Canadian Sediment Quality Guidelines for the Protection of Aquatic Life

PROTOCOL

Protocol for the Derivation of Canadian Sediment Quality Guidelines for the Protection of Aquatic Life

Canadian Council of Ministers of the Environment 1995
CCME EPC-98E

Contents

Abstract

Sediments provide habitat for many benthic and epibenthic organisms. They also influence the environmental fate of many chemical substances in aquatic ecosystems by acting as both sinks and subsequently sources of substances that have entered the aquatic environment. Many aquatic organisms may be exposed to chemical substances through their immediate interactions with bed sediments; therefore, benchmarks of environmental quality (such as sediment quality guidelines) are required to support protection and management strategies for freshwater, estuarine, and marine ecosystems. Under the auspices of the Canadian Council of Ministers of the Environment (CCME), Canadian sediment quality guidelines for the protection of aquatic life are being developed through the CCME Task Group on Water Quality Guidelines. These sediment quality guidelines can be used to assess sediment quality, to help set targets for sediment quality that will sustain aquatic ecosystem health for the long term, and to develop site-specific objectives. This document outlines the procedures that are set out for deriving scientifically sound national sediment quality guidelines for the protection of aquatic life. Introductory guidance is also provided on how these guidelines are intended to be used in conjuction with other types of information.

Canadian Environmental Quality Guidelines
Canadian Council of Ministers of the Environment, 1999

Résumé

Les sédiments servent d'habitat à de nombreux organismes benthiques et épibenthiques. Ils influent également sur le devenir environnemental de nombreuses substances chimiques dans les écosystèmes aquatiques en se comportant à la fois comme puits d'accumulation et par la suite comme sources des substances qui se sont introduites dans l'environnement aquatique. Nombre d'organismes aquatiques peuvent être exposés à des substances chimiques par suite de leurs interactions immédiates avec les sédiments du lit; par conséquent, des points de repère en matière de qualité environnementale (comme des recommandations pour la qualité des sédiments) sont nécessaires pour appuyer les stratégies de protection et de gestion des écosystèmes d'eau douce et des écosystèmes estuariens et marins. Des recommandations canadiennes pour la qualité des sédiments visant à protéger la vie aquatique sont en voie d'élaboration par l'intermédiaire du Groupe de travail sur les recommandations pour la qualité des eaux, sous les auspices du Conseil canadien des ministres de l'environnement (CCME). Ces recommandations pour la qualité des sédiments peuvent servir à évaluer la qualité des sédiments, à fixer des objectifs en matière de qualité des sédiments qui favorisent la santé à long terme des écosystèmes aquatiques et à établir des objectifs propres à des sites spécifiques. Le présent document décrit les méthodes en place pour l'élaboration de recommandations nationales pour la qualité des sédiments qui reposent sur des fondements scientifiques solides et qui ont pour but de protéger la vie aquatique. On fournit également une introduction sur l'utilisation prévue de ces recommandations en combinaison avec d'autres types de renseignements.

Acknowledgments

Drafts of this document were prepared by S.L. Smith (Evaluation and Interpretation Branch, Environment Canada, Ottawa, Ont. K1A OH3) and D.D. MacDonald (MacDonald Environmental Sciences Ltd., Ladysmith, B.C. VOR 2E0). Individuals who contributed significantly to this report include Karen Keenleyside (Evaluation and Interpretation Branch, Environment Canada); John Karau and Jim Osborne (Office of Waste Management, Environment Canada); and M. Bewers and D.H. Loring (Bedford Institute of Oceanography, Department of Fisheries and Oceans). Thanks are also extended to Connie Gaudet, Amanda Brady, Larissa Thomas, Rob Kent, Don McGirr, and Michael Wong (Evaluation and Interpretation Branch, Environment Canada); Dwight Williamson (Manitoba Environment); Ed Long (National Oceanic and Atmospheric Administration, Seattle, Wash.); Fred Calder (Florida Department of Environmental Protection, Tallahassee, Fla.); and the members of the Interdepartmental Working Group on Marine Environmental Quality Guidelines, the members of the Marine Environmental Quality Advisory Group, and the members of the CCME Task Group on Water Quality Guidelines for their technical advice and reviews.

Preface

Canadian sediment guidelines for the protection of aquatic life are being developed under the auspices of the Canadian Council of Ministers of the Environment (CCME). Sediment quality issues have become an important focus in the environmental assessment, protection, and management of aquatic ecosystems. Historically, water quality activities were motivated by concerns for human health (e.g., drinking water quality guidelines) (Health and Welfare Canada 1993), but attention has shifted in recent years towards the protection of other components of the ecosystem (e.g., sediments, soil) and other water uses. These water uses include freshwater and marine aquatic life, recreation and aesthetics, irrigation and livestock watering, and industrial water supplies. In Canada, acceptable water quality for the protection of these uses has been evaluated against the Canadian water quality guidelines (CCREM 1987). Sediment quality guidelines for the protection of aquatic life will be used in a complementary manner to evaluate sediment quality.

The Canadian water quality guidelines were adopted on the basis of a review and evaluation of existing water quality guidelines from other jurisdictions (CCREM 1987). In some cases, scientific information was used to modify the guidelines so that they were applicable to Canadian conditions. Guidelines were not recommended for the parameters for which guidelines from other jurisdictions were deemed inappropriate or for which scientific data were lacking. Recently, a formal protocol

has been developed for the consistent derivation of numerical water quality guidelines for the protection of freshwater and marine aquatic life (CCME 1991a) and for the protection of agricultural water uses (CCME 1993). Similarly, interim soil quality guidelines have been adopted for 60 substances, and a formal protocol for their refinement is currently being developed (CCME 1991b, 1994; Environment Canada 1991).

Water quality guidelines play an important role in protecting water uses and in assessing the impact of environmental contaminants on the quality and uses of aquatic resources. Sediment quality guidelines are also important because sediments have a profound influence on the health of aquatic organisms, which may be exposed to chemicals through their immediate interactions with bed sediments. Therefore, the use of sediment quality guidelines for evaluating the toxicological significance of sediment-associated chemicals has become an important part of the protection and management of freshwater, estuarine, and marine ecosystems.

In 1989, the CCME Environmental Protection Committee mandated the responsibility of establishing Canadian sediment quality guidelines (SQGs) to the CCME Task Group on Water Quality Guidelines. The Evaluation and Interpretation Branch, Ecosystem Conservation Directorate, of Environment Canada provides scientific and technical support to the Task Group on the development of these guidelines. Sediment quality guidelines can be used to help set targets for sediment quality that will sustain aquatic ecosystem health for the long term. They are required to support the interpretation of sediment chemistry data and the overall assessment of sediment quality conditions within the context of specific water uses, and they support the development of site-specific objectives.

The use and interpretation of the terms *criteria*, *guidelines*, *objectives*, and *standards* vary among different agencies and countries. For the purposes of this document, these terms are defined as follows:

Criteria - The scientific data that are evaluated to derive sediment quality guidelines.

Guidelines - Numerical limits or narrative statements recommended to support and maintain designated uses of the aquatic environment.

Objectives - Numerical limits or narrative statements that have been established to protect and maintain designated uses of the aquatic environment at a particular site.

Standards - Sediment quality objectives that are recognized in enforceable environmental control laws of one or more levels of government.

These definitions are consistent with those used in the discussion of Canadian water quality guidelines (CCREM 1987). The term *sediment* refers to the bottom deposits in aquatic environments that are composed of particulate material (of various sizes, shapes, mineralogy) from various sources (e.g., terrigenous, biogenic, authigenic).

In response to the identified need for SQGs in Canada, Environment Canada commissioned a study in 1988 (MacDonald et al. 1992) to review and evaluate the available approaches used to develop such guidelines. The document also provided an extensive compilation of existing sediment quality assessment values from around the world. The approaches reviewed included those of the

- sediment background
- spiked-sediment toxicity test
- water quality guidelines
- interstitial water toxicity
- equilibrium partitioning
- tissue residue
- benthic community structure assessment
- screening level concentration
- sediment quality triad
- apparent effect threshold
- International Joint Commission sediment assessment strategy
- National Status and Trends Program

MacDonald et al. (1992) provided a brief description of the methodology of each of the approaches, their major advantages and limitations, and their current uses. Numerous other reviews of these approaches have been published (Beak Consultants 1987, 1988; Chapman 1989; Sediment Criteria Subcommittee 1989; Adams et al. 1992; Persaud et al. 1992; Lamberson and Swantz 1992: Long and MacDonald 1992).

A preliminary evaluation indicated at that time that no single approach was likely to fully support the immediate need for national, scientifically defensible guidelines, as well as the long-term need for guidelines that explicitly consider the factors that influence the toxicity of sediment-associated contaminants. Until a formalized protocol was developed, MacDonald et al. (1992) recommended that effect-based sediment quality assessment values from other jurisdictions be evaluated for their applicability to Canadian conditions, modified

with existing scientific data (if necessary) to increase their applicability to Canadian conditions, and adopted as interim sediment quality guidelines if deemed suitable. This recommendation parallels the initial CCME strategy followed for adopting interim water quality and soil quality guidelines.

Further to the recommendations of MacDonald et al. (1992), a study was commissioned by Environment Canada to validate and update the National Status and Trends Program database (developed by the National Oceanic and Atmospheric Administration), which contained information on the biological effects of sediment-associated contaminants. The results of these initiatives provided a sound basis for developing a formal protocol (presented in Chapter 1) for the development of national SQGs for use in Canada.

The formal protocol established for the derivation of numerical SQGs is applicable to the protection of both freshwater and marine (including estuarine) aquatic life associated with bed sediments (separate guidelines are derived for each of these systems). The protocol relies mainly on the National Status and Trends Program approach, with the complementary use of the spiked-sediment toxicity test approach in the future, once methodological concerns have been resolved. Because the development of SQGs relies on current scientific information, they will be refined as new and relevant data become available.

Sediment quality guidelines are developed from the available scientific information on the biological effects of sediment-associated chemicals. These tools are intended to provide guidance to provincial, federal, territorial, and nongovernmental agencies involved in the protection, assessment, and management of sediment quality. SQGs provide a scientific review the existing toxicological information for a chemical, which can be used to support the establishment of sediment quality objectives to protect aquatic life, which are developed to reflect a number of site-specific considerations. The recommended SQGs may be employed as nationally consistent screening tools; however, variations in environmental conditions across Canada will affect sediment quality in different ways. Complementary information on background concentrations of natural substances is evaluated during the development of SQGs and should be considered during their implementation and the development of site-specific sediment quality objectives. Impairment of sediments of superior quality to national guideline concentrations should not be advocated.

This document focuses on the procedures to be used in deriving national SQGs for the protection of aquatic life. National SQGs are currently being developed on a chemical-by-chemical basis through the CCME Task Group on Water Quality Guidelines. This document also provides introductory guidance on how these guidelines are intended to be used with other types of information and will be followed by other documents providing national guidance specific to the implementation of SQGs and the development of site-specific sediment quality objectives.

Glossary

ASTM	American Society for Testing and Materials
BEDS	Biological Effects Database for Sediments
CCREM	Canadian Council of Resource and Environment Ministers
CCME	Canadian Council of Ministers of the Environment
EC_{50}	median effective concentration
ERL	effects range low
ERM	effects range median
ISQG	interim sediment quality guideline
LC_{50}	median lethal concentration
LOEL	lowest-observed-effect level
NC	no concordance
NE	no effect
NG	no gradient
NOEL	no-observed-effect level
NSTP	National Status and Trends Program
PEL	probable effect level
SG	small gradient
SQG	sediment quality guideline
SSTT	spiked-sediment toxicity test
TEL	threshold effect level
TOC	total organic carbon

Chapter 1
The Derivation of Canadian Sediment Quality Guidelines for the Protection of Aquatic Life

INTRODUCTION

Sediments provide habitat for many benthic and epibenthic organisms and are an important component of aquatic ecosystems. Sediments also influence the environmental fate of many toxic and bioaccumulative substances in aquatic ecosystems. Many substances form associations with particulate matter and are eventually incorporated into bed sediments (Allan 1986). Consequently, sediments may also act as long-term sources of these substances to the aquatic environment (Larsson 1985; Salomons et al. 1987; Loring and Rantala 1992).

Chemical contaminants in sediments have been associated with a wide range of impacts on the plants and animals that live within and upon bed sediments. Both acute and chronic toxicity of sediment-associated chemicals to algae, invertebrates, fish, and other organisms have been measured in laboratory toxicity tests (Thomas et al. 1986; Kosalwat and Knight 1987; Dawson et al. 1988; Long and Morgan 1990; Burton 1991, 1992; Burton et al. 1992; Lamberson et al. 1992). Field surveys have identified the subtler effects of environmental contaminants, such as the development of tumours and other abnormalities in bottom-feeding fish (Malins et al. 1984; Couch and Harshbarger 1985; Malins et al. 1985; Goyette et al. 1988). Sediment-associated chemicals also have the potential to accumulate in the tissues of aquatic organisms (Foster et al. 1987; Knezovich et al. 1987). Elevated tissue concentrations in benthic or other aquatic organisms can result in the bioaccumulation of chemicals in higher levels of the aquatic food web (Government of Canada 1991a, 1991c). (Bioaccumulation in the context of this document is defined as the process by which substances are accumulated by aquatic organisms from all routes of exposure.) The bioaccumulation of chemicals in aquatic organisms presents a potential hazard to sensitive wildlife species, birds, and humans that rely on these organisms for food.

The continual release of chemicals into the environment as a consequence of human activities has resulted in varying degrees of contamination of aquatic ecosystems across Canada (Waldichuk 1988; Goyette and Boyd 1989; Allan and Ball 1990; Government of Canada 1991a, 1991b, 1991c; Trudel 1991; Wells and Rolston 1991). Because sediments tend to integrate contaminant inputs over time, they represent potentially significant hazards to the health of aquatic organisms and to the overall health of aquatic ecosystems. Therefore, sediment quality guidelines (SQGs) for the protection of aquatic life are required to assess the toxicological significance of sediment-associated chemicals in freshwater, estuarine, and marine ecosystems.

CANADIAN PERSPECTIVE

Ideally, SQGs should be developed from detailed dose–response data that describe the acute and chronic toxicity of individual chemicals in sediments to sensitive life stages of sensitive species of aquatic organisms. These data should be generated in controlled laboratory studies in which the influence of important environmental variables that affect toxicity are identified and quantified. The results of these studies should then be validated in field trials to ensure that any SQGs derived from these data are applicable in a broad range of locations in Canada. A detailed understanding of the factors that influence toxicity would also support site-specific sediment quality assessments by providing a basis for evaluating the applicability of guidelines under site-specific conditions (e.g., total organic carbon [TOC], grain size, acid volatile sulphide) (DeWitt et al. 1988; Di Toro et al. 1990; Landrum and Robbins 1990; Swartz et al. 1990; Carlson et al. 1991; Di Toro et al. 1991; Loring and Rantala 1992; Ankley et al. 1991, 1993). These relationships need to be defined to the extent that the relative importance of the modifiers of sediment toxicity are predictable under field situations.

Currently, this ideal approach is not supported by adequate scientific information to facilitate the development of national SQGs. To date, only a limited number of controlled laboratory studies have been conducted to assess the effects of sediment-associated chemicals on estuarine, marine, and freshwater organisms (Long and Morgan 1990; Burton 1991; MacDonald 1993). Additional research into sediment-spiking methodologies is also required before the results of spiked-sediment bioassays can be used to generate dose–response data appropriate for guideline development. However, other

types of data that are routinely collected throughout North America contribute to our understanding of the toxic effects of these chemicals. Specifically, a wide variety of sediment toxicity tests have been conducted to assess the biological significance of concentrations of chemicals in sediments from specific geographic locations. These toxicity tests include those performed on benthic organisms (e.g., bivalve mollusks, shrimp, amphipods, polychaetes, nematodes, chironomids) and on pelagic organisms (e.g., sea urchin larvae, oyster larvae, luminescent bacteria). Further, numerous field studies have been conducted that assess the diversity and abundance of benthic infaunal and epifaunal species. Comprehensive data on the concentrations of chemicals in these sediments have also been collected for many of these field studies (Long and Morgan 1990; MacDonald 1993). Specific characteristics of the sediments and the overlying water column also aid in the interpretation of the corresponding toxicity data for the studies for which this information has been collected. These field studies, which report matching sediment chemistry and biological-effect data (i.e., data are collected from the same locations at the same time), provide information relevant to the SQG derivation process.

Available approaches to the development of SQGs were evaluated to determine which procedures would be most applicable in Canada to the development of national SQGs for the protection of aquatic life (MacDonald et al. 1992). In addition to the technical basis of these approaches, their practicality, scientific defensibility, and applicability were evaluated. The results of these comprehensive evaluations of the major approaches led to the establishment of a formal protocol that builds on the strengths of two complementary approaches: the National Status and Trends Program (NSTP) approach (see Appendix A) and the spiked-sediment toxicity test (SSTT) approach (see Appendix B).

The formal protocol described in the following sections is considered to be appropriate because it fulfils both the immediate and long-term needs for national SQGs in Canada. The protocol is practical in that it can be implemented in the short term using existing data. It is scientifically defensible because it is supported by a weight of evidence of the available toxicological data on sediment-associated chemicals. For SQGs to be effective in Canada, they must be formulated from an understanding of biological effects (preferably cause-and-effect relationships, including data on sensitive end points like growth, reproduction, and genotoxicity), and they should account for the factors (e.g., TOC) that influence the bioavailability of sediment-associated chemicals. Since information on the toxicity of field-collected sediments is used in deriving guidelines, the various factors that affect the bioavailability of chemicals are implicitly considered, as well as the effects of mixtures of chemicals. Therefore, the guidelines derived using this protocol are applicable to sediments under field conditions. The guideline derivation procedure is also applicable to all classes of chemical and mixtures of chemicals that are likely to occur in Canadian sediments. This procedure provides long-term applicability in that it provides a focus for research activities that will support guideline development, and it allows for the refinement of SQGs as new scientific information becomes available. Therefore, the SQGs developed will provide relevant benchmarks for addressing the protection of benthic organisms and for assessing the potential impact of sediment-associated chemicals on aquatic biota.

Sediment quality guidelines formulated on the basis of biological-effect data of sediment-associated chemicals are intended to be used as nationally consistent benchmarks. During their implementation, however, allowance must be made for the incidence of natural inorganic and organic substances in sediments. Adverse biological effects may be observed below measured chemical concentrations that are attributable to natural enrichment. However, management concerns over the potential for adverse effects of sediment-associated chemicals (particularly trace metals) must be practically focused on those chemicals whose concentrations have been augmented above those that would be expected to occur naturally. Therefore, the potential for adverse biological effects as indicated by the exceedances of SQGs must be evaluated in conjunction with other information such as the natural background concentrations of substances (see Chapter 2). In some management scenarios, it may also be necessary to consider concentrations of ubiquitous organic chemicals (i.e., the low level contamination of certain substances that are found throughout many environmental compartments) that are representative of reference or “clean” sites.

GUIDING PRINCIPLES

The following guiding principles for the development of Canadian SQGs for the protection of aquatic life are based on those adopted by the Canadian Council of Ministers of the Environment (CCME 1991a) for the development of Canadian water quality guidelines.

- SQGs are numerical concentrations or narrative statements that are set with the intention to protect all

forms of aquatic life and all aspects of their aquatic life cycles during an indefinite period of exposure to substances associated with bed sediments.

- In deriving SQGs for the protection of aquatic life, all components of the aquatic ecosystem (e.g., bacteria, algae, macrophytes, invertebrates, fish) are considered, if the data are available. However, evaluation of the available data should focus on ecologically relevant species.

- Interim SQGs (ISQGs) are derived when data are available but limited, and information gaps are explicitly outlined.

- Unless otherwise specified, SQGs refer to the total concentration of the substance in surficial sediments (i.e., the upper few centimetres) on a dry weight basis (e.g., $mg \cdot kg^{-1}$ dry weight). However, sediments represent a complex and dynamic matrix of biotic and abiotic components that may influence the bioavailability of sediment-associated chemicals. When sufficient information is available to define the influence of any factor on the toxicity of a specific substance (e.g., TOC for nonpolar organic substances) (Swartz et al. 1990; Di Toro et al. 1991), the guidelines will be developed to reflect this relationship. Consideration of these relationships will increase the applicability of guidelines to a wide variety of sediments throughout Canada.

- SQGs are refined as new and relevant scientific data become available. The refinement of these guidelines in the longer term will provide a means of ensuring their broader applicability.

OVERVIEW OF GUIDELINE DEVELOPMENT

The process for developing Canadian SQGs follows the general framework that has been established for the derivation of water quality guidelines (CCME 1991a). This process involves a comprehensive evaluation of the available scientific information on a particular substance to support the development of national SQGs (Figure 1). This section gives a brief overview of this process, with details of the guideline derivation procedures described in the next section.

Literature Search and Evaluation

A comprehensive review of the scientific literature is performed for each substance for which guidelines are to be derived. Introductory information is briefly summarized on the substance's physical and chemical properties, its production and uses, and its sources to the aquatic environment. More detailed discussions are included with respect to concentrations of the chemical found in Canadian sediments (including natural background concentrations), the chemistry and fate of the chemical in sediments, and the available toxicological data for the sediment-associated chemical. Each toxicological study retrieved from the scientific literature is evaluated for its overall acceptability to ensure that high quality data are used in developing SQGs. Characteristics of the sediment and the overlying water column are reviewed if these factors have been measured, since this information is used to help interpret the corresponding toxicological data. Finally, a review of existing guidelines from other jurisdictions is provided. Data gaps are explicitly outlined to stimulate research that will generate the necessary data to support guideline development.

Background Concentrations of Natural Substances

Regional background concentrations of natural inorganic and organic substances occurring in sediments are also established, if possible, for freshwater and marine systems. This information should be considered during the implementation of SQGs since, in some cases, national guidelines (which are toxicologically based) may be lower than the respective concentrations of naturally occurring substances at a particular site (see Chapter 2 for general guidance on the use of SQGs with information on background concentrations). This information is also an important component in the development of site-specific sediment quality objectives.

An interpretive tool has been developed that provides an effective means of distinguishing the probable origin (i.e., natural vs. anthropogenic) of many metals in marine sediments (Schropp and Windom 1988; Schropp et al. 1989; Loring 1990, 1991; Schropp et al. 1990; Loring and Rantala 1992; MacDonald 1993). This method involves determining the ratio of measured trace element concentrations to that of a reference element at a number of uncontaminated sites (such ratios are relatively constant in the earth's crust). Although normalizations to a reference element can be accomplished using a number of naturally occurring elements (e.g., aluminum, iron, lithium), lithium appears to be the most appropriate for redox positive sediments in marine systems in eastern Canada (Loring 1990, 1991).

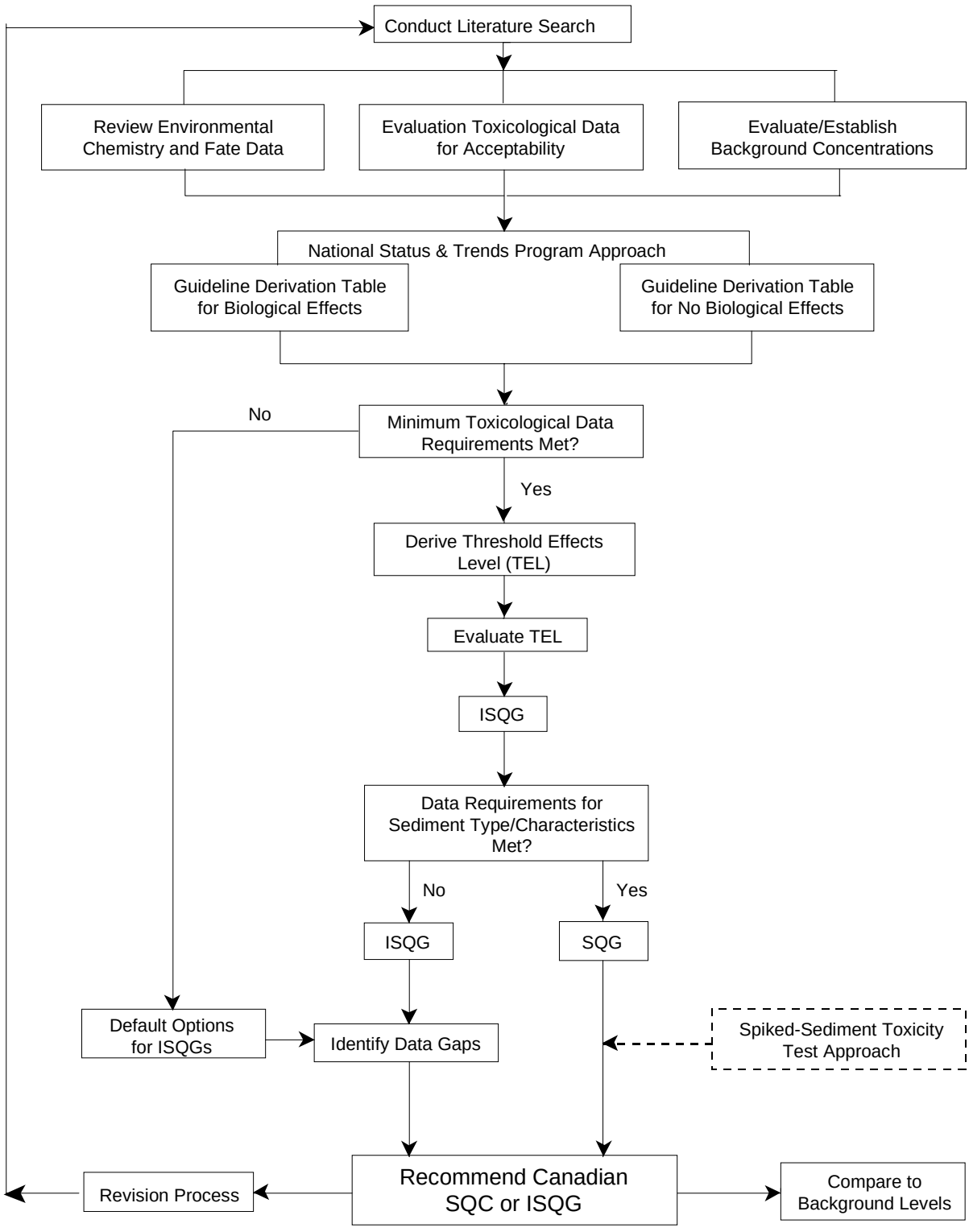

Figure 1. Derivation of Canadian sediment quality guidelines.

Data from a number of uncontaminated sites are used to develop correlations between log transformed concentrations of various trace elements and those of the reference element. Relationships between trace elements and lithium are typically of the form $y = ax + b$ (i.e., linear) (Loring and Rantala 1992). The constant b is significant when the measured concentration of the trace element is high in coarse materials that are virtually devoid of lithium (e.g., sand). A simple linear regression is performed on the data and the 95% confidence limits are calculated. These relationships vary among locations (i.e., both the slope a and the constant b may vary); however, they provide a method for defining reasonable upper limits to the natural occurrence of trace elements in marine sediments at various sites (i.e., anthropogenic enrichment of trace elements is suspected when the trace element to reference element ratio at a site exceeds the upper 95% confidence limits) (Schropp and Windom 1988; Schropp et al. 1989; Loring 1990, 1991; Schropp et al. 1990; Loring and Rantala 1992; MacDonald 1993).

A factor is added to account for the presence of trace elements associated with the organic fraction of the sediments when organic carbon concentrations exceed 3% (Loring and Rantala 1992). The normalized trace element concentration in sediments is then defined as $C_s = a + xC_{Li} + (|y| + y)/2$, where $y = (C_{TOC}) - 0.03$.

This method has been demonstrated only for some marine sediments (Schropp and Windom 1988; Schropp et al. 1989; Loring 1990, 1991; Schropp et al. 1990; Loring and Rantala 1992; MacDonald 1993). Its applicability for distinguishing the probable origin (i.e., natural vs. anthropogenic) of trace elements in freshwater sediments is unknown and should be investigated. Alternative methods for evaluating the origin of some chemicals in freshwater sediments may include the choice of an appropriate reference area that is unaffected by point-source discharges, or the use of the "pre-colonial" sediment horizon (Persaud et al. 1992). Since synthetic organic chemicals are released into the environment only as a result of human activities, equivalent tools for distinguishing their probable origin (i.e., natural vs. anthropogenic) are needed for only the organic substances that occur naturally (such as polycyclic aromatic hydrocarbons). Concentrations of naturally occurring organic substances from essentially uncontaminated sediments at reference sites far from point-source discharges of contaminants could be used as a guide to defining surrogate background concentrations of these substances.

Although the theoretical background concentrations of synthetic organic chemicals released into the environment as a result of human activities are zero, ubiquitous (or low-level) contamination of the environment has occurred. The concentrations of such organic contaminants could be established from reference sites chosen far from point sources of contaminants. Such reference concentrations could be used as a guide to the expected levels of ubiquitous contamination at other sites.

Derivation of Guidelines

Guidelines are derived separately for freshwater and marine sediments using toxicological data compiled for each of these systems. If the minimum data set requirements are met for the NSTP approach, ISQGs are derived using the weight of evidence of available toxicological information. Full SQGs are developed when the ISQGs can be linked to specific sediment types and/or characteristics of either the sediments or of the overlying water column (i.e., if a relationship is demonstrated to exist and is predictable under field conditions). SQGs will also be derived in the future using the SSTT approach, once methodological concerns have been resolved. In the short term, however, this approach will be used to support the weight of evidence of the NSTP approach. Guidelines derived using the above approaches do not specifically address the potential for adverse effects on higher trophic levels of the food chain resulting from the bioaccumulation of persistent toxic substances. However, these issues will be addressed through the use of additional methods (e.g., involving the evaluation of bioaccumulation factors and tissue residue guidelines for the protection of wildlife consumers of aquatic life) before a full sediment quality guideline is recommended. In cases where there is insufficient information to support the derivation of SQGs or ISQGs using this formal protocol, the default process of MacDonald et al. (1992) is recommended to adopt suitable sediment quality assessment values from other jurisdictions for use in the interim.

GUIDELINE DERIVATION PROCEDURES

Specific procedures are described in this section for deriving Canadian SQGs for the protection of aquatic life. These steps include evaluating the quality of available toxicological data and the quantity of acceptable toxicological data (as per the minimum data set requirements), and deriving guidelines using the NSTP approach. Procedures are also described for the SSTT approach, which will support the development of SQGs in the future. The recommendations of MacDonald et al. (1992) are also summarized.

The National Status and Trends Program Approach

The National Status and Trends Program approach involves the compilation of data for many chemicals that are generated from models (equilibrium partitioning theory), SSTTs, and field studies (co-occurrence data consisting of matching sediment chemistry and biological-effect data) (Long and Morgan 1990; Long 1992; MacDonald 1993; Long et al. 1994). This information is used to establish associations between concentrations of chemicals in sediments and adverse biological effects, and thus strongly supports the development of national SQGs based on a weight-of-evidence approach. For a more detailed description of the NSTP approach and supporting rationale for the procedures described below, see Appendix A.

Briefly, acceptable toxicological data that are incorporated into guideline derivation tables (as illustrated in Appendix A) are compiled on a chemical-by-chemical basis and are sorted according to ascending chemical concentrations. (These tables are referred to collectively as the Biological Effects Database for Sediments, or BEDS.) Separate tables are compiled for freshwater and marine sediment toxicity information. Each entry (row) consists of the measured chemical concentration, location, analysis type (or approach), test duration, end point measured, species and life-stage tested, whether associated biological effects or no biological effects were observed, and the study reference. Entries identified by an asterisk (*) in the "Effect/No Effect" column comprise the "effect" data set (i.e., observed biological effects were associated with the measured chemical concentration). Entries for which no effects were observed comprise the no-effect data set and are indicated as NE (no effect [i.e., nontoxic, reference, or control]), NG (no gradient), SG (small gradient), or NC (no concordance). Data on characteristics of the sediment and overlying water column are also summarized, where available. Data are expressed as the total concentrations of the chemical in sediments on a dry weight basis.

The guideline derivation tables compiled for each chemical provide the basis for deriving ISQGs using this weight-of-evidence approach. The data evaluation criteria, minimum toxicological data requirements, and the procedures for deriving guidelines using the information compiled in BEDS are described below.

Evaluation of Toxicological Data

Accurate and precise data on sediment chemistry are essential to understanding the relevance of toxicological test results. Standard sampling methods are also important to ensure that sediment collection, transport, and handling procedures do not affect the results of chemical and biological tests (ASTM 1990a; Loring and Rantala 1992; Environment Canada 1994a). Since the bioavailability of sediment chemicals is linked to the physical and chemical bonding of each chemical with particles in the sample, corresponding analyses of factors (e.g., TOC, particle size) that influence the bioavailability of sediment-associated chemicals are important in order to comprehensively explain observed adverse effects.

In general, toxicological data must be generated using appropriate test methods. For example, methods should include regular cycles of exposure to light and dark test conditions, and verification throughout the test on the condition of the test species. In addition, the analytical chemistry of the test sediment and overlying water should be determined at the beginning and end of the test. Tests should report data on the health of the test species prior to testing. Information on the survival of the test species for at least a week before the test should be available, and test species should not be used where significant mortalities have occurred during this time. Current sediment toxicity tests under development are briefly reviewed in Chapter 2.

With respect to the NSTP approach, candidate toxicological data sets are initially screened for acceptability to ensure that high quality data are incorporated into BEDS and that the guideline derivation tables are internally consistent. The majority of studies that have been screened into BEDS consist of field studies (i.e., co-occurrence data where sediment chemistry and biological-effect data have been measured for the same location at the same time). Studies are designated as acceptable for inclusion in BEDS if the criteria below are met. (See Appendix A for further details.)

- The procedures used for the collection, handling, and storage of saltwater and freshwater sediments should be consistent with standardized protocols (e.g., ASTM 1990a; Loring and Rantala 1992; Environment Canada 1994a). For example, sediments that have been stored for more than two weeks or frozen should not be used in biological tests.

- Data used in co-occurrence analyses must contain matching sediment chemistry and biological-effect data (i.e., data must be collected from the same locations at the same time).

- The concentrations of one or more analyte(s) must vary by at least a factor of 10 at different sampling sites

represented in a single co-occurrence data set (i.e., at a particular location).

- Toxicity tests should employ generally accepted laboratory practices of exposure and environmental controls. Tests that follow standardized guides or protocols (e.g., ASTM 1990b, 1990c; Environment Canada 1992a, 1992b, 1992c) are acceptable. Tests that employ more novel protocols should be evaluated on a case-by-case basis.

- Concentrations of the chemical in sediment must be measured (with the number of measurements taken dependent on the nature of the chemical and duration of the toxicity test). Calculated (nominal) concentrations of the substances in sediments are not acceptable.

- Static, static-renewal, or flow-through aquatic tests may be employed for assessing the toxicity of sediment-associated chemicals. However, acceptable tests should demonstrate that adequate environmental conditions for the test species were maintained throughout the test.

- Preferred end points include effects on embryonic development, early life-stage survival, growth, reproduction, and adult survival. However, other ecologically relevant end points may also be considered with respect to the pathology, behaviour (e.g., avoidance, burrowing), and physiology of the organism.

- Responses and survival of controls must be measured and within acceptable limits, and should be appropriate for the life stage of the species used in the test.

- Appropriate analytical procedures must be used to generate data on the total concentrations of analytes in bulk sediment samples. (Note that, for the purposes of determining background concentrations of natural substances at a site, hydrofluoric acid is used to digest samples before analysis for total metal concentrations [Schropp and Windom 1988; Loring and Rantala 1992]).

- Measurements of abiotic variables should be reported so that any factors that may affect toxicity can be included in the evaluation process. In the overlying water, variables should include pH, dissolved oxygen, total suspended solids, suspended and dissolved organic carbon, and water hardness (and/or alkalinity) or salinity. In the sediment, reported variables should include TOC, particle size distribution, acid volatile sulphide, pH, redox conditions, and sediment type.

- Appropriate statistical procedures should be used and reported in detail.

Data are considered unacceptable for inclusion in BEDS if insufficient information was reported to assess the adequacy of the test design, procedures, and/or results. If any such deficiencies in information can be obtained or clarified through the author(s), the study may eventually be acceptable for inclusion in BEDS. (Note that full SQGs can be derived when the weight of evidence of toxicological information is linked to specific sediment types and/or characteristics of either the sediment or of the overlying water column, although this information [e.g., TOC] is presently not required in order to include a study in BEDS.)

Minimum Data Set Requirements

The use of the NSTP approach for deriving ISQGs relies on the compilation and analysis of all of the available North American data, including data from many locations and on many species that are normally associated with sediments. Minimum toxicological data requirements have been set to ensure that guidelines developed are supported by the weight of evidence linking chemical concentrations to biological effects and that aquatic biota are adequately protected. ISQGs for freshwater or marine sediments can be derived from the studies compiled in BEDS provided that the minimum toxicological data set requirements below are met.

Full SQGs can be derived from these ISQGs if supporting information is available to link the ISQGs with specific sediment types and/or characteristics of the sediment or of the overlying water column (i.e., a weight of evidence must clearly define the relationships of these factors with adverse biological effects).

Table 1. Minimum Toxicological Data Set Requirements for Interim Sediment Quality Guidelines (using the NSTP approach).

• The effect data set for the chemical under consideration must contain at least twenty (20) entries in the guideline derivation table prepared from BEDS.
• The no-effect data set for the chemical under consideration must contain at least twenty (20) entries in the guideline derivation table prepared from BEDS.

Derivation of Guidelines

If the minimum data set requirements for the NSTP approach are met, the derivation of ISQGs can proceed. For each chemical, a functional threshold effect level (TEL) is calculated as the square root of the product (i.e., the geometric mean) of the lower 15th percentile concentration of the effect data set (the E_{15}) and the 50th percentile concentration of the no-effect data set (the NE_{50}). This TEL is calculated to consistently determine a range of sediment chemical concentrations that is dominated by no-effect data entries (i.e., adverse biological effects are never or almost never observed below the TEL). All relevant information on the behaviour of the chemical in sediments and the available toxicological information are evaluated in order to recommend the TEL as the ISQG. If the uncertainty associated with the TEL is high (as indicated after an evaluation of the information in the guideline derivation tables), a safety factor may be applied to the TEL (the rationale for choosing a safety factor and the uncertainties that may be considered are provided in Appendix C). Otherwise, the TEL is considered representative of the concentration below which adverse effects are not anticipated.

To derive full SQGs, defensible relationships between specific sediment (e.g., TOC, particle size distribution) and/or overlying water column characteristics (e.g., pH) and observed sediment toxicity must be supported by a weight of evidence of the available information. Currently, the available information indicates that only ISQGs can be developed. However, some of these data gaps should be filled as information becomes available to support the SSTT approach.

The Spiked-Sediment Toxicity Test Approach

The spiked-sediment toxicity test (SSTT) approach is a complementary procedure that will be used in the future to confirm and strengthen guidelines developed using the NSTP approach. The SSTT approach uses information on the responses of test organisms to specific sediment-associated chemicals under controlled laboratory conditions (Chapman and Long 1983; Ingersoll 1991; Lamberson and Swartz 1992). Sediments are spiked with known concentrations of chemicals, either alone or in combination, to establish definitive cause-and-effect relationships between chemicals and biological responses. At the end of the test period, the response of the test organism is examined in relation to a biological end point (e.g., mortality, reproduction, growth). As in the development of water quality guidelines in Canada (CCREM 1987) or water quality criteria in the United States (USEPA 1985, 1986), acute and chronic effect data generated from sediment toxicity tests can be used to identify concentrations of chemicals in sediment below which aquatic life would not be adversely affected. For further background on this approach and supporting rationale for the procedures described below, see Appendix B. Recommendations for minimum toxicological data requirements and procedures for deriving SQGs using the SSTT approach are described below. SQGs will be developed using this approach once methodological concerns have been resolved.

Minimum Data Set Requirements

Minimum toxicological data requirements have been set to ensure that guidelines developed using spiked-sediment toxicity information provide adequate protection to aquatic organisms. Guidelines can be derived from studies conducted on sensitive species that are not required in the minimum data set (e.g., fish, aquatic plants, protozoa, fungi, bacteria) provided that the following minimum data set requirements are met.

Table 2. Minimum Data Set Requirements for Marine Sediment Quality Guidelines (using the SSTT approach).

- At least four (4) studies are required on two (2) or more sediment-resident invertebrate species that occur in North American waters. At least one (1) of these must be a benthic amphipod species.
- At least two (2) of these studies must be partial or full life-cycle tests that consider ecologically relevant end points (e.g., growth, reproduction, developmental effects).

Table 3. Minimum Data Set Requirements for Freshwater Sediment Quality Guidelines (using the SSTT approach).

- At least four (4) studies are required on two (2) or more sediment-resident invertebrate species that occur in North American waters. These must include at least one (1) benthic crustacean species and one (1) benthic arthropod species (other than a crustacean).
- At least two (2) of these studies must be partial or full life-cycle tests that consider ecologically relevant end points (e.g., growth, reproduction, developmental effects).

Derivation of Guidelines

If the minimum data set requirements are met for the SSTT approach, the derivation of SQGs can proceed. For each chemical, SQGs are derived preferentially from the lowest-observed-effect level (LOEL) from a chronic study using a nonlethal end point. The most sensitive LOEL is multiplied by an appropriate safety factor to derive the SQG. In situations where an acute study for another species is most sensitive, the SQG is derived by multiplying the most sensitive short-term median lethal (LC_{50}) or median effective (EC_{50}) concentration by an appropriate safety factor (to convert it to a long-term no-effect concentration). Rationale for the choice of these safety factors (which are evaluated on a case-by-case basis and may consider a number of uncertainties) is provided in Appendix C.

Species not required in the minimum data set may be used to calculate SQGs provided that the life stage under investigation is aquatic and the minimum data set requirements are satisfied. Each study selected for the derivation of an SQG must have a demonstrated dose–response relationship and the LOEL must be statistically significant.

Interim Sediment Quality Guidelines Adopted from Other Jurisdictions

Sediment quality assessment values (e.g., guidelines, objectives, standards) from other jurisdictions should be evaluated for the chemicals for which insufficient information exists to proceed with the derivation of guidelines using the formal protocol. A sediment quality assessment value should be adopted as an ISQG (using the default process of MacDonald et al. 1992) until data requirements can be met for the formal protocol. An assessment value from another jurisdiction may be adopted (after comprehensive review) if it is scientifically defensible and the derivation process upon which it is based is consistent with the CCME guiding principles. As recommended by MacDonald et al. (1992), the three-tiered system outlined below (modified from Beak Consultants 1988) gives preference to biological-effect based values.

1. Select the lowest of the guidelines that incorporates data on effects of sediment-associated chemicals on sediment-dwelling organisms (i.e., the apparent effect threshold, screening level concentration, sediment quality triad, or NSTP approach), if any of these have been or can be calculated.

2. For organic contaminants, select the lower value obtained using the equilibrium partitioning and the water quality guideline approaches (for which a suitable Canadian water quality guideline exists) if no effect-based guidelines are available.

3. With respect to the development of site-specific sediment quality objectives, select the upper background limit of a trace element if an interim guideline cannot be developed using either of the above procedures or if it is below the upper background concentration for trace elements. (Although the theoretical background concentrations of synthetic organic chemicals released into the environment as a result of human activities are zero, it may be necessary to estimate ubiquitous levels of these substances from reference or "clean" sites located far from point sources of contaminants).

The underlying rationale behind these recommendations is that the effect-based approaches rely directly on observed adverse biological effects of sediment-associated chemicals. They are thought to be most ecologically relevant and scientifically defensible. Guidelines derived using the partitioning methods are based only indirectly on biological effects (i.e., only to the extent that the water quality guidelines used in their calculations are effect-based). Data gaps are explicitly outlined in order to stimulate research that will generate the data necessary to develop SQGs using the formal protocol.

RECOMMENDATION OF CANADIAN SEDIMENT QUALITY GUIDELINES

A Canadian SQG is recommended when the ISQG derived using the NSTP approach is supported by a weight of evidence of the available information that links the ISQG to specific sediment types and/or characteristics of either the sediments or of the overlying water column. A Canadian ISQG is recommended when the ISQG derived using the NSTP approach is based on the available toxicological information only (i.e., the minimum data requirements have been met for an interim guideline, but the interim guideline cannot be linked to specific sediment types and/or characteristics as is required for a full guideline). With respect to ISQGs, data gaps are explicitly outlined in each guideline document to encourage the scientific community to generate the required information (e.g., information linking sediment toxicity to sediment geochemical characteristics). SQGs will also be developed using the SSTT approach when methodological questions are resolved.

All of the available scientific information is evaluated in order to support the recommendation of a Canadian SQG or ISQG. The weight of evidence of toxicological data compiled in the guideline derivation tables for each chemical should support the assumption that the potential for adverse biological effects of a substance increases with increasing concentrations of the chemical. This evaluation is facilitated by the derivation of a second sediment quality assessment value (i.e., the probable effect level, or PEL, above which adverse biological effects are usually or always observed). This value, in conjunction with the ISQG, is used to identify ranges in chemical concentrations associated with adverse biological effects and the incidence of adverse effects within each of these concentration ranges. (See Appendix A for more detail.) This evaluation provides an indication of the reliability (i.e., degree of confidence) in the guidelines established, and provides a means of estimating the probability of observing similar adverse effects at sites with sediment chemical concentrations that fall within the defined concentration ranges.

Contaminated benthos can introduce sediment contaminants into the aquatic food web through predation by organisms at higher trophic levels; therefore, the potential exists for these substances to be transferred through the food web (Lee 1992). Because they are formulated from toxicological information only, the SQGs and ISQGs derived using the NSTP approach and the SSTT approach do not specifically address the potential for adverse effects on higher trophic levels of the food chain as a result of the bioaccumulation of persistent toxic chemicals. Additional procedures will be used to adequately address these concerns. For example, Canadian tissue residue guidelines for the protection of wildlife consumers of aquatic life can be used, along with an appropriate bioaccumulation factor, to calculate SQGs that would protect higher trophic levels.

For each substance, a report is produced summarizing the information on its behaviour in sediments, the adverse biological effects associated with its presence in sediments, and the rationale for the recommended guideline. These reports are reviewed by scientific experts in the field, various federal departments (including Natural Resources Canada, Environment Canada, the Department of Fisheries and Oceans, Health Canada, Indian and Northern Affairs Canada, Transport Canada, and Public Works Canada), and the provinces through the CCME Task Group on Water Quality Guidelines. Upon approval by the CCME Task Group, the reports are submitted to the CCME Environmental Protection Committee and to the Canadian Council of Ministers of the Environment for national endorsement.

THE ROLE OF SEDIMENT QUALITY GUIDELINES

Sediment quality guidelines for the protection of aquatic life are derived from the available toxicological information on the biological effects of sediment-associated chemicals on aquatic organisms. The resulting guidelines provide scientific benchmarks to be used as a basis for the evaluation, protection, and enhancement of sediment quality. These guidelines can help in setting targets for sediment quality, within broader management strategies, that will sustain aquatic ecosystem health for the long term. As benchmarks, they can help to evaluate the toxicological significance of sediment chemistry data, and thus to identify and focus the cleanup of contaminated sites, to predict the impacts of activities from various sectors (e.g., agriculture, forestry, and mining) on the aquatic environment, and to evaluate the effectiveness of proposed or existing management strategies in protecting the aquatic environment. In addition, they provide a scientific review of the available toxicological information for a chemical that can be used to support the establishment of sediment quality objectives to protect aquatic life, which are developed to reflect a number of site-specific considerations. Whether the national SQG is defined as “full” or “interim,” it is used in a similar fashion. However, the limitations in the information used to derive SQGs and ISQGs should be acknowledged in all applications of the guidelines.

Sediment quality guidelines and the sediment toxicity information compiled for each chemical provide a common scientific basis for the establishment of sediment quality objectives. The objectives are developed to apply directly to a particular site and consider a number of distinct characteristics of that site, including chemical, physical (e.g., natural background concentrations, geochemical characteristics), and biological (e.g., sensitive species) characteristics. The objectives are intended to provide the same level of protection as the SQGs, but take into account such site-specific characteristics. Establishment of sediment quality objectives usually requires agreement between all of the parties responsible for managing the designated uses of the site. Depending on the circumstances at the site, an objective may be equal to the national SQG, greater than the SQG (i.e., less stringent), or less than the SQG (i.e., more stringent). Sediment quality guidelines and objectives that are recognized in enforceable environmental control laws of one or more levels of government are then defined as standards. A document is currently under development to provide interpretive guidance on the use of national SQGs (C. Gaudet 1994, Environment Canada, Ottawa, pers. com.).

Canadian sediment quality guidelines are developed with the intention to be conservative, since they are to be used on a national scale. Although SQGs are considered to be applicable to a variety of sediment types, they are not intended to define uniform values of sediment quality on a nationwide basis. However, they may be employed as nationally consistent screening tools. There is a significant potential for differences in the bioavailability (and hence the toxicity) of contaminants in different sediment types. However, because of limitations in the type of information currently available on the toxicity of sediment-associated chemicals, it will be difficult to link the guidelines to specific sediment types and/or characteristics associated with either the sediments or the overlying water column. Therefore, during the implementation of these guidelines, caution should be used in interpreting the biological significance of chemical concentrations in sediments of different types or characteristics from those compiled in the guideline derivation tables for each chemical. The potential for observing adverse biological effects, as indicated by the exceedances of SQGs or ISQGs, must be evaluated in conjunction with other information, such as natural background concentrations of substances, various biological tests, and other assessment values (specifically the PEL). The use of SQGs in exclusion of this type of information can lead to erroneous conclusions or predictions about ambient sediment quality conditions. The use of SQGs in conjunction with other information is briefly discussed in Chapter 2. When SQGs are used along with all other relevant information, they support practical and informed decision-making regarding sediment quality. In the future, field validation of these guidelines at a variety of sites across Canada will provide a means of evaluating their predictability in Canada, and will provide a basis for refining the guidelines to increase their applicability in Canada, as necessary.

Chapter 2
The Use of Sediment Quality Guidelines as Benchmarks

INTRODUCTION

Canadians have begun to recognize that developmental activities represent potential hazards to the health and integrity of their aquatic ecosystems. Fisheries closures on shellfish and advisories on finfish consumption in the vicinity of pulp mills in British Columbia provide graphic examples of the potential social and economic impacts that can result from the release of chemicals into the environment (M. Nassichuk 1992, Department Fisheries and Oceans, Vancouver, pers. com.). The conservation and protection of our aquatic resources has become a high priority goal, and management efforts are focusing on reducing inputs of toxic substances into the environment as well as cleaning up contaminated areas to restore the quality of degraded ecosystems (CEPA 1985; Government of Canada 1990).

Sediments are important in influencing the fate of chemicals in aquatic ecosystems and they provide habitat for many benthic and epibenthic organisms. Concerns regarding the protection and management of sediment quality have raised important questions about the toxicological significance of sediment-associated chemicals and their potential to impair the designated uses of aquatic environments. Therefore, SQGs for the protection of aquatic life are needed to provide relevant benchmarks to help address these concerns.

SQGs are only one of the many scientific tools available to help in the protection and management of sediment quality. They can be used to help interpret whether existing or predicted sediment quality conditions pose a threat to benthic organisms. The use of SQGs in exclusion of other information (such as background concentrations of naturally occurring substances, other assessment values such as the PEL, or biological tests) can lead to erroneous conclusions or predictions about sediment quality. Therefore, SQGs and all other relevant information should be considered, to support practical and informed decision-making regarding sediment quality. These considerations are equally important whether the focus is to maintain, protect, or improve sediment quality conditions at a particular site.

This chapter is intended to provide a brief generic example of the use of national SQGs along with other relevant information. The use of SQGs as national benchmarks provides a broadly applicable example for many potential users of these guidelines and does not imply the preclusion of other specific uses that have not been discussed (such as their use with site-specific information and contaminant transport models to help evaluate discharge limits, their use to help focus the cleanup of contaminated sites, or as the scientific basis for developing site-specific objectives). Potential application scenarios of SQGs, as well as guidance on when or how to

proceed with various management options, will be provided in future documents.

The following guidance is given within the context of a general sediment assessment framework, which integrates the types of information that should be considered along with SQGs. Specific management options, such as defining site-specific objectives to maintain and protect sediment quality conditions or defining remediation objectives to improve sediment quality conditions, cannot be appropriately chosen without an understanding of ambient sediment quality. The framework provides environmental managers with a consistent process for using SQGs with other relevant information to assess sediment quality (i.e., to help answer the question of whether existing or predicted concentrations of chemicals in sediment pose a hazard to sediment-associated organisms).

APPLICATION OF SEDIMENT QUALITY GUIDELINES AS BENCHMARKS

Sediment quality guidelines can be used as benchmarks for evaluating sediment chemistry information to identify situations that may be harmful to aquatic organisms associated with bed sediments. They can also be used as benchmarks to help set targets for sediment quality, within broader management strategies, that will sustain aquatic ecosystem health for the long term. The latter use will be discussed more fully in a future document. For sediments of superior quality, "impairment" to the national SQGs should not be advocated.

In using SQGs as benchmarks, adverse biological effects are not predicted when the measured concentrations of sediment-associated chemicals at a site are at or below the national SQGs. (Note that the term site in the context of this chapter is meant to be generic, whether it refers to a region, a basin, a specific site, or a given quantity of sediment.) Further investigation of sediment quality at the site is usually not necessary, but may be warranted under some circumstances (e.g., when sediments at the site have low levels of TOC, when other variables are suspected to be increasing the bioavailability of chemicals, or when SQGs do not exist for particular chemicals that are measured in the sediment). The potential for observing adverse biological effects is recognized when the concentration of one or more sediment-associated chemicals is greater than the national SQG, with the incidence and severity of these effects generally increasing with increasing chemical concentrations (Long et al. 1994). (See also Appendix A.)

A GENERIC EXAMPLE

The context of a general sediment assessment is used in the following sections to illustrate how SQGs function as benchmarks and how they should be used with other complementary information. A general framework providing an example of this is outlined in Figure 2. (Note that this framework is not intended to replace accepted sediment-testing protocols or monitoring programs that have already been established.) Depending on the goals of the assessment, various tools (e.g., SQGs, biological tests) and site-specific information (e.g., background concentrations of naturally occurring substances) can be integrated to achieve the desired goals. The components of this framework are briefly discussed in the following sections.

In a broad context, the initial phase of a sediment quality assessment should involve the identification of sediment quality issues and concerns for the area that are primarily associated with (existing and/or potential) point and nonpoint sources of contaminants. The goal of the assessment also needs to be clearly defined, including its affirmation of the broader environmental management goal (of which sediment quality is only one component) that has been defined for the region. A regional assessment of sediment quality may be required to determine the relative conditions of ambient sediments at a number of sites, with the aim of prioritizing and focusing future activities. In contrast, the assessment of sediment quality at a specific site may need to characterize site conditions more comprehensively in order to implement specific management options (such as the maintenance and/or protection of ambient sediment quality conditions, the development of site-specific objectives, or the potential remediation of an area). Besides being very different in geographical scale, the complexity of regional and site-specific sediment quality assessments may vary depending on the situation. The types of tools and information used in the assessment is ultimately the choice of the environmental manager. The following sections provide a brief discussion on the various assessment tools available and the types of site-specific information that should be considered when assessing sediment quality.

Water and Land Use Activities

The first step in the framework involves reviewing available information on existing and potential land and water use for the site(s) under consideration. Information on past, present, and future industries and businesses in

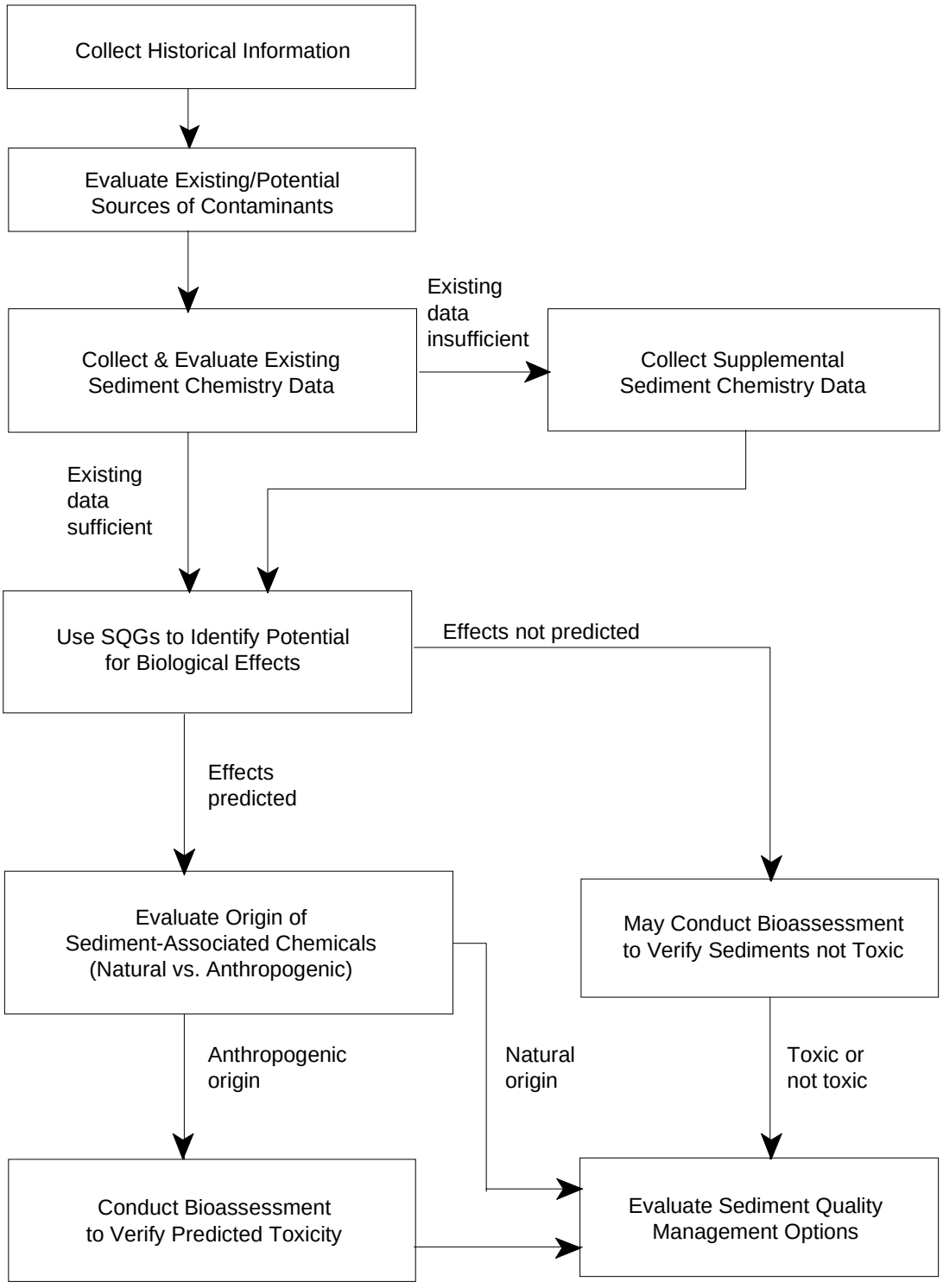

Figure 2. A framework for the assessment of sediment quality.

the area; the location of wastewater treatment plants; land use activities in upland areas; stormwater drainage systems; and residential developments is important. Such information provides a basis for identifying historical, current, and potential sources of contaminants to the aquatic ecosystem. Information on the existing and potential chemical composition of point and nonpoint source discharges of contaminants, on the water quality conditions (in many cases, based on comparisons with water quality guidelines), and on the physical/chemical properties of these substances provides a basis for identifying potential sediment chemistry concerns at the site(s). Subsequently, a list of substances of potential concern can be compiled. This site-specific information on the past, present, and future uses of the site(s) provides a basis for making decisions regarding the nature and extent of the investigations that should be conducted. More detailed descriptions of the type of information that should be collected and how such data can be used to assess ambient sediment quality is reviewed elsewhere (Baudo and Muntau 1990; Mudroch and MacKnight 1991; MacDonald 1993).

Existing Sediment Chemistry Data

All of the available sediment chemistry data for the site(s) under consideration should be assembled to initiate a preliminary assessment of sediment quality conditions. The applicability of these data must be fully evaluated to determine the overall quality of the existing data set, the degree to which the data are thought to represent the current conditions at the site(s), and the degree to which they address the issues and concerns identified. In addition, available information on other physical and chemical characteristics of the sediment (e.g., TOC, particle size distribution) should be compiled in conjunction with the data on chemical concentrations. Such factors may influence the bioavailability (and hence the toxicity) of sediment-associated chemicals, and the site(s) should be assessed to determine the distinct conditions that exist.

An important consideration in the evaluation of the quality of the available sediment chemistry data is the quality assurance/quality control measures that were implemented during collection, transport, and analysis of sediment samples. Conventional practices have recently been established that provide guidance on the field aspects of sediment sampling programs (ASTM 1990a; USEPA and ACE 1991; Mudroch and MacKnight 1991; Environment Canada 1994a). A diversity of analytical procedures have been developed to quantify concentrations of chemicals in sediments, and acceptable methods have been reported (USEPA and ACE 1991). More novel analytical procedures may be evaluated based on the reported accuracy and precision of the technique (i.e., the results of analyses performed on standard reference materials, split samples, or spiked-sediment samples). The detection limits reported must be relevant to assessing the potential for biological effects at the site. The applicability of the detection limits may be assessed by comparing them to the SQGs recommended for that substance.

The applicability of existing sediment chemistry data should also be evaluated in terms of temporal and spatial variability in sediment quality. The age of the chemistry data is an important consideration. Natural degradative processes (Mosello and Calderoni 1990), meterological events (such as storms) that result in the transport of sediments (Allan 1986), industrial developments, and regulatory activities may alter the sources and composition of chemicals released into the environment. As a result of these processes, historical chemical data may not be representative of existing conditions. The list of variables for which chemical analyses are completed should incorporate knowledge of the potential contaminants from land and water use activities in the area. Since the chemistry of bed sediments may vary spatially (Mah et al. 1989; Long and Morgan 1990; Hakanson 1992), data from a number of stations are required to provide a representative picture of sediment quality conditions in an area. The actual number of stations required will depend on the size of the area under consideration, the concentrations of sediment-associated chemicals, the variability of chemical concentrations, and the overall goal of the assessment.

If existing sediment chemistry data are considered acceptable, the potential for observing adverse biological effects at the site(s) can be assessed. If the sediment chemistry data are considered to be of unacceptable quality or are not considered to adequately represent the site(s), additional sediment chemistry data should be collected.

Supplemental Sediment Chemistry Data

A focused, well-designed field survey must be implemented to collect the types of supplemental sediment chemistry data that will support the goal of the assessment. The initial list of substances of potential concern defined during the evaluation of the land and water use activities provides a defensible means of identifying substances for inclusion in a field survey. The field survey should be designed to delineate temporal and spatial variability in sediment quality and should

explicitly outline the quality assurance/quality control measures that will be implemented. Collection, handling, and storage of sediment samples should follow established protocols (ASTM 1990a; Environment Canada 1994a). Analytical methods and detection limits should be appropriate for the substances under consideration.

The Potential for Adverse Biological Effects

The next step in the framework for assessing sediment quality involves determining the potential for observing adverse biological effects at the site(s) under consideration. Information on concentrations of chemicals in sediments provides essential data for evaluating the nature and spatial extent of sediment chemistry. However, these data alone do not provide a measure of adverse biological effects or an estimate of the potential for such effects. SQGs are used to determine whether sediment-associated chemicals are present at concentrations with the potential to impair the aquatic life at that site(s).

As described in Chapter 1, a weight-of-evidence approach is used to derive national SQGs. A second sediment quality assessment value, the PEL, can also be derived using the guideline derivation tables (MacDonald 1993). The PEL represents the lower limit of the range of chemical concentrations that are usually or always associated with adverse biological effects. The national SQG and the PEL are used to define three ranges of chemical concentrations for a particular chemical, those that are rarely (<SQG), occasionally (between the SQG and the PEL), and frequently (>PEL) associated with adverse biological effects (see Appendix A) (MacDonald 1993; Long et al. 1994). The quantification of the incidence of biological effects within each of these concentration ranges provides a useful tool for estimating the probability of observing similar adverse effects within the defined concentration ranges of particular chemicals. Therefore, the frequency with which and degree to which measured sediment chemical concentrations at a site fall within each of these concentration ranges are useful to distinguish sites and chemicals of little toxicological concern, of potential toxicological concern, or significantly hazardous to exposed organisms.

Sediments with measured chemical concentrations that are equal to or lower than the national SQGs are considered to be of acceptable quality. In general, further investigations of these sediments would be of relatively low priority. Management options at these sites would focus on the protection of existing sediment quality conditions. However, in some cases, biological testing may be required for validation of this conclusion (for example, in sediments with low levels of TOC, when other variables are suspected to be increasing the bioavailability of sediment-associated chemicals, or when SQGs do not exist for particular chemicals that are measured in the sediment).

Sediments with measured chemical concentrations between the national SQG and the PEL are considered to represent potential hazards to exposed organisms. Although adverse biological effects are possible within this range of concentrations, their occurrence, nature, and severity are difficult to reliably predict on an a priori basis. Specific conditions at these sites are likely to control the expression of toxic effects. Further investigations on these sediments are needed to determine whether sediment-associated chemicals represent significant hazards to aquatic organisms. Such investigations may include the determination of background concentrations for naturally occurring substances and/or a suite of biological tests designed to evaluate the toxicological significance of particular chemicals (with respect to key species of aquatic biota and factors at the site that may be influencing the bioavailability of the chemical).

Sediments with measured chemical concentrations equal to or greater than the PEL are considered to represent significant and immediate hazards to exposed organisms. Sediments with concentrations of one or more chemicals that fall within this range of concentrations should be considered to be of the highest priority for appropriate management actions to improve sediment quality and restore the desired level of protection, if necessary. Biological assessment is recommended at these sites to determine the nature and extent of effects that are being manifested as a result of the sediment-associated contaminants.

Background Concentrations of Natural Substances

The determination of background concentrations of naturally occurring substances is important when adverse biological effects have been predicted using SQGs (i.e., measured chemical concentrations at a site are >SQG). Information on background concentrations of natural substances is used to determine the extent to which human activities have contributed to the concentrations of sediment-associated chemicals measured at a site, and is particularly important for metals and certain organic substances that may be enriched through natural processes. Although natural levels of these substances may have an adverse effect on certain organisms, defensible management

options should consider the contribution of natural processes in order to focus on the sites and chemicals that are primarily influenced by human activities.

An interpretive tool has been developed that provides an effective means of distinguishing the probable origin (i.e., natural vs. anthropogenic) of many metals in sediments (Schropp and Windom 1988; Schropp et al. 1989; Loring 1990, 1991; Schropp et al. 1990; Loring and Rantala 1992). This method involves determining the ratio of metal concentrations to those of a reference element. Because such ratios are relatively constant in the earth's crust, they can be used to interpret the degree of anthropogenic enrichment of metals at other locations. This method has been demonstrated only for some marine sediments, however, and its applicability to freshwater sediments is unknown (but should be evaluated).

The development and use of this interpretive tool involves intensive sampling at a number of uncontaminated sites within a region. For example, data on sediment metal concentrations were collected from roughly 100 sites throughout Florida, sites thought to be representative of natural estuarine areas in the state (Schropp and Windom 1988). These data were then used to develop correlations between log-transformed concentrations of various metals and those of a reference element (such as lithium, iron, or aluminum). Simple linear regressions were performed on these data and the 95% confidence limits were calculated. In Florida, significant correlations with aluminum concentrations were obtained for arsenic, cadmium, chromium, copper, lead, nickel, and zinc. These relationships provided the basis for interpreting data on the concentrations of metals in sediments at various sites, such that anthropogenic enrichment of metal levels was suspected when metal-to-aluminum ratios exceeded the upper 95% confidence limits. Subsequent evaluations have confirmed the effectiveness and utility of this interpretive tool at a variety of locations (in the St. Lawrence estuary and the Gulf of the St. Lawrence [Loring 1991], and in the United States [Schropp et al. 1989; Schropp et al. 1990; MacDonald 1993]). Loring (1990, 1991) has established similar correlations between concentrations of lithium and several metals. Lithium appears to be most appropriate for the normalization of metal data from sediments derived from glacial erosion of crystalline rocks, which are common in Canada (Loring 1990, 1991).

Since synthetic organic contaminants are released into the environment only as a result of human activities, equivalent tools for distinguishing their probable origin (i.e., natural vs. anthropogenic) are needed for only those substances (such as polycyclic aromatic hydrocarbons) that have important natural sources. Concentrations of naturally occurring organic substances from essentially uncontaminated sediments at reference sites far from point-source discharges could be used as a guide to defining surrogate background concentrations of these substances. For freshwater sediments, alternative methods for evaluating the origin (i.e., natural vs. anthropogenic) of sediment-associated chemicals may include the choice of an appropriate reference area that is unaffected by point-source discharges or of the "pre-colonial" sediment horizon for determining the background concentrations of natural substances (Persaud et al. 1992).

Biological Assessment

Biological testing is an important component of a sediment quality assessment. The nature and extent of the available information on the effects of sediment-associated chemicals in North America is such that most of the data do not support the establishment of cause-and-effect relationships (most of the data compiled in BEDS support "associative" relationships only, and the quantity of spiked-sediment bioassay data for any given chemical is limited). The relationships between characteristics of the sediment and/or overlying water column (e.g., TOC, pH) and observed adverse effects also need to be defined to the extent that the relative importance of these modifiers of sediment toxicity are predictable under field conditions. Therefore, there is some level of uncertainty associated with predicting toxicological effects in the field (although this uncertainty can be evaluated to a great extent through the calculation of the incidence of effects within defined concentration ranges of chemicals). (See Appendix A.) Biological tests used in conjunction with chemical analyses of sediments can provide definitive information regarding the toxicity of sediment-associated chemicals under a wide variety of circumstances.

Further investigation involving biological testing is recommended to support a sediment quality assessment when the concentrations of one or more chemicals are higher than the national SQGs and established background concentrations of natural substances. Biological testing may also be used to assess the toxicity of sediments that may contain unmeasured chemicals or that have distinct physical characteristics (e.g., low levels of TOC). It may also be necessary to consider sensitive species representative of the site(s). Such studies can be used to assess the applicability of the national SQGs to the site conditions and will contribute site-specific toxicological information that will support the development of sediment

quality objectives for the site. Biological testing should be performed at sites where sediment conditions are considered to be significant and immediate hazards to exposed organisms (i.e., with measured chemical concentrations equal to or greater than the PEL) to determine the nature and extent of effects that are being manifested at these sites.

A number of tests have been developed to evaluate the toxicological significance of sediment contamination. These tests range in complexity from spiked-sediment bioassays (which study a single contaminant and a single species) to microcosm studies (which investigate the long-term effects of chemical mixtures on ecosystem dynamics). In addition, tests may be designed to assess the toxicity of whole sediments (solid phase), suspended sediments, elutriates, sediment extracts, or pore (interstitial) water. The organisms that are routinely tested include microorganisms, algae, aquatic macrophytes, invertebrates, and fish (Schiewe et al. 1985; Burton and Stemmer 1988; ASTM 1990b, 1990c; E.V.S. Consultants 1990; Burton 1991; USEPA and ACE 1991; Phipps et al. 1993). While requirements for biological tests differ among applications, sediment toxicity tests should follow established or approved methods (such as those established by provincial, federal, or national agencies). Such methods may be modified to assess the toxicity to resident species, toxicity over longer periods (i.e., to address chronic toxicity), or for different end points. However, the basic principles of accepted protocols should be followed.

Whole-sediment bioassays are the most relevant for assessing the effects of chemicals associated with bottom sediments. Environment Canada (1992a) has developed 10-day toxicity tests using six Canadian species of sediment-burrowing amphipods *(Amphiporeia virginiana*, *Corophium volutator*, *Eohaustorius estuarius*, *Foxiphalus xiximeus*, *Leptocheirus pinguis*, and *Rhepoxynius abronius)*. Likewise, the American Society for Testing and Materials (ASTM 1990b) has developed and approved four tests for assessing the toxicity of marine and estuarine sediments to four species of amphipod. These bioassays may be modified to assess toxicity to other benthic invertebrate species that occur in estuarine and marine environments, including other amphipods, other crustaceans, polychaetes, and bivalves (ASTM 1990b). A 20-day sublethal test for polychaetes and a 10-day sublethal test for mussels in whole sediment are also under development by Environment Canada (J. Osborne 1993, Office of Waste Management, Environment Canada, Ottawa, pers. com.). The ASTM is also considering procedures for conducting sediment toxicity tests with polychaetes and echinoderms (Ingersoll 1991). Similar techniques have also been developed to assess the toxicity of sediment-associated chemicals in freshwater (ASTM 1990c; Burton 1992; Burton et al. 1992). Environment Canada (R. Scroggins 1993, Technology Development Branch, Environment Canada, Ottawa. pers. com.) is developing growth inhibition/ survival tests for freshwater amphipods *(Hyalella azteca)*, chironomids *(Chironomus tentans / C. riparius)*, and mayflies (*Hexagenia* sp.).

In addition to whole-sediment toxicity tests, various procedures are available for assessing the potential for adverse effects on aquatic organisms due to the re-suspension of sediments or the partitioning of chemicals into the water column. The bacterial luminescence or Microtox® test is frequently used (Burton and Stemmer 1988; Environment Canada 1992b). Tests using algae, invertebrates, and fish have also been employed to assess the toxicity of the suspended and/or aqueous phases. Environment Canada (1992c) provides guidance on the use of an echinoderm fertilization assay for testing the toxicity of sediment pore water or elutriate and is developing a similar test for oyster larvae. The use of oyster and echinoderm embryos and larvae in sediment toxicity testing of marine sediments is currently under evaluation by ASTM (Ingersoll 1991). In addition, procedures for conducting water column bioassays and bioaccumulation tests have been recommended by the USEPA and ACE (1991) and Lee et al. (1989), and a document on sediment re-suspension testing is under consideration by ASTM.

Other types of biological information may also be used in the sediment quality assessment process. For example, comparison of the diversity and abundance of benthic invertebrate communities at test sites with appropriate reference sites (e.g., sites with similar particle size distributions, TOC) provides a means of assessing the relative toxicity of test sediments (Diaz 1992; La Point and Fairchild 1992; Persaud et al. 1992; Reynoldson and Zarull 1993). Various statistical procedures may be used to help identify the chemicals that are associated with the observed biological effects when adequate sediment chemistry data are available. In addition, spiked-sediment bioassays may be used to establish cause-and-effect relationships for specific substances or mixtures of chemicals (Swartz 1987; Burton 1991; USEPA 1992a). Information on levels of chemicals in aquatic biota may also provide a basis for determining the significance of chemical levels in sediments relative to the protection of the health of wildlife consumers of aquatic organisms.

Management Options

Along with the information obtained through a preliminary sediment quality assessment, management options are evaluated against the sediment quality guidelines or objectives. A number of management options are possible whether the ultimate goal is to maintain, protect, or improve sediment quality. For sites where further investigation is not warranted, appropriate management options should be chosen to protect existing sediment quality at the site. Continued monitoring of sediment quality conditions and evaluation against national SQGs will provide a means of identifying changes in sediment quality that may lead to problems (which can then be addressed proactively). Other possibilities include continued monitoring for the assessment of trends or the development of sediment quality objectives that address distinct characteristics of the site. The identification of priority chemicals and sites of concern during the sediment quality assessment can focus further biological investigations on potential problem areas or identified areas of immediate concern. At sites that are seriously contaminated, some remedial action (including source control measures) may be necessary to achieve environmental management goals. These remedial actions could include removal and treatment of toxic materials, isolation of contaminated sediments, or no action at all (i.e., permitting natural degradative and sedimentation processes to mitigate contaminant effects) (Sullivan and Bixby 1989).

Many other factors may also influence the sediment quality management strategies that are ultimately implemented at a site. These factors include the management goals for the site, the nature and severity of the contamination, the potential for exposure of aquatic organisms, availability and costs of remediation technologies, unique characteristics that should be preserved at the site, public expectations, and other political, social, and economic factors. The integration of such information challenges the environmental manager to formulate an effective management strategy that will address the sediment quality issues and concerns that have been identified.

Appendix A
The National Status and Trends Program Approach

DESCRIPTION

The formal protocol for developing national sediment quality guidelines (SQGs) (Chapter 1)relies primarily on the National Status and Trends Program (NSTP) approach. The NSTP approach was used initially to develop informal guidelines to evaluate coastal sediment chemistry data collected nationwide under the National Status and Trends Program of the National Oceanic and Atmospheric Administration (Long and Morgan 1990). This approach to developing SQGs involves the evaluation and compilation of data from a wide variety of sources to establish associations between concentrations of chemicals in sediments and adverse biological effects (i.e., cause-and-effect relationships cannot be inferred from this data). Matching biological and chemical data are evaluated and compiled into a database from numerous models (equilibrium partitioning theory), spiked-sediment toxicity tests (SSTTs), and field studies (co-occurrence data). This weight of evidence is used to derive an upper and a lower guideline value for each chemical (Long and Morgan 1990; Long 1992; MacDonald 1993; Long et al. 1994). The two guideline values derived are used to define three ranges of chemical concentrations, those that are rarely, occasionally, and frequently associated with adverse biological effects (see Figure A-1). The identification of ranges in chemical concentrations has been recommended in the development of SQGs (USEPA 1992b).

The inherent strength of the NSTP approach is the use of a weight of evidence to support the development of guidelines. The approach can be applied to a wide variety of chemicals, and equally to virtually any sediment type that occurs in freshwater, estuarine, and marine environments. The information is compiled from numerous geographic locations throughout North America, and the data have been generated using many different species and biological end points. Most of the information compiled relies mainly on field-collected data that consider complex mixtures of chemicals (and thus their interactive effects), various sediment types (i.e., with different particle sizes and concentrations of substances), and thus, varying conditions of bioavailability. Therefore, the resultant guidelines are considered to be widely applicable.

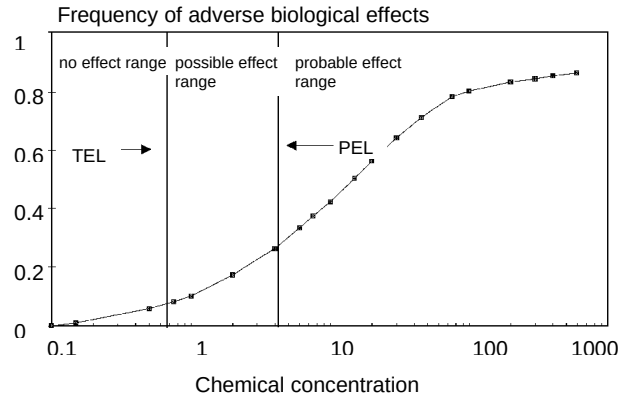

Figure A-1. Conceptual example of effect ranges for a sediment-associated chemical.

The database developed by Long and Morgan (1990) to derive SQGs for marine and estuarine sediments has been updated and expanded. This expansion focused on the inclusion of data from additional sites (including available Canadian data), various biological end points (particularly chronic effects), and information on more chemicals (MacDonald 1993). This expanded database (BEDS) is designed to be updated periodically as new information becomes available. Information is compiled separately for freshwater and marine systems. Toxicological studies incorporated into BEDS include measures of altered benthic communities (e.g., depressed species richness or total abundance), significantly or relatively elevated sediment toxicity (field studies), histopathological disorders in demersal fish (field studies), results of SSTTs (EC_{50} or LC_{50} concentrations), and toxic concentrations predicted by equilibrium partitioning models.

This approach has been extensively reviewed by experts from across North America, has been described orally in numerous technical and scientific forums, and has been described in various publications (Long and Morgan 1990; Long 1992; MacDonald 1993; Long and MacDonald 1992; Long et al. 1994). Guidelines developed using this approach (Long and Morgan 1990; Long et al. 1994) have been used by the National Oceanic and Atmospheric Administration to identify priority regions (ones having the highest probability for observing adverse biological effects) within which surveys have subsequently been implemented to further investigate sediment quality conditions. The guidelines have also been used in assessing hazardous waste sites, dredged material, and monitoring data in the United States (Long and MacDonald 1992). The Florida Department of Environmental Protection has derived sediment quality assessment guidelines on the basis of this approach for use in identifying regional priorities (MacDonald 1993). The California Water Resources Control Board may be using a similar approach to develop sediment quality objectives (Lorenzato et al. 1991). The International Council for Exploration of the Sea (Study Group on the Biological Significance of Contaminants in Marine Sediments) has also elected to adopt this approach in the development of guidelines for participating nations (Long and MacDonald 1992).

DESCRIPTION OF BEDS

Candidate studies were screened for acceptability prior to their inclusion in BEDS to ensure internal consistency in the information compiled and the use of high quality data to support guideline development (Long and Morgan 1990; Long 1992; MacDonald 1993; Long et al. 1994). Each study, and the data sets generated from them, was evaluated to determine the acceptability of the experimental design, the test protocols, the analytical methods, and the statistical procedures used (see the screening criteria outlined in Chapter 1). Studies were included in BEDS only if they contained matching sediment chemistry and biological effect data and if there was at least a 10-fold difference in the concentrations of at least one of the chemicals measured among sampling sites (this criterion being included to maximize the likelihood that observed differences in biological responses between locations were, at least in part, associated with measured chemical concentrations). Data were expressed as the total concentrations of chemicals in sediment samples on a dry weight basis. Information on the factors that influence bioavailability (e.g., TOC, grain size, acid volatile sulphide) was summarized when available. The sediment quality assessment values that have been derived by other jurisdictions (e.g., equilibrium partitioning values in the United States) were either incorporated directly into BEDS (if the concentrations of chemicals were originally expressed on a dry weight basis) or converted to concentrations expressed on a dry weight basis at 1% TOC (if the assessment values were originally expressed on a TOC basis). Conversion of chemical levels to dry weight concentrations at 1% TOC was considered to provide relatively conservative assessment values for entry into BEDS.

Each record in BEDS includes information on the location of the study, the concentration of the chemicals, the biological response observed, the test duration, the species tested or benthic community assessed, the approach used, the particle size distribution, and the factors that could affect the bioavailability of the chemicals. The

information on individual chemicals in BEDS was sorted and guideline derivation tables were prepared for each chemical according to ascending chemical concentrations. Individual entries in the data tables are recorded by rows, with specific information recorded by columns. (See the example provided in Table A-1.) (Note that a single co-occurrence study can provide several data entries for a particular chemical or number of chemicals, depending on the number of biological end points and chemicals measured in the study.)

Information is also included in the guideline derivation tables as to whether the predicted or observed biological response was associated with a gradient in the concentrations of the chemical (see the "Effect/No Effect" column in Table A-1). An entry was assigned an asterisk (*) if an adverse biological effect was reported and concordance was apparent between the observed biological response and the measured chemical concentration (Long and Morgan 1990; Long 1992; MacDonald 1993; Long et al. 1994). Sediment quality assessment values reported for other jurisdictions, such as apparent effect thresholds and screening level concentrations, were entered directly into the tables. Data obtained from SSTTs were also included. Concentrations derived based on equilibrium partitioning theory were also assigned an asterisk. For each chemical, all of the entries designated by an asterisk are collectively referred to as the effect data set.

Concentrations associated with nontoxic, reference, or control conditions were described as having no observed effect (NE). The data in which there was little or no concordance between the chemical concentrations in sediment and the observed biological effect were noted as having no concordance (NC), no gradient (NG), or a small gradient (SG). For each chemical, all of these entries are collectively referred to as the no-effect data set.

Data generated from individual field studies were evaluated in co-occurrence analyses with either of two methods (Long and Morgan 1990; Long 1992; MacDonald 1993; Long et al. 1994). If the statistical significance of the data was reported, the mean chemical concentrations in the statistical group (i.e., toxic versus nontoxic) were compared. If no such statistical evaluations were reported, the frequency distributions of the biological data were examined across all of the sites in the study and mean concentrations in subjectively determined groups were compared (e.g., sites were grouped as most toxic versus least toxic). The results of the co-occurrence analyses were only included in the effect data set if concordance between the measured concentration of the chemical and the observed biological response was apparent. Concentrations of individual chemicals are considered to be associated with the observed toxic response if the mean concentration at sites at which significant adverse effects were observed was a factor of 2 or more greater than the mean concentration at sites at which effects were not observed (i.e., at toxic versus nontoxic sites). These entries contribute to the effect data set. When the mean chemical concentration differed by less than a factor of 2 between the toxic and nontoxic groups, it was assumed that other factors (whether measured or not) were more important in the etiology of the observed effect than the concentration of the chemical under consideration. In these cases, entries are designated as no gradient, a small gradient, or no concordance, and are considered part of the no-effect data set.

DEFINITION OF RANGES IN CHEMICAL CONCENTRATIONS

Two slightly different methods have been reported to derive guidelines using the information evaluated and compiled for each chemical in the guideline derivation tables. Long and Morgan (1990) derived guidelines on the basis of the effect data set only. Guidelines were calculated as the lower 10th percentile or the effects range low (ERL) and the 50th percentile (median) or the effects range median (ERM). The ERL represents a lower threshold value above which adverse effects on sensitive life stages and/or species are expected to begin, and the ERM represents a threshold value above which adverse effects on most species are frequently or always observed. The calculation of percentiles of the data tends to minimize the influence of single (potentially outlier) data points on the development of guidelines (Klaplow and Lewis 1979).

The method used by MacDonald (1993) to derive guidelines considered both the effect data set and the no-effect data set (this is the procedure adopted and described in Chapter 1). Guidelines were calculated as the threshold effect level (TEL) and the probable effect level (PEL) for each chemical. The TEL was calculated as the square root of the product (i.e., the geometric mean) of the lower 15th percentile concentration of the effect data set and the 50th percentile concentration of the no-effect data set. The PEL was calculated as the square root of the product (i.e., the geometric mean) of the 50th percentile concentration of the effect data set and the 85th percentile concentration of the no-effect data set. The TEL represents the upper limit of the range of sediment chemical concentrations that is dominated by no-effect data entries. Within this range concentrations of sediment-associated chemicals are not considered to represent significant

Table A-1. Summary of a portion of the available data on the effects of sediment-associated cadmium ($mg \cdot kg^{-1}$) in marine and estuarine ecosystems.

Cadmium conc. ± SD	Effect/ no effect	Area	Analysis type	Test type	End point measured	Species	Life stage	TOC (%)	Reference
1	NE	San Francisco Bay, CA	COA	10-d	Least toxic (13.6 ± 7.76% mortality)	Amphipod	Adult	1.4 ± 0.79	Chapman et al. 1987
1	NE	San Francisco Bay, CA	COA	10-d	Least toxic (4.63 ± 2.91% avoidance)	Amphipod	Adult	1.44 ± 0.74	Chapman et al. 1987
1	NE	San Francisco Bay, CA	COA	48-h	Least toxic (18+/-8.01% abnormal)	Mussel	Larva	1.2 ± 0.38	Chapman et al. 1987
1	NE	San Francisco Bay, CA	COA	48-h	Least toxic (17.3% mortality)	Mussel	Larva	1.25	Chapman et al. 1987
1	NE	San Francisco Bay, CA	COA	4-wk	Least toxic (116 ± 4.3 young produced)	*Tigriopus californicus* (copepod)	Adult	1.23 ± 0.09	Chapman et al. 1987
1	NE	Burrard Inlet, BC	SQO	—	Sediment quality objective	Aquatic biota	—	—	Swain and Nijman 1991
1	NG	San Francisco Bay, CA	COA	10-d	Moderately toxic (28.3 ± 7.51% mortality	Amphipod	Adult	2.01 ± 0.98	Chapman et al. 1987
1	NG	San Francisco Bay, CA	COA	10-d	Most toxic (95% mortality)	Amphipod	Adult	4.03	Chapman et al. 1987
1	NG	San Francisco Bay, CA	COA	10-d	Highly toxic (37% avoidance)	Amphipod	Adult	4.03	Chapman et al. 1987
1	NG	San Francisco Bay, CA	COA	48-h	Moderately toxic (25.1 ± 6.61% abnormal)	Mussel	Larva	1.26 ± 0.17	Chapman et al. 1987
1	NG	San Francisco Bay, CA	COA	48-h	Highly toxic (66.8% abnormal)	Mussel	Larva	3.59	Chapman et al. 1987
1	NG	San Francisco Bay, CA	COA	48-h	Moderately toxic (57.1% ± 13.6% mortality)	Mussel	Larva	1.14 ± 0.33	Chapman et al. 1987
1	NG	San Francisco Bay, CA	COA	4-wk	Moderately toxic (94.9 ± 10.1 young produced)	*Tigriopus californicus* (copepod)	Adult	2.87 ± 1.07	Chapman et al. 1987
1	SG	Curtis Creek, Baltimore, MD	COA	10-d	Significantly toxic (55% mortality)	*Hyalella azteca* (amphipod)	Juvenile	—	McGee et al. 1993
1.01 ± 1.09	*	Laboratory	SSTT	10-d	LC_{50}	*Rhepoxynius abronius* (amphipod)	—	—	Ott 1986
1.04 ± 1.13	*	Gulf of Mexico	COA	48-h	Significantly toxic (32.6 ± 14.2% abnormality)	*Crassostrea gigas* (oyster)	Larva	0.567 ± 0.153	Chapman et al. 1991
1.04 ± 1.21	NE	Long Island Sound, NY, CT	COA	10-d	Not significantly toxic (23 ± 4.24% mortality)	*Ampelisca abdita* (amphipod)	Subadult	2.46 ± 1.22	Bricker et al. 1993
1.08 ± 1.2	NE	Puget Sound, WA	COA	2-d	Not significantly toxic (6.67 ± 8.07% abnormal development)	*Dendraster excentricus* (echinoderm)	Embryo	1.51 ± 0.33	Pastorok and Becker 1990
1.1 ± 2.0	NC	Southern California	COA	—	Low abundance (57.6 ± 13.6 N/0.1 sq.m.)	Benthic species	—	—	Word and Mearns 1979
1.1	SG	Puget Sound, WA	COA	2-d	Significantly toxic (3.8% abnormal chromosome)	*Dendraster excentricus* (echinoderm)	Embryo	1.5	Pastorok and Becker 1990
1.11 ± 0.355	SG	Long Island Sound, NY, CT	COA	—	—	Microtox *(Photobacterium phosphoreum)*	—	2.51 ± 0.45	Bricker et al. 1993
1.12 ± 0.777	NE	Long Island Sound, NY, CT	COA	48-h	Significantly toxic (EC_{50}; 0.014 ± 0.006 mg dw/mL)	*Mulinia lateralis* (bivalve)	Larva	2.12 ± 1.04	Bricker et al. 1993
1.13 ± 0.867	SG	Long Island Sound, NY, CT	COA	48-h	Significantly toxic (96 ± 1.66% normal development)	*Mulinia lateralis* (bivalve)	Larva	2.52 ± 0.997	Bricker et al. 1993
1.14 ± 0.155	SG	Long Island Sound, NY, CT	COA	48-h	Significantly toxic (68.5 ± 11.4% mortality)	*Mulinia lateralis* (bivalve)	Larva	2.33 ± 0.364	Bricker et al. 1993
1.2 ± 1.0	*	Fraser River Estuary, BC	COA	—	Sediments devoid of feral clams	*Macoma balthica* (bivalve)		1.95	McGreer 1982
1.2	*	San Francisco Bay, CA	AET	10-d	San Francisco Bay AET	*Rhepoxynius abronius* (amphipod)	Adult	—	Long and Morgan 1990
1.2 ± 0.36	NE	Pensacola Harbor and Bay, FL	COA	10-d	Not significantly toxic (9 ± 1.73% mortality)	*Nereis virens* (polychaetes)	Adult	—	EG and G Bionomics 1980

*Adverse biological effect and concordance between observed biological response and measured chemical concentration.

AET = apparent effect threshold
COA = co-occurrence analysis
NC = no concordance
NE = no effect
NG = no gradient
SG = small gradient
SSTT = spiked-sediment toxicity test

hazards to aquatic organisms. The PEL represents the lower limit of the range of chemical concentrations that is usually or always associated with adverse biological effects. The geometric mean is used to account for the uncertainty in the distribution of the data sets (Sokal and Rohlf 1981).

With respect to the methods described by Long and Morgan (1990) and MacDonald (1993), both the lower guideline values (the ERL and the TEL) are assumed to represent the concentration below which toxic effects are rarely or never observed. Both the upper guideline values (the ERM and the PEL) are assumed to represent the concentration above which toxic effects are usually or frequently observed. The range in chemical concentrations between these two guideline values is assumed to represent the range in which effects are occasionally observed. Adverse biological effects are possible within this range of concentrations; however, it is difficult to reliably predict the occurrence, nature, and/or severity of these effects on an a priori basis. Specific conditions at sites with chemical concentrations within this range are likely to control the expression of toxic effects. When chemical concentrations fall within this range, further investigation is recommended to determine whether sediment-associated chemicals represent significant hazards to aquatic organisms (such as determining background concentrations for naturally occurring substances or conducting biological tests to evaluate or confirm the toxicological significance of sediment-associated chemicals to sensitive species of aquatic biota). The conceptual basis for identifying ranges in concentrations assumes that the potential for toxicity increases with increasing concentrations of the chemical (Figure A-1). Because of the variability in the toxicity data available, ranges in chemical concentrations have been defined (rather than absolute values) to provide a more flexible interpretive tool with broad applicability.

The establishment of ranges in chemical concentrations is one of the most attractive features of this approach. The use of two guideline values provides a practical means of further distinguishing the chemicals and the locations that are likely to be associated with adverse biological effects by evaluating the degree and frequency to which measured sediment chemical concentrations at a particular site exceed the TEL and the PEL (or ERL and ERM). This information can then be used in characterizing sites as being of minimal, potential, or significant toxicological concern in order to focus appropriate management strategies at these sites.

The establishment of ranges in chemical concentrations also allows the estimation of the probability of adverse biological effects (Figure A-1). This probability is calculated on the basis of the frequency distributions of the toxicity data for each chemical. Within each of the concentration ranges, the incidence of adverse biological effects is quantified by dividing the number of effect entries (indicated by an asterisk) by the total number of entries, and expressing this ratio as a percentage. The guidelines for cadmium in marine sediments reported by MacDonald (1994; TEL = 0.676 $mg \cdot kg^{-1}$; PEL = 4.21 $mg \cdot kg^{-1}$) illustrate this calculation. In this example, only 5.6% of the cadmium concentrations within the no-effect range (0 to 0.68 $mg \cdot kg^{-1}$) were associated with adverse biological effects (MacDonald 1994). This suggests that there is a low probability of observing adverse effects when cadmium concentrations fall within this range. In the possible- and probable-effect range for cadmium, the incidence of adverse biological effects was 20.1% and 70.8%, respectively. MacDonald (1994) calculated a TEL and PEL for fluoranthene of 0.11 $mg \cdot kg^{-1}$ and 1.49 $mg \cdot kg^{-1}$, respectively. The incidence of adverse biological effects was 9.5%, 20.2%, and 79.7% in the no-effect, possible-effect, and probable-effect ranges. The positive correlation observed between the frequency of effects and chemical concentrations for both cadmium and fluoranthene inspires confidence in the guideline values established for these chemicals. These examples demonstrate how analysis of the distribution of observed biological effects within each of the concentration ranges provides a means of estimating the relative reliability (i.e., degree of certainty) of the guidelines derived. In comparison, for mercury 7.8%, 23.6%, and 36.7% adverse effects were observed in the no-effect, possible-effect, and probable-effect ranges, respectively (the TEL = 0.13 $mg \cdot kg^{-1}$ and the PEL = 0.696 $mg \cdot kg^{-1}$) (MacDonald 1994).

Despite the slight differences in calculating the two guideline values using these methods, the agreement between the ERL values and the TEL values, and between the ERM values and the PEL values, is very good (on average, they vary within a factor of 2) (Long et al. 1994). The reliability (or accuracy) of these guidelines has been assessed through an evaluation of the incidence of effects within each of the ranges (i.e., whether the frequency of observed biological effects increased with increasing chemical concentrations [MacDonald 1993; Long et al. 1994]). The precision of the guidelines was estimated by comparing the ERL and ERM values reported initially by Long and Morgan (1990) to the new ERL and ERM values calculated from the expanded

database (Long and MacDonald 1992; Long et al. 1994;). This evaluation indicated that the ERL and ERM values changed, in general, by a factor of 2 or less, suggesting that the guidelines are fairly insensitive to the addition of a considerable amount of new data once a minimum amount of data has been compiled (the new data incorporated was approximately three times the quantity of the original database).

SUPPORTING RATIONALE

Along with the supporting information described above, the following sections describe the rationale for the specific procedures outlined in the formal protocol (Chapter 1) for deriving SQGs using the NSTP approach. The procedures used in Chapter 1 have been based on the method reported by MacDonald (1993).

Minimum Data Requirements

The specific number of studies required to support the derivation of SQGs using the information compiled in the guideline derivation tables for each chemical was determined from the results of sequential calculations of an interim guideline for a particular substance using data sets of increasing sizes (e.g., from 4 to 60 data points) to determine when the estimate of the guideline stabilized. Using the procedure for deriving a TEL described by MacDonald (1993), an interim guideline was calculated from randomly selected data points (i.e., chemical concentrations; starting with 4 data points selected at random 10 times and each time the interim guideline was calculated). The number of data points was sequentially increased and the interim guideline calculated each time until the estimate of the guideline stabilized. This procedure was conducted using data for cadmium, chromium, fluoranthene, and PCBs. These results indicated that the variability in the estimate of an interim guideline is minimal when 15 to 20 entries from each data set (i.e., the effect data set and the no-effect data set) are used to derive the interim guideline. Therefore, it was concluded that at least 20 entries from each data set were required to support the derivation of SQGs.

To ensure that the guideline calculated is in a concentration range that is associated with an absence of biological effects, both the effect data set and the no-effect data set are considered in the derivation procedure. Variability in the results among individual toxicity assessments may be related to differences in experimental protocols, analytical methods, sediment type used, and a number of other factors. In addition, the results of the various co-occurrence analyses may be affected by the presence of other analytes (measured or unmeasured) in the sediment that co-vary with the substance under consideration, differences in the texture or particle sizes between sites, and a variety of other factors at the site. An examination of the data available for a number of chemicals indicates varying degrees of overlap in chemical concentrations (i.e., unknown factors are influencing observed biological effects) between the effect and no-effect data sets. Therefore, defining a range of chemical concentrations within which adverse effects would not be expected (or rarely expected) is more defensible than establishing an absolute threshold value for no adverse effects (i.e., choosing the lowest effect data point in the guideline derivation table). While it would be desirable to consistently normalize sediment chemical concentrations to factors that have a major influence on chemical bioavailability (e.g., to TOC, grain size), this is not currently possible. Additional information (e.g., TOC, acid volatile sulphide, grain size) is summarized along with the corresponding toxicity data when available. Data gaps will be explicitly outlined to encourage the scientific community to generate the necessary information to support such relationships. As the relationships between sediment types/characteristics and observed sediment toxicity are more clearly defined and supported by the weight of evidence in the available data, guidelines may be developed to reflect these relationships.

Calculation Procedure

While there are many procedures that can be used to evaluate the information contained in the guideline derivation tables (e.g., Long and Morgan 1990; MacDonald 1993), the procedure by MacDonald (1993) was developed to consistently determine a concentration below which adverse effects are not anticipated. The basis for defining a TEL using this approach relies on the establishment of ranges in chemical concentrations that are rarely, occasionally, and frequently associated with adverse biological effects, assuming that the potential for toxicity increases with increasing concentrations of the chemical. Because of the variability in the toxicity data compiled in BEDS, a functional TEL is calculated for each chemical (as described previously) to consistently determine a range of sediment chemical concentrations that is dominated by no-effect data entries (i.e., narratively, adverse biological effects are never or almost never observed). The TEL is calculated as the square root of the product (i.e., the geometric mean) of the lower 15th percentile concentration of the effect data set (E_{15}) and the

50th percentile concentration of the no-effect data set (NE_{50}) to satisfy the narrative definition of this range of concentrations. The geometric mean in this calculation is used to account for the uncertainty in the distribution of the data sets (Sokal and Rohlf 1981).

For example, if there were a total of 100 entries in each of the data sets, then the TEL would fall in a range of concentrations within which there would be, on average, 15 entries (below the E_{15}) from the effect data set and 50 entries (above the NE_{50}) from the no-effect data set. The frequency of biological effects within this range of concentrations would be approximately 15/65, or 23%. The exact incidence of effects would depend on the specific distribution of the data. In the majority of cases, the 50th percentile of the no-effect distribution would be expected to be lower than the 15th percentile of the effect distribution. The incidence of effects between these concentrations would therefore generally be lower than 23%.

The weight of evidence of toxicological data in the guideline derivation tables should also support the assumption that the potential for toxicity of the substance increases with increasing concentrations of the chemical. This is facilitated by the identification of ranges in chemical concentrations associated with adverse biological effects and the calculation of the incidence of adverse biological effects within each of these ranges (as described previously) (MacDonald 1993). The relative degree of certainty in the guidelines derived using this approach is estimated by the degree of positive concordance observed among the frequency of effects and the chemical concentrations. This evaluation provides an indication of the degree of confidence in the guidelines established and provides a means of estimating the probability of observing similar adverse effects at sites with sediment chemical concentrations that fall within these defined concentration ranges.

As a result of the limited information on sediment quality in Canada, particularly for some marine regions, it is uncertain whether the data compiled in the guideline derivation tables are representative of the entire range of sediment quality conditions in Canada. For this reason, care should be exercised in using these guidelines for sites having atypical levels of the factors that influence the bioavailability of chemicals (i.e., outside the range represented in the data tables). In the future, field validation of these guidelines at a variety of sites across Canada will provide a means of confirming their overall applicability in Canada, and will provide a basis for refining the SQGs or interim SQGs as necessary.

Appendix B
The Spiked-Sediment Toxicity Test Approach

DESCRIPTION

The spiked-sediment toxicity test (SSTT) approach is a complementary procedure that will be used in the near future to confirm and strengthen guidelines developed using the National Status and Trends Program (NSTP) approach. Many attributes of the SSTT approach support its incorporation into the formal protocol for developing SQGs for use in Canada. A thorough review of this approach has been provided by Lamberson and Swartz (1992). This method can be used for all classes of chemicals and a wide range of sediment types. The information generated using this approach provides precise dose–response data on specific chemicals, as well as quantitative data on the interactive effects of chemical mixtures. Results obtained from such controlled laboratory tests have a high degree of precision. The approach can also specifically account for the factors that influence the toxicity of chemicals in sediments.

The importance of addressing the bioavailability of sediment-associated chemicals is emphasized by the results of numerous toxicity tests (DeWitt et al. 1988; Nebeker et al. 1989; Swartz et al. 1990; Di Toro et al. 1990; Ankley et al. 1991; Carlson et al. 1991; Di Toro et al. 1991; Di Toro et al. 1992; Ankley et al. 1993). When sufficient information is available to define the relationship of any factor to the toxicity of a specific substance, guidelines can be developed to reflect this. Consideration of these relationships will increase the applicability of the guidelines to a wide variety of sediments.

The various toxicity test procedures that have been developed are generally straightforward and well documented, and dose–response data have been generated for a variety of chemicals (Swartz 1987; Burton 1991; Lamberson and Swartz 1992; MacDonald 1993).

Additional research is required, however, to further refine techniques for conducting SSTTs and to develop standardized methods. Methods of collecting, handling, spiking, and storing sediments must also be considered. Environment Canada (1994a), as well as other agencies/ groups (e.g., ASTM 1990a), is providing such guidance. Environment Canada is currently developing specific guidance on the use of spiked-control sediment toxicity tests in routine laboratory testing in Canada (Environment Canada 1994b). Recommendations of this report will be useful in assessing the adequacy of procedures used in conducting SSTTs.

Many of the attributes of the SSTT approach provide a strong complement to those outlined for the NSTP approach. It is only through the evaluation of dose–response data generated from SSTTs that direct cause–effect linkages between sediment chemical concentrations and biological responses can be developed. The predictive value of the guidelines derived using the SSTT approach should be tested by comparing them with field data on chemical concentrations in natural sediments and observed biological effects (Lamberson and Swartz 1992). Therefore, use of the SSTT approach in conjunction with the NSTP approach (which relies primarily on data generated from field studies) will strengthen the applicability of the guidelines derived.

SUPPORTING RATIONALE

The studies on specific taxonomic groups of aquatic organisms that are required to support the SSTT approach reflect the data that are currently being collected on sediment toxicity to benthic organisms (i.e., the data requirements reflect the availability and use of standardized protocols). The taxonomic groups selected ensure that relatively sensitive sediment-dwelling organisms, like *Rhepoxynius* sp. and *Hyalella* sp., are reflected in the minimum data sets. Two factors that are correlated with the sensitivity of a species to sediment-associated chemicals are its phylogenetic position and its relation to the substrate (Swartz 1987). Amphipods and other crustaceans are generally more sensitive than mollusks and polychaetes, with infaunal organisms appearing to be more sensitive than epifaunal, demersal, or pelagic biota. Data on the effects of sediment-associated chemicals on fish and aquatic plants are not as yet being routinely generated. However, data on these organisms (e.g., Payne et al. 1988) will be considered in the derivation of SQGs when available.

An analysis of the spiked-sediment toxicity information for cadmium, copper, DDT, and fluoranthene provided the basis for choosing the minimum number of studies required to derive SQGs using the SSTT approach. A sufficient number of spiked-sediment studies have been conducted on these chemicals to permit them to be used as test cases in determining the minimum number of studies required to derive guidelines (the quantity of data available for each of these substances ranged from 4 studies for DDT to 19 studies for cadmium). A series of guidelines was derived for each chemical by incrementing the number of studies used in the guideline derivation process by a single study each time until all the studies had been included. The ratio of the highest and lowest guidelines derived for each chemical examined was compared with the number of studies used to calculate these guidelines. This comparison indicated that additional studies above a minimum of four studies no longer significantly affected the numerical value of the SQGs derived (i.e., the calculated SQGs stabilized with a minimum of four studies).

SQGs are derived preferentially from the lowest-observed-effect level (LOEL) from a chronic study using a nonlethal end point. A safety factor, which may account for a number of uncertainties, is applied to this end point in order to calculate the SQG. The rationale for the choice of safety factors is provided in Appendix C.

Appendix C
Derivation of Safety Factors

INTRODUCTION

Safety factors (or uncertainty factors) are commonly used in the development of environmental quality guidelines (CCME 1991a, 1993, 1994). They are unitless numbers that are used to account for various uncertainties that are associated with deriving guidelines (which are intended to represent “safe” chemical concentrations in the field) from limited and incomplete toxicological data sets. The size of the safety factor chosen reflects the types and number of uncertainties that are addressed. Thus, larger safety factors represent a greater degree of uncertainty.

The uncertainties typically addressed in the use of safety factors to derive environmental quality guidelines include the following (H. Vandermeulen 1993, Evaluation and Interpretation Branch, Environment Canada, pers. com.):

1. differences within species due to the age, life-cycle stage, sex, and genetic variability of the organisms

2. differences among species

3. differences in the measured toxicity of a substance due to the sensitivity of the end point measured (e.g., growth versus acute toxicity)

4. the interpretation of end points (i.e., extrapolation from effect concentrations to no-effect concentrations)

5. factors that control the bioavailability (and hence toxicity) of chemicals

6. extrapolation from laboratory to field conditions (which includes possible synergistic or antagonistic responses to various chemical combinations that occur in the field)

The use of safety factors in the derivation of Canadian sediment quality guidelines (SQGs) is intended to achieve a better estimate of the concentrations of sediment-associated chemicals that will not harm aquatic organisms associated with bed sediments over an indefinite period of exposure. The following sections briefly outline the uncertainties addressed and the rationale for choosing a safety factor in deriving national SQGs.

THE NSTP APPROACH

The major uncertainties that safety factors typically address (as listed above) are accounted for in the kinds of data that are compiled to support the National Status and Trends Program (NSTP) approach. The information compiled in the guideline derivation tables (prepared from BEDS) includes a diverse array of data from field, laboratory, and modelling studies. The data have been generated using many different species (including a broad range of fish and invertebrate species covering various life history stages) and biological end points (both effect and no-effect levels are compiled, including data from acute toxicity tests as well as long-term exposure studies such as in situ benthic invertebrate community assessments). Most of the information compiled relies on field-collected data from a wide variety of sites and conditions across North America. Therefore, the data implicitly consider the effects of chemical mixtures and factors that affect the bioavailability of sediment-associated chemicals. Bioavailability is also more directly addressed in some studies where information on characteristics of the sediment and the overlying water column are measured. Direct cause-and-effect relationships, however, cannot be inferred from these data (the data can only establish associations between the concentrations of chemicals in sediments and adverse biological effects).

In deriving ISQGs from the guideline derivation tables, a TEL is calculated to consistently determine a range of sediment chemical concentrations that is dominated by no-effect data entries (i.e., adverse biological effects are rarely or never observed). On average, 23% of the concentrations below the TEL will be associated with adverse biological effects (usually fewer, but this depends on the distribution of the data; see App. A). Bruce and Versteeg (1992) note that, in general, a 20% effect on a population will have little or no impact relative to the natural variation in community and population structure. In addition, Norberg-King (1988) used a 25% effect level as an estimate of the minimal effect concentration. This rationale has also been employed in the calculation of a threshold effect concentration for the derivation of soil quality criteria (CCME 1994). In general, the TEL will be recommended as the ISQG. However, in some cases, an evaluation of all the relevant data, including the behaviour of the chemical in sediments and the available toxicological information compiled in the guideline derivation tables, may indicate the need to apply a safety factor to the TEL (for example, if the narrative definition of the TEL is not supported, that is, the incidence of effects below the TEL is greater than 23%). In these cases, rationale for the safety factor chosen will be provided in the guideline document for each chemical.

THE SSTT APPROACH

In contrast, the majority of uncertainties that safety factors typically address (as listed above) are not accounted for in the kinds of data that are currently being generated from spiked-sediment toxicity tests (SSTTs). However, the information generated using the SSTT approach does provide precise dose–response data on specific chemicals, as well as quantitative data on the factors that influence the toxicity of chemicals in sediments. The differences in sensitivity of various species to sediment-associated chemicals are assumed to be adequately addressed

through the minimum data requirements set out in the formal protocol, considering that amphipods and other crustaceans are generally more sensitive than mollusks and polychaetes, with the infaunal organisms appearing to be more sensitive than the epifaunal, demersal, or pelagic biota (Swartz 1987). The data requirements ensure that relatively sensitive sediment-dwelling organisms (e.g., *Rhepoxynius* sp. and *Hyalella* sp.) are included in the guideline derivation process.

Only limited data are available from SSTTs to determine the margin of safety needed to account for the uncertainties listed above. Uncertainties with respect to interspecies differences as well as factors controlling the bioavailability of chemicals are assumed to be addressed by the SSTT approach itself (as noted previously). As an example of how safety factors may be derived, results of SSTTs on the amphipod *Rhepoxynius abronius* were evaluated for three chemicals (cadmium, zinc, and fluoranthene). This species is used most extensively in SSTTs because standard testing protocols have been established for this organism (ASTM 1990b; Environment Canada 1992a). The rationale for choosing a safety factor that accounts for the uncertainties mentioned previously is outlined below to provide an example of how a safety factor may be chosen for use in the SSTT approach.

Only one study was available to evaluate the influence of intraspecies differences (specifically, life-cycle stage) on the toxicity of sediment-associated cadmium. Robinson et al. (1988) determined the 10-day LC_{50} of cadmium to both adult and juvenile amphipods. The ratio of adult to juvenile LC_{50}s was 1.4. Therefore, a safety factor of 1.4 may be chosen to address the differences in the sensitivities of various life stages of amphipods.

The influence of the end point measured on the toxicity of cadmium to amphipods was evaluated from two studies (Swartz et al. 1985; Mearns et al. 1986). The results of both of these 10-day tests indicated that there was little difference between the LC_{50} end points and the EC_{50}s for emergence and reburial. This difference was slightly greater in the results of 4-day bioassays. The highest EC_{50} to LC_{50} ratio (1.24) was chosen as an adequate safety factor to address this specific uncertainty.

Available spiked-sediment bioassay studies for cadmium and fluoranthene were evaluated to determine an appropriate safety factor for converting an LC_{50} to a NOEL (Swartz et al. 1985; Kemp et al. 1986; Mearns et al. 1986; Robinson et al. 1988; Swartz et al. 1990). In this assessment, the original dose–response data were re-evaluated using the USEPA Probit analysis. As spontaneous response levels of up to 10% are considered to be acceptable for controls in amphipod bioassays, the calculated LC_{10} values from each test were taken as the NOELs. The largest ratio generated by comparing the LC_{50}s to LC_{10}s was 2.3 for fluoranthene (Swartz et al. 1990). This ratio may be chosen as a safety factor to convert median lethal concentrations of sediment-associated chemicals to no-observed-effect concentrations.

Few data were available to convert acutely lethal concentrations to chronic lethal concentrations. Swartz et al. (1985) conducted companion 4-day and 10-day bioassays with adult amphipods in which two end points were considered. The results of these tests indicated that the ratio of the acute to short-term chronic toxicity values for cadmium was on the order of 3.75 (for reburial, the ratio was 3.2). An acute to short-term chronic ratio of 2.22 was calculated for zinc from the results of 3-day and 10-day tests on *Rhepoxynius* (Oakden et al. 1984). The largest ratio, 3.75, may be chosen as the safety factor to convert acutely toxic concentrations to short-term chronically toxic concentrations.

Since little conclusive evidence exists for the possible synergistic or antagonistic toxicity of specific chemical combinations that occur in the field, the margin of safety required to address this uncertainty is difficult to estimate. Therefore, it would be difficult to estimate an appropriate safety factor to address this uncertainty.

Using the information above, two safety factors were derived as examples for use in the SSTT approach. The safety factor used for deriving a guideline from the median lethal concentration of an acute study should incorporate all of the individual safety factors (and corresponding uncertainties) that have been described above. These individual safety factors were multiplied together, resulting in an overall safety factor of 15.0 (or its inverse, 0.067). It would be reasonable to recommend a slightly more conservative safety factor, 20 (or its inverse, 0.05), for use in the SSTT approach, since information on long-term chronic toxicity and the extrapolation from laboratory to field conditions was absent.

Finally, the safety factor used for deriving a guideline from a chronic study should incorporate all of the individual safety factors (and corresponding uncertainties) that have been described above, except for the acute to short-term chronic ratio. These individual safety factors were multiplied together, resulting in an overall safety factor of 4 (or its inverse, 0.25). To estimate differences between laboratory and field conditions, it would be

reasonable to recommend a slightly more conservative safety factor, 5 (or its inverse, 0.2), for use in the SSTT approach due to the paucity of long-term chronic spiked-sediment bioassays (i.e., studies exceeding 10 days) and the absence of information.

The above examples provide the type of rationale that would be required to derive appropriate safety factors for use in the SSTT approach. Since the suggested safety factors are based on limited data for only a few chemicals, they should be reviewed on a chemical-by-chemical basis as additional information becomes available.

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Excerpt from Publication No. 1299; ISBN 1-896997-34-1

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