used to assess the reliability of the sediment quality criteria for sensitive ($SedQC_{SCS}$) and typical ($SedQC_{TCS}$) contaminated sediments.

Location	Sampling Date	n	Number of Toxic Samples (%)[1]	Reference
EMAP Louisianian Province (LA, MS, FL, TX)	1991-1993	352	131 (37%)	Engle and Harwell (1996)
Hudson-Raritan, NY/NJ	1991, 1993	48	33 (69%)	Long *et al.* (1995b)
Hudson-Raritan Estuary, NY	1991	10	2 (20%)	Rice *et al.* (1995)
Long Island Sound, NY/CT	1991	61	48 (79%)	Wolfe *et al.* (1994)
Puget Sound, WA	1991	62	15 (24%)	WDOE (1994)
Central and North Coast, Los Angeles, Santa Ana, San Diego, San Fransico Bay, CA	1992-1997	448	312 (68%)	Sapudar *et al.* (1994); Fairey *et al.* (1996); Anderson *et al.* (1997); Hunt *et al.* (1998); Downing *et al.* (1998); Jacobi *et al.* (1998)
Brunswick Harbor Entrance, GA	1992	9	0 (0%)	Windom (1995)
Palos Verdes, CA	1992	5	0 (0%)	Bay *et al.* (1994)
Savannah River Entrance, GA	1992	8	0 (0%)	Windom (1995)
Wilmington Harbor, NC	1992	5	0 (0%)	Ward *et al.* (1992)
Newark Bay Watershed, NY/NJ	1993-1994	168	54 (32%)	Adams *et al.* (1998)
South Carolina (SC) and Georgia (GA)	1993-1994	158	7 (4%)	Long *et al.* (1998c)
Western Florida, FL	1993-1994	62	2 (3%)	Long *et al.* (1997)
Boston Harbor, MA	1993	29	9 (31%)	Long *et al.* (1996)

Table 7. Listing of matching sediment chemistry and toxicity data sets compiled in the national marine and estuarine database used to assess the reliability of the sediment quality criteria for sensitive ($SedQC_{SCS}$) and typical ($SedQC_{TCS}$) contaminated sediments.

Location	Sampling Date	n	Number of Toxic Samples (%)[1]	Reference
New Bedford Harbor, MA	1993	70	45 (64%)	Nelson *et al.* (1996)
EMAP Carolinian Province (GA, VA, NC, SC,	1994-1995	185	22 (12%)	Hyland *et al.* (1996); Hyland *et al.* (1998)
Bayou Casotte Turning Basin, MS	1994	4	0 (0%)	EA Engineering, Science, and Technology Inc. (1994)
Biscayne Bay, FL	1995-1996	214	36 (17%)	Long *et al.* (1999)
Overall		***2262***	***855 (38%)***	

[1]Toxicity to the freshwater amphipods, *Ampelisca abdita* and *Rhepoxynius abronius*, in 10 day toxicity tests (endpoint: survival). Individual samples were designated as toxic based on a statistically significant difference from the control or reference.

[2]If the sample date was unknown, the publishing date of the report was used.

BC = British Columbia; CT = Connecticut; FL = Florida; GA = Georgia; LA = Louisiana; MA = Massachusetts; MS = Mississippi; NC = North Carolina; NJ = New Jersey; NS = Nova Scotia; NY = New York; SC = South Carolina; TX = Texas; VA = Virginia; WA = Washington.
SedQC = Sediment Quality Criteria; TCS typical contaminated sties; SCS = sensitive contaminated sites; n = number of samples; NOAA = National Oceanic and Atmospheric Administration; NS&T = National Status and Trends; WDOE = Washington Department of Ecology; EMAP = Environmental Monitoring and Assessment Program.

Table 8. Incidence of toxicity within ranges of contaminant concentrations defined by the freshwater sediment quality criteria ($SedQC_{SCS}$ and $SedQC_{TCS}$; based on CCME 1999), based on the results of 28 to 42-day amphipod toxicity tests[1] (survival or growth of *Hyalella azteca),* using the national database.

Chemicals of Potential Concern (COPCs)	Number of Samples Evaluated[2]	$<SedQC_{SCS}$	$\geq SedQC_{SCS}$ to $\leq SedQC_{TCS}$	$\geq SedQC_{SCS}$	$>SedQC_{TCS}$
Metals					
Arsenic	86	7 of 36 (19%)	10 of 15 (67%)	38 of 50 (76%)	28 of 35 (80%)
Cadmium	270	31 of 221 (14%)	8 of 13 (62%)	40 of 49 (82%)	32 of 36 (89%)
Chromium	117	23 of 72 (32%)	18 of 31 (58%)	29 of 45 (64%)	11 of 14 (79%)
Copper	272	49 of 239 (21%)	15 of 17 (88%)	27 of 33 (82%)	12 of 16 (75%)
Lead	273	26 of 203 (13%)	24 of 35 (69%)	48 of 70 (69%)	24 of 35 (69%)
Mercury	184	30 of 123 (24%)	11 of 27 (41%)	34 of 61 (56%)	23 of 34 (68%)
Zinc	282	30 of 202 (15%)	19 of 35 (54%)	47 of 80 (59%)	28 of 45 (62%)
Polycyclic Aromatic Hydrocarbons (PAHs)					
Low Molecular Weight (LMW) PAHs					
2-Methylnaphthalene	189	12 of 145 (8%)	2 of 8 (25%)	34 of 44 (77%)	32 of 36 (89%)
Acenaphthene	202	22 of 169 (13%)	6 of 9 (67%)	20 of 33 (61%)	14 of 24 (58%)
Acenaphthylene	230	33 of 191 (17%)	7 of 12 (58%)	20 of 39 (51%)	13 of 27 (48%)
Anthracene	246	30 of 204 (15%)	11 of 13 (85%)	29 of 42 (69%)	18 of 29 (62%)
Fluorene	235	31 of 197 (16%)	4 of 10 (40%)	22 of 38 (58%)	18 of 28 (64%)
Naphthalene	231	25 of 197 (13%)	10 of 11 (91%)	28 of 34 (82%)	18 of 23 (78%)
Phenanthrene	257	23 of 193 (12%)	14 of 21 (67%)	42 of 64 (66%)	28 of 43 (65%)
High Molecular Weight (HMW) PAHs					
Benz(a)anthracene	253	20 of 185 (11%)	13 of 23 (57%)	42 of 68 (62%)	29 of 45 (64%)
Benzo(a)pyrene	253	39 of 217 (18%)	5 of 7 (71%)	22 of 36 (61%)	17 of 29 (59%)
Chrysene	256	37 of 212 (17%)	7 of 12 (58%)	27 of 44 (61%)	20 of 32 (63%)
Dibenz(a,h)anthracene	224	32 of 190 (17%)	5 of 9 (56%)	16 of 34 (47%)	11 of 25 (44%)
Fluoranthene	256	49 of 230 (21%)	4 of 6 (67%)	15 of 26 (58%)	11 of 20 (55%)
Pyrene	259	28 of 192 (15%)	13 of 27 (48%)	39 of 67 (58%)	26 of 40 (65%)
Total PAHs[3]	267	46 of 230 (20%)	8 of 17 (47%)	20 of 37 (54%)	12 of 20 (60%)

Table 8. Incidence of toxicity within ranges of contaminant concentrations defined by the freshwater sediment quality criteria ($SedQC_{SCS}$ and $SedQC_{TCS}$; based on CCME 1999), based on the results of 28 to 42-day amphipod toxicity tests[1] (survival or growth of *Hyalella azteca),* using the national database.

Chemicals of Potential Concern (COPCs)	Number of Samples Evaluated[2]	<$SedQC_{SCS}$	≥$SedQC_{SCS}$ to ≤$SedQC_{TCS}$	≥$SedQC_{SCS}$	>$SedQC_{TCS}$
Polychlorinated Biphenyls (PCBs)					
Aroclor 1254	148	15 of 140 (11%)	2 of 4 (50%)	4 of 8 (50%)	2 of 4 (50%)
Total PCBs[4]	159	8 of 123 (7%)	2 of 10 (20%)	16 of 36 (44%)	14 of 26 (54%)
Organochlorine Pesticides					
Chlordane[5]	69	4 of 32 (13%)	1 of 1 (100%)	26 of 37 (70%)	25 of 36 (69%)
Dieldrin	59	3 of 33 (9%)	1 of 1 (100%)	24 of 26 (92%)	23 of 25 (92%)
Endrin	180	45 of 178 (25%)	2 of 2 (100%)	2 of 2 (100%)	ND
Heptachlor	29	3 of 27 (11%)	ND	2 of 2 (100%)	2 of 2 (100%)
Heptachlor epoxide	30	3 of 28 (11%)	ND	2 of 2 (100%)	2 of 2 (100%)
Lindane	46	21 of 45 (47%)	ND	1 of 1 (100%)	1 of 1 (100%)
Sum DDD[6]	58	3 of 33 (9%)	2 of 2 (100%)	22 of 25 (88%)	20 of 23 (87%)
Sum DDE[7]	60	2 of 32 (6%)	1 of 1 (100%)	25 of 28 (89%)	24 of 27 (89%)
Sum DDT[8]	54	4 of 34 (12%)	1 of 1 (100%)	19 of 20 (95%)	18 of 19 (95%)
PCDD/PCDFs					
2,3,7,8-TCDD TEQ[9]	13	1 of 5 (20%)	1 of 4 (25%)	4 of 8 (50%)	3 of 4 (75%)

ND = no data; $SedQC_{SCS}$ = sediment quality criteria for sensitive contaminated sites; $SedQC_{TCS}$ = Sediment quality criteria for typical contaminated sites; PCDDs = polychlorinated dibenzo-*p*- dioxins; PCDFs = polychlorinated dibenzofurans; TCDD TEQ = tetrachlorodibenzo-*p*- dioxin toxic equivalent; WHO = World Health Organization

[1]Individual samples were designated as toxic based on a statistically significant difference from the control or reference sample. If the measurement of the COPC is less than the detection limit, the value of 1/2 the detection limit was assigned.

[2]Excluding results for which the detection limit was greater than the probable effect level (PEL; CCME 1999), and results for which the COPC did not contribute substantially to the observed toxicity (see methods section for details).

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Table 8. Incidence of toxicity within ranges of contaminant concentrations defined by the freshwater sediment quality criteria ($SedQC_{SCS}$ and $SedQC_{TCS}$; based on CCME 1999), based on the results of 28 to 42-day amphipod toxicity tests[1] (survival or growth of *Hyalella azteca),* using the national database.

Chemicals of Potential Concern (COPCs)	Number of Samples Evaluated[2]	$<SedQC_{SCS}$	$\geq SedQC_{SCS}$ to $\leq SedQC_{TCS}$	$\geq SedQC_{SCS}$	$>SedQC_{TCS}$

[3]The concentrations of 2-methylnaphthalene, acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene and phenanthrene were summed to calculate LMW-PAHs. The concentrations of benz(a)anthracene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene, fluoranthene and pyrene were summed to calculate HMW-PAHs. The concentrations of LMW-PAHs and HMW-PAHs were summed to calculate Total PAHs.

[4]The concentrations of PCB Aroclors were summed to calculate Total PCBs.

[5]The concentrations of alpha- and gamma-chlordane, or cis-, and trans-chlordane were summed to calculate total chlordane.

[6]The concentrations of p,p'-DDD and o,p'-DDD were summed to calculate Sum DDD.

[7]The concentrations of p,p'-DDE and o,p'-DDE were summed to calculate Sum DDE.

[8]The concentrations of p,p'-DDT and o,p'-DDT were summed to calculate Sum DDT.

[9]Calculated using the WHO (van den Berg *et al.* 1998) toxic equivalency factors (TEFs) for fish, based on the concentrations of PCDDs, PCDFs, and co-planar PCB congeners.

Table 9. Reliability of the freshwater sediment quality criteria for assessing and managing sensitive ($SedQC_{SCS}$) and typical ($SedQC_{TCS}$) contaminated sediments.

Chemicals of Potential Concern (COPCs)	$SedQC_{SCS}$	Reliability	$SedQC_{TCS}$	Reliability
Metals (µg/kg DW)				
Arsenic	11 000	H	20 000	H
Cadmium	2 200	H	4 200	H
Chromium	56 000	L	110 000	H
Copper	120 000	M	240 000	H
Lead	57 000	H	110 000	H
Mercury	300	M	580	H
Zinc	200 000	H	380 000	H
Polycyclic Aromatic Hydrocarbons (PAHs; µg/kg DW)				
2-Methylnaphthalene	120	H	240	H
Acenaphthene	55	H	110	H
Acenaphthylene	80	H	150	M
Anthracene	150	H	290	H
Fluorene	89	H	170	H
Naphthalene	240	H	470	H
Phenanthrene	320	H	620	H
Benz(a)anthracene	240	H	460	H
Benzo(a)pyrene	480	H	940	H
Chrysene	530	H	1 000	H
Dibenz(a,h)anthracene	84	H	160	M
Fluoranthene	1 500	M	2 800	H
Pyrene	540	H	1 100	H
Total PAHs	10 000	H	20 000	H
Polychlorinated Biphenyls (PCBs; µg/kg DW)				
Aroclor 1254	210	H	410	ND
Total PCBs	170	H	330	H
Organochlorine Pesticides (µg/kg DW)				
Chlordane	5.5	H	11	H
Dieldrin	4.1	H	8.0	H
Endrin	39	M	75	ND
Heptachlor	1.7	H	3.3	ND
Heptachlor epoxide	1.7	H	3.3	ND
Lindane	0.86	L	1.7	ND
Sum DDD	5.3	H	10	H
Sum DDE	4.2	H	8.1	H
Sum DDT	3.0	H	5.7	H
PCDD/PCDFs (µg TEQ/kg DW)				
2,3,7,8-TCDD TEQ	0.13	ND	0.26	ND

L = Low; M = Moderate; H = High; ND = no data; PCDDs = polychlorinated dibenzo-*p*- dioxins; PCDFs = polychlorinated dibenzofurans; TCDD TEQ = tetrachlorodibenzo-*p*- dioxin toxic equivalent.

Table 10. **Incidence of toxicity within ranges of contaminant concentrations defined by the marine and estuarine sediment quality criteria ($SedQC_{SCS}$ and $SedQC_{TCS}$; based on CCME 1999), based on the results of 10-day amphipod toxicity tests[1] (survival of *Ampelisca abdita* and *Rhepoxynius abronius*), in the national database.**

Chemicals of Potential Concern (COPCs)	Number of Samples Evaluated[2]	$<SedQC_{SCS}$	$\geq SedQC_{SCS}$ to $\leq SedQC_{TCS}$	$\geq SedQC_{SCS}$	$>SedQC_{TCS}$
Metals					
Arsenic	1847	482 of 1780 (27%)	32 of 55 (58%)	36 of 67 (54%)	4 of 12 (33%)
Cadmium	1849	373 of 1718 (22%)	49 of 68 (72%)	98 of 131 (75%)	49 of 63 (78%)
Chromium	1893	334 of 1516 (22%)	153 of 279 (55%)	213 of 377 (56%)	60 of 98 (61%)
Copper	1876	194 of 1358 (14%)	115 of 238 (48%)	325 of 518 (63%)	210 of 280 (75%)
Lead	1890	274 of 1494 (18%)	95 of 184 (52%)	245 of 396 (62%)	150 of 212 (71%)
Mercury	1772	239 of 1426 (17%)	86 of 162 (53%)	215 of 346 (62%)	129 of 184 (70%)
Zinc	1881	210 of 1383 (15%)	177 of 315 (56%)	307 of 498 (62%)	130 of 183 (71%)
Polycyclic Aromatic Hydrocarbons (PAHs)					
Low Molecular Weight (LMW) PAHs					
2-Methylnaphthalene	1480	261 of 1374 (19%)	38 of 58 (66%)	70 of 106 (66%)	32 of 48 (67%)
Acenaphthene	1344	153 of 1163 (13%)	31 of 77 (40%)	99 of 181 (55%)	68 of 104 (65%)
Acenaphthylene	1365	141 of 1181 (12%)	38 of 81 (47%)	104 of 184 (57%)	66 of 103 (64%)
Anthracene	1518	155 of 1242 (12%)	43 of 97 (44%)	152 of 276 (55%)	109 of 179 (61%)
Fluorene	1426	185 of 1254 (15%)	44 of 80 (55%)	113 of 172 (66%)	69 of 92 (75%)
Naphthalene	1463	228 of 1379 (17%)	25 of 47 (53%)	51 of 84 (61%)	26 of 37 (70%)
Phenanthrene	1593	205 of 1337 (15%)	54 of 113 (48%)	155 of 256 (61%)	101 of 143 (71%)
High Molecular Weight (HMW) PAHs					
Benz(a)anthracene	1594	193 of 1326 (15%)	61 of 110 (55%)	169 of 268 (63%)	108 of 158 (68%)
Benzo(a)pyrene	1611	205 of 1334 (15%)	53 of 107 (50%)	169 of 277 (61%)	116 of 170 (68%)
Chrysene	1589	188 of 1317 (14%)	60 of 107 (56%)	167 of 272 (61%)	107 of 165 (65%)
Dibenz(a,h)anthracene	1517	187 of 1268 (15%)	50 of 103 (49%)	151 of 249 (61%)	101 of 146 (69%)
Fluoranthene	1601	221 of 1371 (16%)	55 of 99 (56%)	143 of 230 (62%)	88 of 131 (67%)
Pyrene	1622	216 of 1364 (16%)	70 of 120 (58%)	165 of 258 (64%)	95 of 138 (69%)
Total PAHs[3]	1600	249 of 1467 (17%)	45 of 76 (59%)	90 of 133 (68%)	45 of 57 (79%)

Table 10. Incidence of toxicity within ranges of contaminant concentrations defined by the marine and estuarine sediment quality criteria ($SedQC_{SCS}$ and $SedQC_{TCS}$; based on CCME 1999), based on the results of 10-day amphipod toxicity tests[1] (survival of *Ampelisca abdita* and *Rhepoxynius abronius*), in the national database.

Chemicals of Potential Concern (COPCs)	Number of Samples Evaluated[2]	$<SedQC_{SCS}$	$\geq SedQC_{SCS}$ to $\leq SedQC_{TCS}$	$\geq SedQC_{SCS}$	$>SedQC_{TCS}$
Polychlorinated Biphenyls (PCBs)					
Aroclor 1254	187	47 of 169 (28%)	2 of 2 (100%)	17 of 18 (94%)	15 of 16 (94%)
Total PCBs[4]	1588	137 of 1207 (11%)	44 of 146 (30%)	205 of 381 (54%)	161 of 235 (69%)
Organochlorine Pesticides					
Chlordane[5]	1440	144 of 1186 (12%)	30 of 85 (35%)	160 of 254 (63%)	130 of 169 (77%)
Dieldrin	1063	110 of 927 (12%)	43 of 70 (61%)	90 of 136 (66%)	47 of 66 (71%)
Endrin	1035	142 of 1033 (14%)	1 of 1 (100%)	2 of 2 (100%)	1 of 1 (100%)
Heptachlor	1271	166 of 1225 (14%)	14 of 23 (61%)	30 of 46 (65%)	16 of 23 (70%)
Heptachlor epoxide	1113	101 of 1071 (9%)	12 of 20 (60%)	29 of 42 (69%)	17 of 22 (77%)
Lindane	1050	93 of 947 (10%)	26 of 62 (42%)	40 of 103 (39%)	14 of 41 (34%)
Sum DDD[6]	1542	121 of 1135 (11%)	32 of 103 (31%)	211 of 407 (52%)	179 of 304 (59%)
Sum DDE[7]	1606	347 of 1546 (22%)	4 of 7 (57%)	26 of 60 (43%)	22 of 53 (42%)
Sum DDT[8]	1369	84 of 1107 (8%)	31 of 66 (47%)	158 of 262 (60%)	127 of 196 (65%)
PCDD/PCDFs					
2,3,7,8-TCDD TEQ[6]	29	4 of 20 (20%)	1 of 1 (100%)	9 of 9 (100%)	8 of 8 (100%)

$SedQC_{SCS}$ = sediment quality criteria for sensitive contaminated sites; $SedQC_{TCS}$ = sediment quality criteria for typical contaminated sites; PCDDs = polychlorinated dibenzo-*p*- dioxins; PCDFs = polychlorinated dibenzofurans; TCDD TEQ = tetrachlorodibenzo-*p*- dioxin toxic equivalent; WHO = World Health Organization

[1]Individual samples were designated as toxic based on a statistically significant difference from the control sample. If the measurement of the COPC is less than the detection limit, the value the detection limit was assigned.

[2]Excluding results for which the detection limit was greater than the probable effect level (PEL; CCME 1999), and results for which the COPC did not contribute substantially to the observed toxicity (see methods section for details).

(footnotes continued on next page)

Table 10. Incidence of toxicity within ranges of contaminant concentrations defined by the marine and estuarine sediment quality criteria (SedQC_{SCS} and SedQC_{TCS}; based on CCME 1999), based on the results of 10-day amphipod toxicity tests[1] (survival of *Ampelisca abdita* and *Rhepoxynius abronius*), in the national database.

Chemicals of Potential Concern (COPCs)	Number of Samples Evaluated[2]	$<\mathrm{SedQC}_{SCS}$	$\geq\mathrm{SedQC}_{SCS}$ to $\leq\mathrm{SedQC}_{TCS}$	$\geq\mathrm{SedQC}_{SCS}$	$>\mathrm{SedQC}_{TCS}$

[3]The concentrations of 2-methylnaphthalene, acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene and phenanthrene were summed to calculate LMW-PAHs. The concentrations of benz(a)anthracene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene, fluoranthene and pyrene were summed to calculate HMW-PAHs. The concentrations of LMW-PAHs and HMW-PAHs were summed to calculate Total PAHs.

[4]The concentrations of PCB congeners or Aroclors were summed to calculate Total PCBs.

[5]The concentrations of alpha- and gamma-chlordane, or cis-, and trans-chlordane were summed to calculate total chlordane.

[6]The concentrations of p,p'-DDD and o,p'-DDD were summed to calculate Sum DDD.

[7]The concentrations of p,p'-DDE and o,p'-DDE were summed to calculate Sum DDE.

[8]The concentrations of p,p'-DDT and o,p'-DDT were summed to calculate Sum DDT.

[9]Calculated using the WHO (van den Berg *et al.* 1998) toxic equivalency factors (TEFs) for fish, based on the concentrations of PCDDs, PCDFs, and co-planar PCB congeners.

Table 11. Reliability of the marine and estuarine sediment quality criteria for assessing and managing sensitive ($SedQC_{SCS}$) and typical ($SedQC_{TCS}$) contaminated sediments.

Chemicals of Potential Concern (COPCs)	$SedQC_{SCC}$	Reliability	$SedQC_{TCS}$	Reliability
Metals (µg/kg DW)				
Arsenic	26 000	M	50 000	L
Cadmium	2 600	M	5 000	H
Chromium	99 000	M	190 000	H
Copper	67 000	H	130 000	H
Lead	69 000	H	130 000	H
Mercury	430	H	840	H
Zinc	170 000	H	330 000	H
Polycyclic Aromatic Hydrocarbons (PAHs; µg/kg DW)				
2-Methylnaphthalene	120	H	240	H
Acenaphthene	55	H	110	H
Acenaphthylene	79	H	150	H
Anthracene	150	H	290	H
Fluorene	89	H	170	H
Naphthalene	240	H	470	H
Phenanthrene	340	H	650	H
Benz(a)anthracene	430	H	830	H
Benzo(a)pyrene	470	H	920	H
Chrysene	520	H	1 000	H
Dibenz(a,h)anthracene	84	H	160	H
Fluoranthene	930	H	1 800	H
Pyrene	870	H	1 700	H
Total PAHs	10 000	H	20 000	H
Polychlorinated Biphenyls (PCBs; µg/kg DW)				
Aroclor 1254	440	M	850	H
Total PCBs	120	H	230	H
Organochlorine Pesticides (µg/kg DW)				
Chlordane	3.0	H	5.7	H
Dieldrin	2.7	H	5.2	H
Endrin	39	H	75	ND
Heptachlor	1.7	H	3.3	H
Heptachlor epoxide	1.7	H	3.3	H
Lindane	0.61	H	1.2	L
Sum DDD	4.8	H	9.4	H
Sum DDE	230	M	450	M
Sum DDT	3.0	H	5.7	H
PCDD/PCDFs (µg TEQ/kg DW)				
2,3,7,8-TCDD TEQ	0.13	H	0.26	ND

L = Low; M = Moderate; H = High; ND = no data; DW = dry weight; PCDDs = polychlorinated dibenzo-*p*- dioxins; PCDFs = polychlorinated dibenzofurans; TEQ = toxic equivalents; TCDD = tetrachlorodibenzo-*p*- dioxin.

Figures

Figure 1. General process for managing contaminated sites in British Columbia.

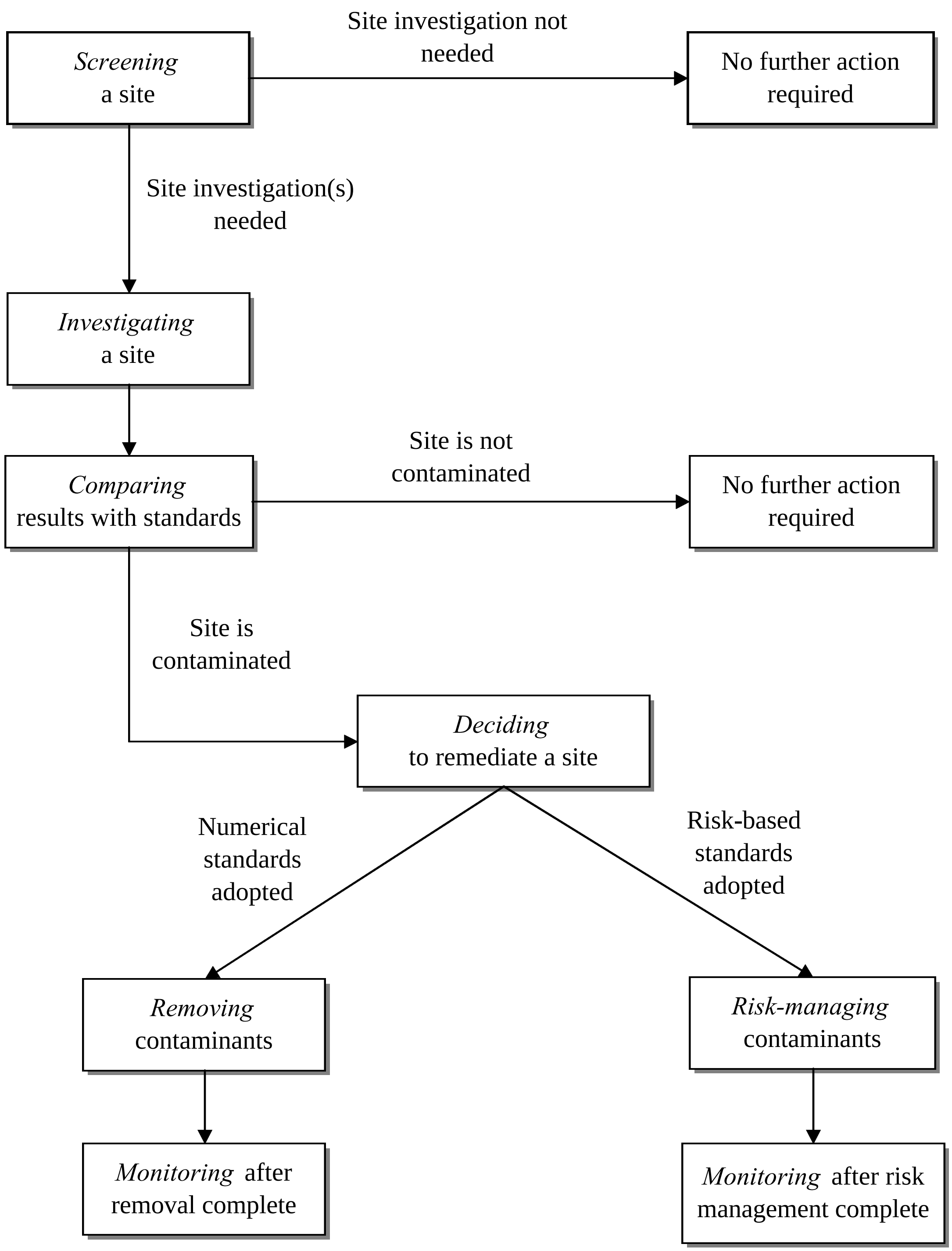


Figure 2. Overview of the recommended process for managing sediment contaminated sites in British Columbia.

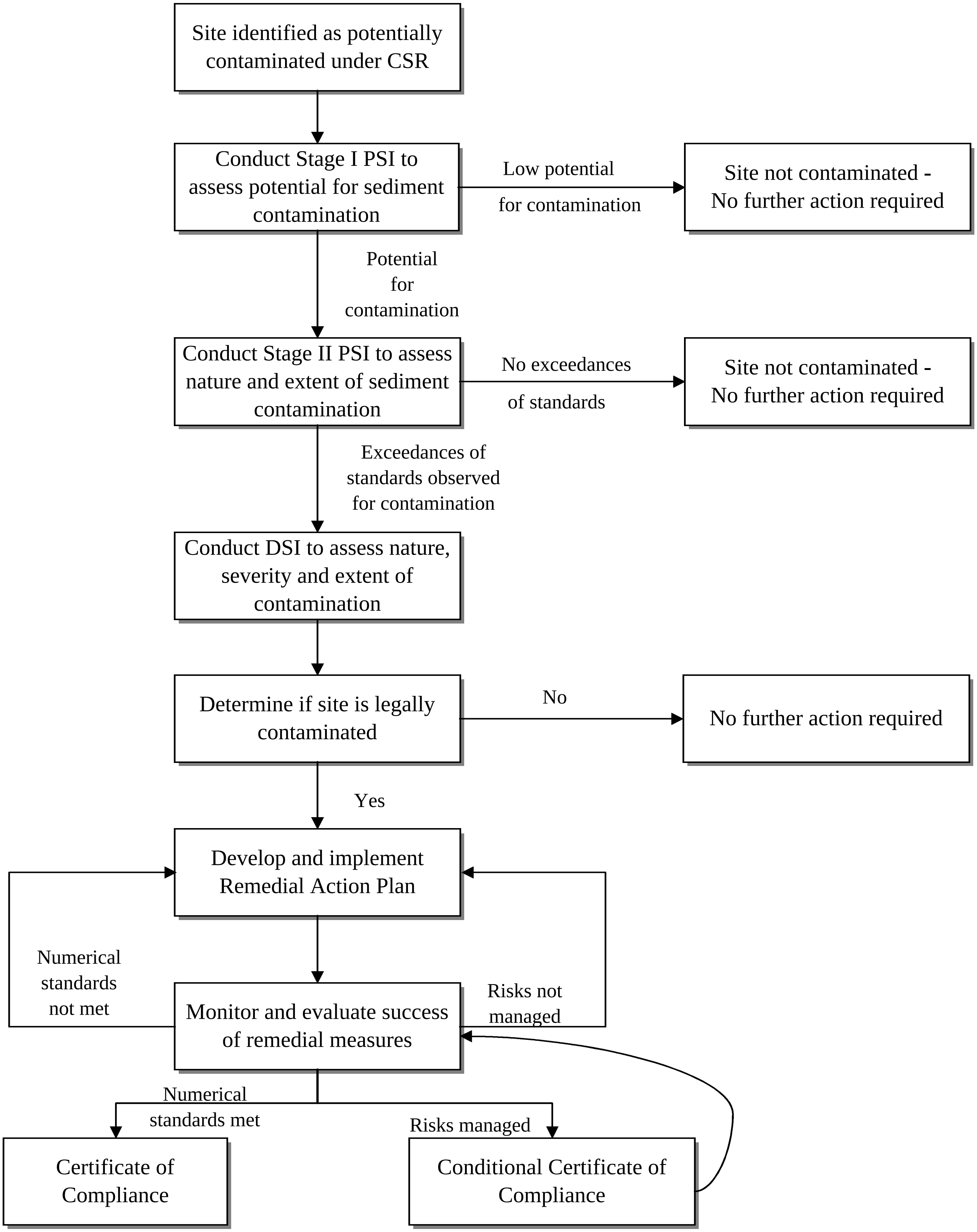


Figure 3. Relationship between mean probable effect level-quotients (PEL-Qs) and incidence of toxicity, based on the results of 28- to 42-day toxicity tests with the freshwater amphipod, *Hyalella azteca*.

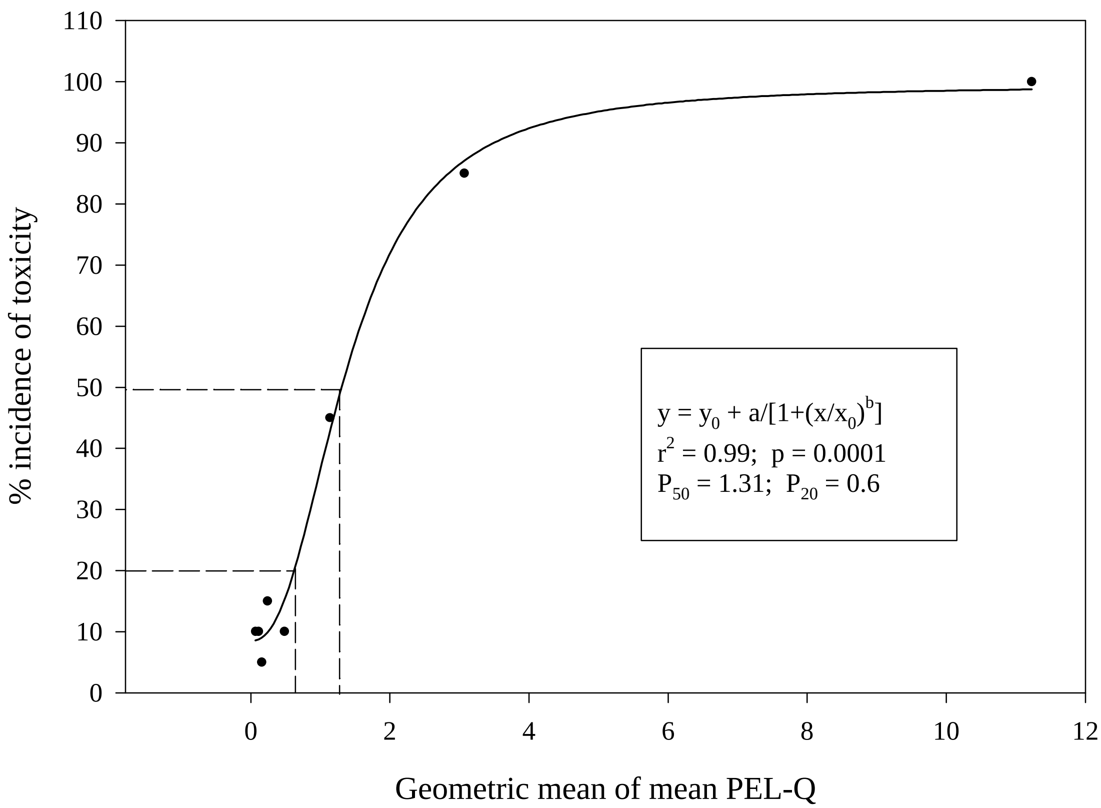
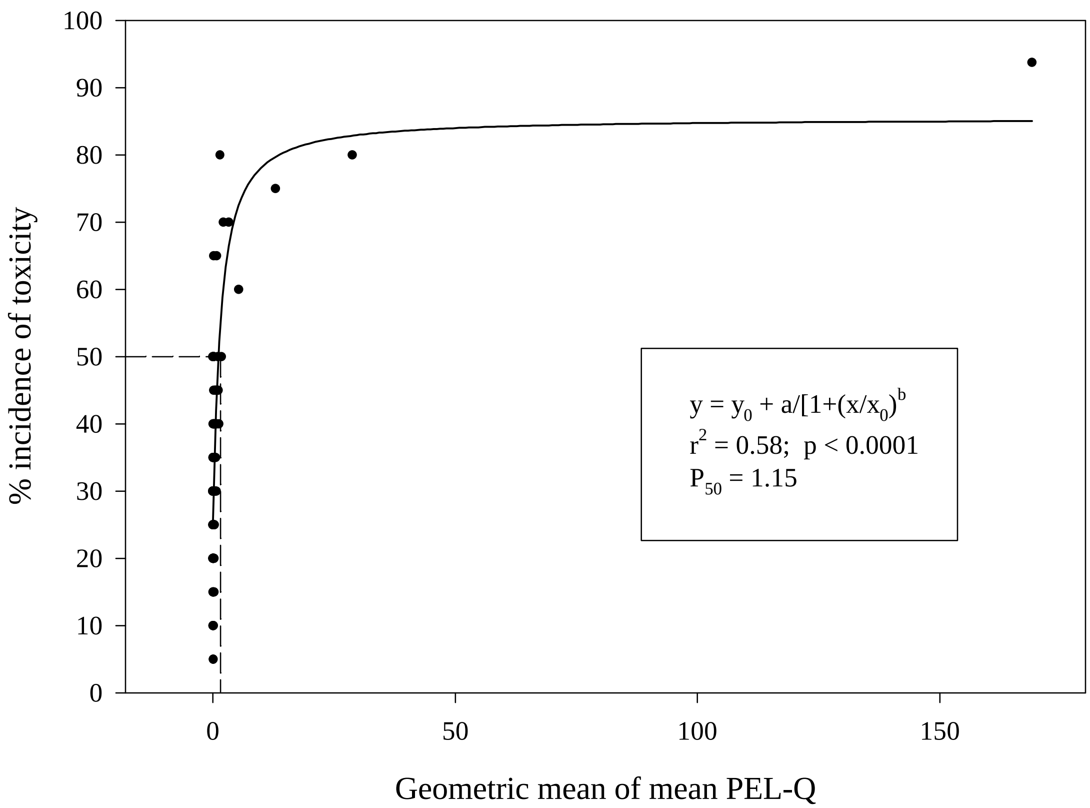


Figure 4. Relationship between mean probable effect level-quotients (PEL-Qs) and incidence of toxicity, based on the results of 10-day toxicity tests with marine and estuarine amphipods (*Ampelisca abdita* and *Rhepoxynius abronius*).



Appendices

Appendix 1 Criteria for Evaluating Candidate Data Sets

A1.1 Introduction

In recent years, the Great Lakes National Program Office (USEPA), United States Geological Survey, National Oceanic and Administration, Minnesota Pollution Control Agency, Florida Department of Environmental Protection, British Columbia Ministry of Water, Air, and Land Protection, MacDonald Environmental Sciences Ltd., and EVS Consultants have been developing a database of matching sediment chemistry and sediment toxicity data to support evaluations of the predictive ability of numerical sediment quality guidelines (SQGs) in the Great Lakes Basin and elsewhere in North America (Field *et al.* 1999; USEPA 2000a; Crane *et al.* 2000). In addition, various project-specific databases have been developed to facilitate access to and analysis of data sets to support natural resource damage assessments and ecological risk assessments at sites with contaminated sediments (MacDonald and Ingersoll 2000; Crane *et al.* 2000; MacDonald *et al.* 2001a; 2001b; Ingersoll *et al.* 2001). The goal of these initiatives was to collect and collate the highest quality data sets for assessing sediment quality conditions at contaminated sites and evaluating numerical SQGs. To assure that the data used in these assessments met the associated data quality objectives (DQOs), all of the candidate data sets were critically evaluated before inclusion in the database. However, the screening process was also designed to be flexible to assure that professional judgement could also be used when necessary in the evaluation process. In this way, it was possible to include as many data sets as possible and, subsequently, use them to the extent that the data quality and quantity dictate.

The following criteria for evaluating candidate data sets were established in consultation with an *ad hoc* Science Advisory Group on Sediment Quality Assessment (which is comprised of representatives of federal, provincial, and state government agencies, consulting firms, and non-governmental organizations located throughout North America and elsewhere worldwide). These criteria are reproduced here because they provide useful guidance on the evaluation of data that have been generated to support sediment quality assessments. In addition, these criteria can be used to support the design of sediment sampling and analysis plans, and associated quality assurance project plans (MacDonald and Ingersoll 2002).

A1.2 Criteria for Evaluating Whole-Sediment, Pore-Water, and Tissue Chemistry

Data on the chemical composition of whole sediments, pore water, and biological tissues are of fundamental importance in assessments of sediment quality conditions. For this reason, it is essential to ensure that high quality data are generated and used to support such sediment quality assessments. In this respect, data from individual studies are considered to be acceptable if:

- Samples were collected from any sediment horizon (samples representing surficial sediments are most appropriate for assessing effects on sediment-dwelling organisms and other receptors, while samples of sub-surface sediments are appropriate for assessing potential effects on sediment-dwelling organisms and other receptors, should these sediments become exposed; ASTM 2003a; ASTM 2003d; USEPA 2000b);
- Appropriate procedures were used for collecting, handling, and storing sediments (e.g., ASTM 2003b; 2003c; USEPA 2001) and samples of other media types;
- The concentrations of a variety of all chemicals of potential concern (COPCs) were measured in samples;
- Appropriate analytical methods were used to generate chemistry data. The methods that are considered to be appropriate included United States Environmental Protection Agency (USEPA) approved methods, other standardized methods [e.g., American Society for Testing and Materials (ASTM) methods, SW-846 methods], or methods that have been demonstrated to be equivalent or superior to standard methods; and,
- Data quality objectives were met. The criteria that are used to evaluate data quality included:

  (i) the investigator indicated that DQOs had been met;

  (ii) analytical detection limits were reported and lower than the probable effect concentrations (PECs) (however, detection limits < threshold effect concentration (TEC) are preferred);

(iii) accuracy and precision of the chemistry data were reported and within acceptable ranges for the method;

(iv) sample contamination was not noted (i.e., analytes were not detected at unacceptable concentrations in method blanks); and,

(v) the results of a detailed independent review indicated that the data were acceptable and/or professional judgement indicated that the data set was likely to be of sufficient quality to be used in the assessment (i.e., in conjunction with author communications and/or other investigations).

A1.3 Criteria for Evaluating Biological Effects Data

Data on the effects of contaminated sediments on sediment-dwelling organisms and other aquatic species provide important information for evaluating the severity and extent of sediment contamination. Data from individual studies are considered to be acceptable for this purpose if:

- Appropriate procedures were used for collecting, handling, and storing sediments (e.g., ASTM 2003b; USEPA 2000b; 2001); Sediments were not frozen before toxicity tests were initiated (ASTM 2003a; 2003e);
- The responses in the negative control and/or reference groups were within accepted limits (i.e., ASTM 2003a; 2003c; 2003d; 2003e; 2003f; 2003g; USEPA 2000a);
- Adequate environmental conditions were maintained in the test chambers during toxicity testing (i.e., ASTM 2003a; 2003d; USEPA 2000a);
- The endpoint(s) measured were ecologically-relevant (i.e., likely to influence the organism's viability in the field) or indicative of ecologically-relevant endpoints; and,
- Appropriate procedures were used to conduct bioaccumulation tests (ASTM 2003c).

Additional guidance is presented in USEPA (1994) for evaluating the quality of benthic community data generated as part of a sediment quality assessment. These criteria include collection of replicate samples, resorting at least 10% of the samples, and independent checks of taxonomic identification of specimens. Guidance is presented in USEPA (2000c) and in Schmidt *et al.* (2000) for evaluating the quality of fish health and fish community data.

A1.4 References

ASTM (American Society for Testing and Materials). 2003a. Standard test methods for measuring the toxicity of sediment-associated contaminants with freshwater invertebrates. E1706-00. *In:* ASTM 2003 Annual Book of Standards Volume 11.05. West Conshohocken, Pennsylvania.

ASTM (American Society for Testing and Materials). 2003b. Standard guide for designing biological tests with sediments. E1525-02. *In:* ASTM 2003 Annual Book of Standards Volume 11.05. West Conshohocken, Pennsylvania.

ASTM (American Society for Testing and Materials). 2001c. Standard guide for collection, storage, characterization, and manipulation of sediments for toxicological testing and for selection of samples used to collect benthic invertebrates. E1391-03. *In:* ASTM 2003 Annual Book of Standards Volume 11.05. West Conshohocken, Pennsylvania.

ASTM (American Society for Testing and Materials). 2003d. Standard guide for determination of the bioaccumulation of sediment-associated contaminants by benthic invertebrates. E1688-00a. *In:* ASTM 2003 Annual Book of Standards Volume 11.05. West Conshohocken, Pennsylvania.

ASTM (American Society for Testing and Materials). 2003e. Standard guide for conducting early life-stage toxicity tests with fishes. E1241-98. *In:* ASTM 2003 Annual Book of Standards Volume 11.05. West Conshohocken, Pennsylvania.

ASTM (American Society for Testing and Materials). 2003f. Standard guide for conducting acute toxicity tests on test materials with fishes, macroinvertebrates, and amphibians. E729-96. *In:* ASTM 2003 Annual Book of Standards Volume 11.05. West Conshohocken, Pennsylvania.

ASTM (American Society for Testing and Materials). 2003g. Standard guide for conducting *Daphnia magna* life-cycle toxicity tests. E1193-97. *In:* ASTM 2003 Annual Book of Standards Volume 11.05. West Conshohocken, Pennsylvania.

Crane, J.L., D.D. MacDonald, C.G. Ingersoll, D.E. Smorong, R.A. Lindskoog, C.G. Severn, T.A. Berger, and L.J. Field. 2000. Development of a framework for evaluating numerical sediment quality targets and sediment contamination in the St. Louis River Area of Concern. EPA 905-R-00-008. Great Lakes National Program Office. United States Environmental Protection Agency. Chicago, Illinois. 107 pp. + appendices.

Field, L.J., D.D. MacDonald, S.B. Norton, C.G. Severn, and C.G. Ingersoll. 1999. Evaluating sediment chemistry and toxicity data using logistic regression modeling. Environmental Toxicology and Chemistry 18(6):1311-1322.

Ingersoll, C.G., D.D. MacDonald, W.G. Brumbaugh, S.D. Ireland, B.T. Johnson, N.E. Kemble, J.L. Kunz, and T.W. May. 2001. Toxicity assessment of sediments from the Grand Calumet River and Indiana Harbor Canal in northwestern Indiana. Archives of Environmental Contamination and Toxicology. In press.

MacDonald, D.D. and C.G. Ingersoll. 2000. An assessment of sediment injury in the grand Calumet River, Indiana Harbor Canal, Indiana Harbor, and the nearshore areas of Lake Michigan. Volume I. Prepared for the U.S. Fish and Wildlife Service. Bloomington, Indiana. 238 pp.

MacDonald, D.D. and C.G. Ingersoll. 2002. A guidance manual to support the assessment of contaminated sediments in freshwater ecosystems. Volume I – An ecosystem-based framework for assessing and managing contaminated sediments. EPA-905-B02-001-A. Prepared for the Great Lakes National Program Office. United States Environmental Protection Agency. Chicago, Illinois. Under contract to Sustainable Fisheries Foundation. Snohomish, Washington.

MacDonald, D.D., D.R. Moore, A. Pawlisz, D.E. Smorong, R.L. Breton, D.B. MacDonald, R. Thompson, R.A. Lindskoog and M.A. Hanacek. 2001a. Calcasieu estuary remedial investigation/feasibility study (RI/FS): Baseline ecological risk assessment (BERA). Volume I. Baseline problem formulation. Prepared for: CDM Federal Programs Corporation. Dallas, Texas.

MacDonald, D.D., D.R. Moore, A. Pawlisz, D.E. Smorong, R.L. Breton, D.B. MacDonald, R. Thompson, R.A. Lindskoog and M.A. Hanacek. 2001b. Calcasieu estuary remedial investigation/feasibility study (RI/FS): Baseline ecological risk assessment (BERA). Volume 2. Baseline problem formulation: Appendices. Prepared for: CDM Federal Programs Corporation. Dallas, Texas.

MESL (MacDonald Environmental Sciences Ltd). 1997. Lower Columbia River from Birchbank to the International Boundary: Water quality and quantity assessment and objectives technical report. Prepared for Environment Canada, Vancouver, British Columbia and the British Columbia Ministry of Environment, Lands and Parks, Victoria, British Columbia.

Schmitt, C.J., D.E. Tillitt, and V.S. Blazer. 2000. The 1995 and 1997 BEST projects. *In:* Biomonitoring of Environmental Status and Trends (BEST) Program: Selected Methods for Monitoring Chemical Contaminants and their Effects in Aquatic Ecosystems. C.J. Schmitt and G.M. Dethloff (Eds.). Information and Technology Report USGS/BRD/ITR–2000-0005. U.S. Geological Survey. Columbia, Missouri.

USEPA (United States Environmental Protection Agency). 1994. Assessment and remediation of contaminated sediments (ARCS) program. Great Lakes National Program Office. EPA 905/B-94/002. Chicago, Illinois.

USEPA (United States Environmental Protection Agency). 2000a. Methods for measuring the toxicity and bioaccumulation of sediment-associated contaminants with freshwater invertebrates. Second edition. EPA 600/R-99/064. Office of Research and Development. Washington, District of Columbia.

USEPA (United States Environmental Protection Agency). 2000b. Prediction of sediment toxicity using consensus-based freshwater sediment quality guidelines. EPA 905/R-00/007. Great Lakes Program Office. Chicago, Illinois.

USEPA (United States Environmental Protection Agency). 2000c. Stressor identification guidance document. EPA-822-B-00-025. Office of Water. Office of Research and Development. Washington, District of Columbia.

USEPA (United States Environmental Protection Agency). 2001. Methods for collection, storage and manipulation of sediments for chemical and toxicological analyses: Technical manual. EPA-823-B-01-002. Office of Water. Washington, District of Columbia.

Appendix 2 Guiding Principles and Administrative Rules for Developing Site-Specific Sediment Quality Standards.

Formulation of site-specific SedQSs is a multi-stepped procedure that requires both environmental management and detailed technical information. The first step in this process is to identify the designated uses of the aquatic environment at the contaminated site. In this way, environmental managers can establish the overall management goals that can be used to guide the remedial measures. With respect to contaminated sediments, three major uses of aquatic ecosystems are generally considered as management objectives, including:

- Protection of aquatic life;
- Protection of wildlife; and,
- Protection of human health (including recreation and aesthetics).

Establishment of the intended uses of the aquatic ecosystem following remediation provides a basis for establishing narrative objectives that will clarify and focus the management goals for the site. For example, at sites that have been designated for the protection of aquatic life, the narrative SQRO might be:

> ***Bed sediments should not be toxic to aquatic organisms and should support a healthy and diverse benthic community.***

Establishment of such a narrative objective, in turn, will support the development of specific biological and chemical indicators that could be used to assess the current status of the bed sediments, provide target clean-up levels to guide remedial actions, and evaluate the effectiveness of those activities.

A2.1 Guiding Principles

The following guiding principles for the development of numerical SQGs for contaminated sites in British Columba are based on the philosophy established by BC Environment and the CCME (BC Environment 1986; CCME 1995):

- Site-specific SedQSs should be developed to protect the most sensitive water uses at a contaminated site.
- Protection of freshwater, estuarine, and marine aquatic life, wildlife, and human health are the primary uses of aquatic ecosystems that are dependent on sediment quality.
- The generic SQGs for the most sensitive use should be adopted as the preliminary SedQSs for a site.
- At sites which have atypical characteristics or receptors, the SedQSs may be modified to account for these site-specific factors.
- The administrative rules (see below) specify the conditions under which the SedQSs may be modified.
- For the purpose of deriving site-specific SedQSs for the protection of aquatic life, information on the aquatic organisms (e.g., algae, invertebrates, fish, and amphibians) that are not relevant to the site under consideration may be omitted from the national data set, provided that the minimum data requirements for deriving SQGs are met (CCME 1995).
- The approach used to develop site-specific SedQSs should follow the formal protocols established by the CCME (1995).
- Technical, social, and economic issues relating to the development of final SedQSs should be reviewed and assessed by the agency(ies) responsible for approval of the remedial action plan.
- Both chemical (numerical SedQSs) and biological (bioassay results, aquatic ecosystem community structure, etc.) indicators should be used to evaluate attainment of the management goals at a site following remediation.

- Unless otherwise specified, the SedQS refers to the total concentration of the substance in bulk sediments, expressed on a dry weight basis.

A2.2 Administrative Rules

Derivation of site-specific SedQSs is a complex process that requires detailed information on the site under investigation, on the contaminants present at the site, and on potential exposure to human and environmental receptors. This process is further complicated by the detailed procedures that have been developed for formulating SQGs (CCME 1995) and for modifying these values to account for site characteristics (MacDonald 1998; EVS Consultants Ltd. 1993). For this reason, a series of administrative rules have been recommended to simplify the process of deriving site specific SedQSs and to ensure that this process is implemented in a fair and consistent manner at contaminated sites throughout province. These administrative rules dictate when it is appropriate to adopt the matrix numerical standards directly, to modify the matrix numerical standards, and to develop risk-based standards.

- The SQG-based approach should be used to derive SedQSs unless the information required to support this approach are not available (i.e., if there are unacceptable data gaps). Under these conditions, the risk-based approach should be used to derive the SedQS.
- The SQG-based approach should be used to derive SedQSs unless there is significant potential for the contaminants that are present at the contaminated site to undergo unpredictable transformations. Under these conditions, the risk-based approach should be used to derive the SedQS.
- SQGs for the COPCs must be available for each of the designated water uses at the contaminated sites before selecting a preliminary SedQS, unless it can be demonstrated that the most sensitive uses are adequately protected by the available SQGs.
- If SQGs are not available for one or more of the water uses at the contaminated sites, then the missing SQGs may be derived using the appropriate protocol.

- If SQGs are not available for one or more of the water uses at the contaminated site and insufficient data is available to support their derivation, then the additional toxicological and/or environmental fate data may be generated that is required to support the derivation of SQGs. Alternatively, site-specific SedQSs may be derived using the risk-based approach.
- The preliminary SedQSs (PSedQSs) shall be adopted as final SedQSs at contaminated sites unless the SedQS for a substance is lower than the upper limit of background at the site under investigation. Procedures for determining background levels at contaminated sites are recommended in MacDonald (1998).
- The procedure used to determine background concentrations of priority substances in sediment at a contaminated site must be approved by the responsible agency.
- If insufficient data are available to determine background concentrations of priority substances in sediment at a contaminated site, then a proponent may (in conjunction with the responsible agency) designate an appropriate reference site and collect the data necessary to determine these levels.
- The PSedQSs shall be adopted as SedQSs at contaminated sites unless the criterion for a substance is lower that the analytical detection limit for that substance.
- The analytical detection limits for chemical substances vary depending on the extraction and quantification techniques used, the medium sampled, and the laboratory considered. It is recommended that the lowest analytical detection limits that are typically achieved at the National Water Quality Laboratory (Burlington, Ontario) should be used to evaluate the applicability of PSedQS.
- The PSedQSs shall be adopted as SedQSs at contaminated sites unless it can be demonstrated that the toxicity data set (i.e., the species and life stages) that was used to derive the SQGs is not entirely relevant to the site under investigation.
- If a site has an atypical assemblage of aquatic organisms, the PSedQS may be recalculated using the toxicological information that is applicable to the contaminated site under investigation.

- To implement the recalculation procedure, the information in the site-specific toxicological data set must satisfy the minimum toxicological data set requirements for deriving interim Canadian SQGs (CCME 1995).
- If insufficient data are available in the site-specific toxicological data set to support the derivation in interim SQGs, supplementary data may be generated by conducting toxicity tests using indicator or resident species.
- The PSedQSs shall be adopted as SedQSs at contaminated sites unless it can be demonstrated that the toxicity of a substance is dependent on an environmental factor (e.g., AVS, water hardness, pH) that was not considered in the derivation of the criterion and the site under investigation has atypical levels of that factor.
- The PSedQS may be modified to account for atypical levels of the factors that are considered to affect the bioavailability and/or toxicity of a substance. For example, the PSedQS for a non-polar organic substance may be modified if the median TOC value at the site falls outside the range of TOC values represented in BEDS (for that substance). In this respect, the arithmetic mean ± two standard deviations should be used to define the typical range of TOC values in BEDS (i.e., 0.1 to 4.7% for marine and estuarine sediments and 0.4 to 10.1% for freshwater sediments).
- The ***recommended SedQS*** developed using these procedures are intended provide the scientific tools required to support the remediation contaminated sites. However, there are a number of additional factors that may be considered by the responsible agency(ies) in the derivation of ***final SedQS***, including the availability of appropriate remediation technology, anticipated clean-up costs, and others.

A2.3 References

BC Environment (British Columbia Ministry of the Environment). 1986. Principles for preparing water quality objectives in British Columbia. Resource Quality Section. Water Management Branch. Victoria, British Columbia. 20 pp

CCME (Canadian Council of Ministers of the Environment). 1995. Protocol for the derivation of Canadian sediment quality guidelines for the protection of aquatic life. Prepared by the Technical Secretariat of the CCME Task Group on Water Quality Guidelines. Ottawa, Canada.

EVS Consultants Ltd. 1993. A framework for ecological risk assessment at contaminated sites in Canada. Prepared for Eco-Health Assessment Branch. Environment Canada. Ottawa, Canada.

MacDonald, D.D. 1998. Applications of sediment quality guidelines in the remediation of sediment contaminated sites in British Columbia. Prepared for Ministry of Environment, Lands and Parks. Victoria, British Columbia.