




CANADA-ONTARIO DECISION-MAKING FRAMEWORK

FOR ASSESSMENT OF GREAT LAKES CONTAMINATED SEDIMENT

Canada Ontario

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Executive Summary

The Canada-Ontario Decision-Making Framework uses an ecosystem approach to sediment assessment and considers potential effects on sediment-dwelling and aquatic organisms, as well as potential for contamination to accumulate in the food chain. It is intended to standardize the decision-making process while also being flexible enough to account for site specific considerations.

In addition to an emphasis on common sense, this framework has four guidance "rules":

1. sediment chemistry data are only to be used alone for remediation decisions when costs of further investigations outweigh costs of remediation and there is agreement to act, or when sites are subject to regulatory action;

2. remediation decisions will be based primarily on biology;

3. lines of evidence (LOE) such as laboratory toxicity tests and models that contradict the results of properly conducted field surveys are clearly incorrect;

4. if the impacts of a remedial alternative will cause more environmental harm than good, then it should not be implemented.

The framework is iterative and sequential in both scope and decision points. Sediments with contaminant concentrations below appropriate sediment quality guidelines (SQGs) that predict toxicity to less than 5% of sediment-dwelling organisms, and which contain no quantifiable concentrations of substances capable of biomagnifying, are excluded from further consideration, as are sediments that do not meet these criteria but whose contaminant concentrations are equal to or below background concentrations. Biomagnification potential is initially addressed by conservative (worst case) modeling based on benthos and sediments, and subsequently by additional food chain data and more realistic assumptions. Toxicity (acute and chronic) and alterations to resident communities are addressed by, respectively, laboratory studies and field observations.

Individual decision points initially comprise relatively simple "yes" or "no" criteria. The integrative decision point for sediments that cannot be so readily assessed, is a weight of evidence (WOE) matrix framework combining up to four main lines of evidence (LOE): chemistry, toxicity, benthic community alteration, and biomagnification potential. Of sixteen possible scenarios, 4 result in definite decisions. Twelve possible scenarios require additional assessment. Typically this framework will be applied to surficial sediments. The possibility that deeper sediments may be uncovered as a result of natural or other processes must also be investigated and may require similar assessment (excluding community alteration since relatively few organisms will be found in sediments below approximately 10 cm depth).

Preface

The governments of Canada and Ontario are committed to the protection of the Great Lakes. They share a joint responsibility to restore and enhance water quality and work together under the formal commitment of the Canada-Ontario Agreement Respecting the Great Lakes Basin Ecosystem to ensure environmental protection. In turn, the Canada-Ontario Agreement helps Canada deliver its commitments with the United States under the Great Lakes Water Quality Agreement (GLWQA). These agreements are available online at www.on.ec.gc.ca/coa and www.on.ec.gc.ca/greatlakes.

Contaminated sediment is a long-standing issue in the Great Lakes and is one factor that contributes to degraded environmental conditions and beneficial use impairments at a number of Great Lakes Areas of Concern (AOCs). Evaluation of the environmental risk posed by contaminated sediment and the development of management options is a major challenge and a harmonized federal-provincial approach to contaminated sediment was needed to avoid inconsistencies in assessments and to provide clarity and transparency in decision-making. To address this issue, the 2002 Canada-Ontario Agreement committed both governments to work together to develop a risk-based decision-making framework for contaminated sediment in the Great Lakes Areas of Concern. This document fulfills that commitment.

The Ontario Ministry of the Environment and Environment Canada originally began the development of sediment guidance for the Great Lakes following an International Joint Commission (IJC) review of the Areas of Concern and sediment contamination (International Joint Commission 1988; see also International Joint Commission, 1999). The Ontario Ministry of the Environment produced two documents: Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario (OMOE, 1993) and An Integrated Approach to the Evaluation and Management of Contaminated Sediments (OMOE, 1996). The Ministry's assessment and management of contaminated sediment involved comparing chemical concentrations in sediment to Ministry Sediment Quality Guidelines (no effect levels, lowest effect levels, and severe effect levels) and natural background levels. If sediment concentrations exceeded one or more of these Sediment Quality Guidelines, additional laboratory or field assessment of contaminated areas was recommended. The Ministry also provided guidance on key considerations for sediment remediation if management action was required.

After the IJC review was released (International Joint Commission 1988), Environment Canada initiated a program to develop biological sediment guidelines using sediment toxicity tests and invertebrate community structure. These biological guidelines for assessing contaminated sediment were completed in 1998 and extensively reviewed by external experts (Reynoldson *et al.*, 1998). The assessment process (**BE**nthic **A**ssessment of **S**edimen**T** (the BEAST) Reynoldson and Day 1998) utilizes benthic invertebrates as these animals are the most exposed and potentially most sensitive to contaminants associated with sediment. Decisions on the spatial extent and severity of contamination are based on the type and number of species present and the response (survival, growth, reproduction) of these animals in standard laboratory tests. The data are compared to the biological guidelines which were developed for both field populations and laboratory responses of benthic invertebrates. The Canadian Council of Ministers of the Environment also developed national Sediment Quality Guidelines based on co-occurrence of chemical and biological data and spiked sediment toxicity test results (if toxicity information was available)(CCME, 2001).

The first of several workshops was held between Ontario and Canada in 1998 to discuss developing an ecologically based sediment decision-making framework. Following the workshop, Environment Canada and the Ontario Ministry of the Environment assembled a team of independent and government scientists who were experts in the fields of sediment geochemistry, toxicity, biomagnification and invertebrate community structure assessment to begin the development of a sediment decision-making framework. In 2002, a COA Sediment Task Group was formed to complete the framework and fulfill the commitment under the COA. To ensure the quality and integrity of the document, Environment Canada and the Ontario Ministry of the Environment conducted extensive targeted consultation and expert review throughout the development of the framework.

The Canada-Ontario Decision-Making Framework for Assessment of Great Lakes Contaminated Sediment provides step-by-step science-based guidance for assessing risks posed by contaminated sediment. The framework is primarily concerned with risks to the environment but considers human health concerns associated with biomagnification of contaminants. It identifies all possible sediment assessment outcomes based on four lines of evidence (sediment chemistry, toxicity to benthic invertebrates, benthic community structure, and the potential for biomagnification) and provides specific direction on next steps in making sediment management decisions. In addition, the framework provides a mechanism for identifying contaminated sediments of greatest concern.

1.0 Introduction

1.1 Background

Contaminated sediment has been identified as one of the major impediments to the restoration of Areas of Concern (AOCs) in the Great Lakes. AOCs comprise locations where the International Joint Commission (IJC) has determined that the aquatic environment is severely degraded.

There is a need for an objective, transparent, pragmatic decision-making framework for contaminated sediments for use in the Great Lakes (and possibly elsewhere). In fact, a sediment decision-making framework for AOCs in the Great Lakes was a commitment made by the federal and provincial governments in the 2002 Canada-Ontario Agreement Respecting the Great Lakes Ecosystem (COA).

The presence of substances in sediments where they would not normally be found, or at concentrations above natural background levels, does not necessarily mean that adverse biological effects are occurring. Other factors, such as the total concentration or the bioavailability of a substance, are more important in assessing if adverse biological effects may occur. This document provides the requisite framework to differentiate between those scenarios where elevated concentrations of contaminants are associated with adverse biological effects and those scenarios where they are not. It is the intention of Environment Canada and the Ontario Ministry of the Environment to use this framework to assess contaminated sediments in the Great Lakes and other waterbodies in the Province of Ontario. An overview of the entire framework is provided in Section 2. The framework is explicitly based on ecological risk assessment (ERA) principles. Sections 3-7 provide additional details of key framework components in the context of the different phases of an ERA. References are provided in Section 8. The Appendices provide two supporting documents: an annotated bibliography (Appendix 1) and a State of the Science Overview and Jurisdictional Scan (Appendix 2).

1.2 Purpose

The purpose of this document is to provide a decision-making framework for contaminated sediments explicitly based on ERA principles, and which also has applications to contaminated sediments in other (freshwater, estuarine and marine) areas. The framework is intended to be sufficiently prescriptive to standardize the decision-making process, but without using a "cook book" assessment approach that would fail to acknowledge the influence of site-specific conditions on the outcome of the decision-making framework, nor allow for appropriate use of best professional judgement. The framework is intended to be:

- objective;
- transparent;
- scientifically rigorous; and,
- readily understandable.

The framework is also intended to be rigid enough, without being inflexible, so that:

- There is consistency between different contaminated sediment assessments;
- Site-specific considerations can be appropriately addressed;
- The localized risks from contaminated sediments are determined;
- The regional risks from contaminated sediments are determined.

Although the basic framework is not expected to change over time, new knowledge is expected to change and improve the tools that comprise the different Lines of Evidence (LOE) within the framework. Accordingly, the best available science should be used in applying the framework. This will require suitable state-of-the-art expertise in the various disciplines comprising the framework.

The decision-making framework is specific for environmental concerns associated with contaminated sediment, including human health concerns related to biomagnification. However, the framework is not otherwise concerned with human health risk assessment (HHRA): it does not address situations where potential human health concerns are associated with dermal contact to contaminated sediment (e.g., swimming, wading), or by other exposure routes (e.g., flooding resulting in aquatic sediments contaminating residential soils or gardens). Nor does it address the issue of unacceptable levels of contaminants that do not biomagnify, such as Cd, Pb, PAHs, in fish or shellfish. In such situations, a screening level HHRA should be considered to assess potential risks and inform the public.

Photo: Environment Canada.

2.0 The Sediment Decision-Making Framework

2.1 Guidance for Implementation

The primary guidance for implementation of this strategy is that it shall be applied within the context of common sense. In other words, it will not be applied inflexibly.

There are four other guidance "rules" for the use of this Framework:

1. Sediment chemistry data (e.g., sediment quality guidelines [SQGs]) will not be used alone for remediation decisions except for two cases. The first case involves "simple contamination where adverse biological effects are likely… when the costs of further investigation outweigh the costs of remediation, and there is agreement to act instead of conducting further investigations." (Wenning and Ingersoll, 2002). This first case is intended to apply to small sites with a limited number of contaminants present at extremely elevated concentrations (e.g., well above predicted effects levels). The second case involves sites subject to regulatory action.

2. Accordingly, any remediation decisions will be based primarily on biology, not chemistry since chemical SQGs are not clean-up numbers by themselves, and need to be used in a risk assessment framework (see Appendix 2, State of the Science Overview and Jurisdictional Scan).

3. LOE (lines of evidence, e.g., laboratory toxicity tests, models) that contradict the results of <u>properly conducted field surveys</u> with appropriate power to detect changes (e.g., see Environment Canada, 2002) "are clearly incorrect" (Suter, 1996) to the extent that other LOE are not indicative of adverse biological effects in the field.

4. If the impacts of a remedial alternative will "cause more environmental harm than leaving the contaminants in place", that alternative should not be implemented (USEPA, 1998).

2.2 Framework

The framework is tiered, and proceeds through the following sequential steps, with corresponding rationale. However, note that different steps do not need to be completed separately; two or more steps can (and in some cases should) be completed jointly (e.g., where this will reduce overall time and costs related to sampling and analysis). For example, if available data are insufficient to rule out management action, sediment toxicity tests may be conducted before chemical analyses are conducted for all chemicals with a SQG. If toxicity tests show that the sediment is not toxic, there would be no reason to measure concentrations of these SQGs.

Thus, the framework is linear in terms of thought processes, but that linearity is not necessarily to be followed in actions such as sample collections or analyses. For example, initial field sampling can involve all possible LOE (e.g., sediments for chemical analyses and toxicity testing; benthos for chemical analyses and taxonomy) with the recognition that, while samples for chemical analyses and taxonomy can be archived, those for toxicity testing cannot be archived and should be tested as soon as possible and no later than 8 weeks following collection (EPA/USACE, 1998).

The framework is conceptually divided into a series of Steps and Decisions that correspond to different ERA tiers. Screening Assessment (for more detail, see Section 3) comprises Steps 1-3 and Decisions 1-2. Preliminary Quantitative Assessment (for more detail, see Section 4) comprises Steps 4-5 and Decisions 3-4. Detailed Quantitative Assessment (for more detail, see Section 5) comprises Steps 6-7 and Decision 5. Step 7 and Decision 6 deal with deeper (than surficial) sediments. The framework is illustrated schematically in its entirety in Figure 1 and in terms of the different ERA tiers at the start of Sections 2.2.1 (Figure 2), 2.2.4 (Figure 3), 2.2.7 (Figure 4), and 2.2.9 (Figure 5). It is described in detail in the sections that follow in terms of the nine individual steps.

As noted by Jaagumagi and Persaud (1996) "Due to the complexity involved in evaluating contaminated sediment, it is essential that scientists with strong expertise in sediment chemistry (chemical fate, transport and speciation), sediment toxicity testing, benthic community assessment, food chain effects and environmental statistics assist stakeholder groups in the interpretation of the data. This is especially important in determining differences or effects of sediment contamination compared to reference conditions."

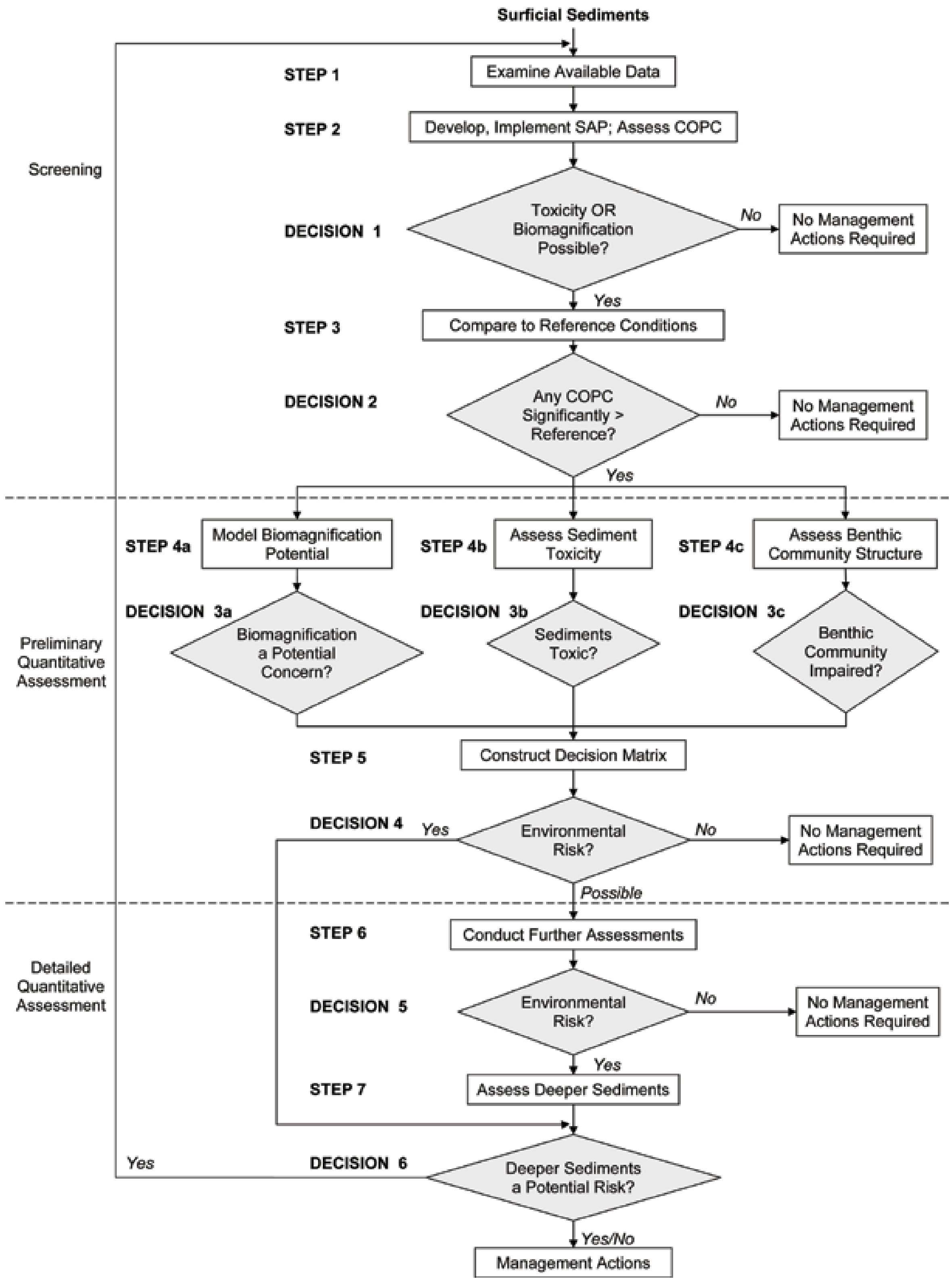


Figure 1. Canada-Ontario Decision-Making Framework For Assessment of Great Lakes Contaminated Sediment. For Explanations of Acronyms, Steps and Decisions, see Text.

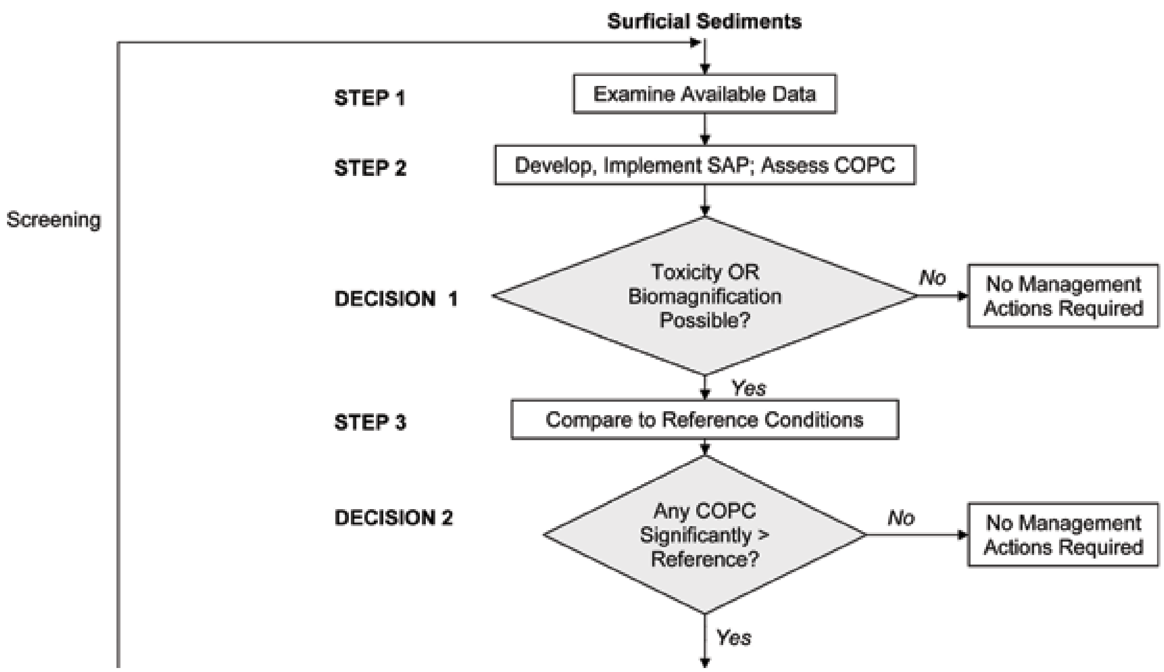


Figure 2. Initial Screening Assessment (Steps 1-3, Decisions 1-2). See also Sections 3.0 and 4.1. Conservative (worst case) assumptions are used to screen out locations and substances that are clearly not of concern and to focus on those that may be of concern.

2.2.1 Step 1: Examine available data

Examine all readily available data for the site (see Section 3.1 re Site Definition), reports and information to determine:

- Contaminants of potential concern (COPC – see Section 3.2) and their concentrations at surface (e.g., < 10 cm) and at depth (e.g., > 10 cm);
- Receptors of potential concern (ROPC – the organisms that may be affected by COPC – see Section 3.3); this information will also assist in selection of toxicity test species (see Section 2.2.5);
- Exposure pathways (by which COPC may reach ROPC);
- Any human health consumption advisories;
- Sediment stability;
- Appropriate assessment endpoints (what is to be protected, e.g., benthos: organisms living in the sediments – see Section 3.4);
- Measures of effect and the level of any effects determined (what is actually measured, e.g., for benthos: species diversity, abundance, dominance – see Section 3.4);
- Appropriate reference areas/locations and their characteristics (see Section 3.5).

Determine whether the site (defined in Section 3.1) has a high level of environmental sensitivity (based on habitat, not land use), and whether contamination is only from off-site sources. A site is defined as the area under investigation which, dependent on size, COPC and other considerations, will generally require multiple samples to assess any environmental impact. Develop an initial Conceptual Site Model (CSM – showing the interrelationships of COPC and ROPC – see Section 3.6), which will be updated as more information becomes available through further investigation.

Information gathered should consider not only surficial sediments (to about 10 cm depth), which are the initial focus, as this is where the majority of sediment-dwelling organisms live, but also deeper sediments and their contamination level and likelihood of being uncovered or even possibly moved such that they could affect surrounding areas. The status of deeper sediments (Step 7, Decision 6) should be considered as data become available.

Rationale: Make use of historic information to appropriately guide subsequent sampling and analyses (which will almost always be required), and to avoid generating new data where data already exist.

2.2.2 Step 2: Develop and implement a sampling and analysis plan

Based on Step 1, above, develop a Sampling and Analysis Plan (SAP – see Section 3.7) for review and approval by stakeholders, then implement same at both exposed and reference sites. The objective of the SAP is to fill in data gaps related to both COPC and ROPC. The SAP should not necessarily be restricted to surficial sediments. A determination is required as to whether there are any COPC in the sediments that could be toxic and/or biomagnify up food chains (increase in concentrations through three or more trophic levels). Common sediment contaminants that may biomagnify include: organic mercury; PCBs; DDT; and, 2,3,7,8-TCDD. If mercury is a COPC, measure both total and methyl mercury concentrations in sediments (mercury only biomagnifies in the methylated form). If PCBs are a concern, measure total PCBs (sum of seven Aroclors: 1016, 1221, 1232, 1242, 1248, 1254, 1260) as sediment quality guidelines are typically based on total PCBs or specific Aroclors[1]. If DDT is a concern, also measure DDD and DDE, its breakdown products.

Decision Point 1: Two questions now need to be addressed (i.e., are COPC levels above SQG-low levels). First, are COPC present in sediments above levels that have been shown to have minimal effects to biota living in the sediments? In other words, could the COPC possibly cause toxic effects? Typically only chemistry data will be available to characterize a site. These data are used in an initial pre-screening step to remove sites from further consideration if concentrations are below appropriate sediment toxicity thresholds. However, occasionally, biomonitoring data may be available for a site that indicates potential adverse effects are occurring. In this situation, the biomonitoring data are sufficient to suggest that additional assessment is needed regardless of the results of the screening step based on chemistry data alone. Second, do COPC present in sediments comprise substances that could biomagnify and affect the health of biological communities at higher trophic levels or of humans consuming biota contaminated with those substances? The first question is addressed by comparing COPC to an appropriate SQG-low (e.g., an SQG that predicts toxicity to less than 5% of the sediment-dwelling fauna, such as the Canadian Threshold Effect Level (TEL) or the Ontario Lowest Effect Level (LEL)). The specific SQG-low that is used for this step may vary based on both regional considerations and best professional judgement. For situations where no SQG exists, compare COPC concentrations to reference areas; sediments where concentrations exceed 20% of reference areas, and are statistically higher than reference areas, suggest anthropogenic exposure has occurred. These substances should be considered as having the potential to cause toxic effects or biomagnify, and further assessment of the sediment is required. The second question is addressed by determining whether or not substances that can biomagnify are present at quantifiable concentrations. Two decisions are possible:

Comparison	Decision
All sediment COPC < SQG-low, **and** no substances present that can biomagnify	No further assessment or remediation required. ***STOP***
One or more sediment COPC > SQG-low, **and/or** one or more substances present that can biomagnify	Potential risk; further assessment required. ***PROCEED TO STEP 3***

1 If a detailed quantitative assessment is conducted, congener specific information may be required for sediments contaminated with PCBs, dioxins and/or furans.

Rationale: Conduct initial analyses as necessary to make a decision as to whether or not the sediments may pose a potential risk to the environment and/or to human health. By design, SQGs are typically conservative, in other words, over-protective. Thus, if sediment COPC concentrations are below SQG that predict minimal effects (SQG-low), there is negligible ecological risk. For example, Porebski *et al.* (1999) found that such SQG performed well as "levels below which unacceptable biological effects were unlikely to occur." Because SQGs have no role in evaluating human health risks or biomagnification (Wenning and Ingersoll, 2002), and there are no such sediment guidelines, initial (conservative) decisions regarding biomagnification potential are simply based on the presence or absence of quantifiable amounts of substances that may biomagnify.

2.2.3 Step 3: Compare to reference conditions - Is there a potential risk based on contaminant concentrations?

Determine whether the concentrations of COPC exceeding SQG-low and/or concentrations of substances that can biomagnify statistically exceed reference concentrations as determined from reference area comparisons.

Decision Point 2: Two separate questions need to be addressed. First, are concentrations of COPC in sediments that are above SQG-low levels statistically different ($p < 0.05$) than reference conditions? Second, are concentrations of COPC that could biomagnify, which are present in sediments at quantifiable levels, not statistically different ($p < 0.05$) than those same COPC in reference areas? Note that in cases where there is little discriminatory power in statistical significance determinations due to very low variability in the reference areas (i.e., a very small difference from reference would be statistically significant but of arguable environmental significance), an additional comparison is possible, specifically: are concentrations of COPC less than 20% above those same COPC in reference areas? The +20% comparison is a straight arithmetic comparison of either mean or individual values, depending on site-specific circumstances (alpha = 0.05; beta = 0.10). Reference conditions include background conditions – either measured or determined from historical data. *Note, in making these comparisons, the data for an immensely contaminated (e.g., > 10 fold the SQGs that predict likelihood of toxicity), but relatively small area, should not necessarily be diluted with data from other, much less contaminated areas.*

Comparison	Decision
[Concentrations of all sediment COPC > SQG-low **and** substances present that can biomagnify] ≤ reference conditions and statistically no different than reference	No further assessment or remediation required. ***STOP***
[Concentrations of one or more sediment COPC > SQG-low **and/or** one or more substances present that can biomagnify] > reference conditions and statistically higher than reference	Potential risk; further assessment required. ***PROCEED TO STEP 4A***

Rationale: In this step, the framework is considering two possibilities: (1) Either all COPC which are greater than SQG low and which can biomagnify are lower than reference (in this case there is no action required because sediment quality reflects background conditions) or (2) there is a difference from reference between one or more COPC (which exceed SQG low) and/or there is a difference from reference between one of more substances that can biomagnify. Inorganic and some organic substances occur naturally and may be naturally enriched in some areas (e.g., naturally mineralized areas, oil seeps). The focus of remediation efforts needs to be on anthropogenic (human) contamination, not natural enrichment. The additional possible determination of a difference of 20% between two sets of chemistry data is well within the bounds of typical analytical variability, may not represent a true (significant) difference because it is likely a consequence of natural sediment heterogeneity (Jaagumagi and Persaud, 1996), and is highly unlikely to be of any environmental concern. The additional use of reference + 20% could be useful to screen out areas of marginal environmental concern, and is the same criterion as used for sediment toxicity test results comparisons (Section 2.2.5).

Photo: Ontario Ministry of the Environment.

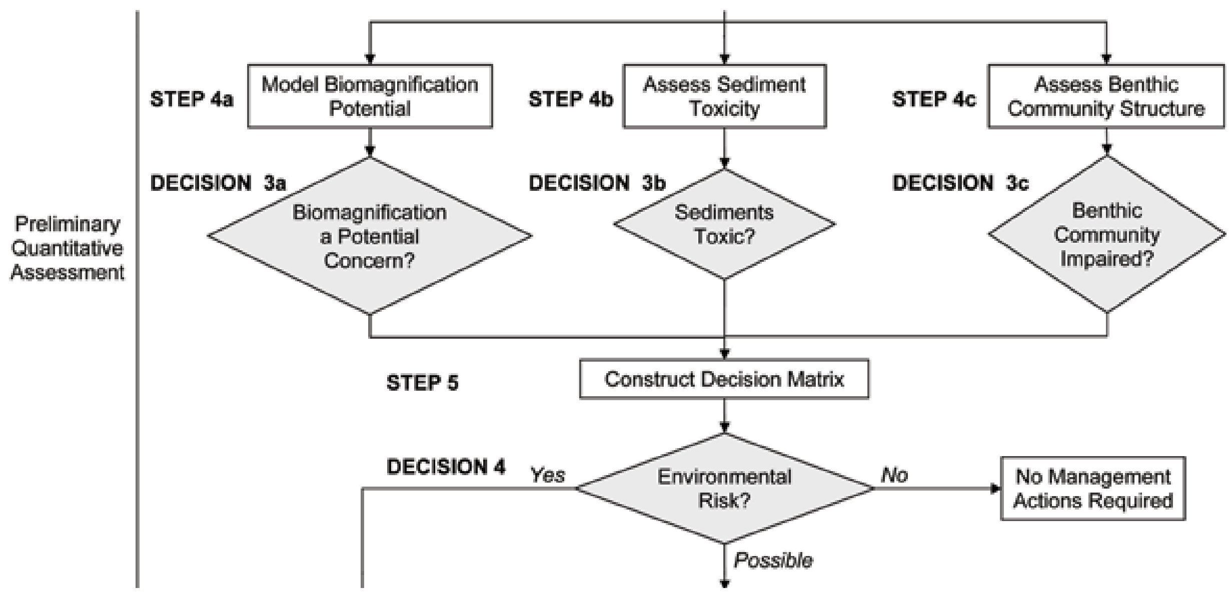


Figure 3. Preliminary Quantitative Assessment (Steps 4-5, Decisions 3-4). See also Sections 5.1, 5.2 and 6.2. Contaminated areas screened in are further investigated, preparatory to determining whether there is or is not a problem, or whether additional investigations are required.

2.2.4 Step 4a: Is biomagnification a potential concern?

If substances that can biomagnify remain of concern, conservatively model concentrations in the sediments, sediment-dwelling organisms, and predators of those organisms through to top predators to determine whether or not there is a potential risk (Grapentine *et al*., 2003a, b – See Section 4.2). Conservative modeling includes, for example: the assumption that maximum contaminant concentrations occur throughout the exposed area; the use of maximum biomagnification factors (BMFs); the assumption that fish feeding is limited to the exposure area. Basically, worst case scenarios, some of which may be unrealistic, are used to allow environmental risks to be either screened out or identified as possibilities to be investigated further.

Decision Point 3a: Determine whether or not contaminant biomagnification is a potential concern.

Comparison	Decision
There is no potential for contaminant biomagnification from the sediments through aquatic food chains	No further assessment or remediation required relative to biomagnification. ***PROCEED TO STEP 4B***
There is potential for contaminant biomagnification from the sediments through aquatic food chains	Potential risk; further assessment of biomagnification potential required. ***PROCEED TO STEP 4B***

Rationale: Conservative assumptions inherent in such a modeling exercise (i.e., worst case assumptions) will allow a determination either that biomagnification is not a concern, or that it may be a concern. In the latter case, additional site-specific assessment may be required (Step 6).

2.2.5 Step 4b: Are the sediments toxic?

For the remaining COPC, use SQG-low and SQG-high (that predict toxicity to 50% or more of the sediment infauna) to map spatial patterns of contamination. Determine the toxicity of representative areas including those most heavily contaminated as well as those moderately and minimally contaminated, and reference areas, synoptic with sediment chemistry determinations (i.e., use subsamples of the same sample for both chemical analyses and toxicity testing).
For situations where COPC are greater than SQG-low but substantially less than SQG-high, best professional judgement should be used to determine if subsequent toxicity testing or bioassessment is required. Typically, laboratory sediment toxicity tests are conducted with three or four appropriately sensitive, standardized sediment-dwelling and/or sediment associated test organisms (e.g., *Hexagenia*, *Hyalella*, chironomids, oligochaetes) that are reasonably similar to those found (or expected to be found) at the site (based on available data – Step 1), and combined end-points that involve survival, growth and reproduction (i.e., acute and chronic endpoints).

Decision Point 3b: Bulk sediment chemical analyses do not consider contaminant bioavailability, nor do they provide reliable information on the toxicity of sediment contaminants (reasonably reliable information can be obtained on the non-toxicity of sediment contaminants, cf. Decision Point 1). Thus, a determination is required as to whether or not the sediments that were previously assessed as contaminated, are toxic to individual organisms, and the extent of any toxicity.

Comparison	Decision
All sediment toxicity endpoints < 20% difference from reference **and** not statistically significantly different than reference	No further assessment required relative to laboratory toxicity. ***PROCEED TO STEP 4C***
One or more sediment endpoints > 20% difference from reference **and** statistically significantly different than reference	Potential risk; further assessment required. ***PROCEED TO STEP 4C***

Rationale: Although sediment toxicity tests have good power to detect differences between responses, a difference of 20% between controls and test/reference sediments is neither different nor environmentally relevant in short-term (e.g., 10-d), acute tests (Mearns *et al*., 1986; Washington State Sediment Management Standards [Ch173-204 WAC-17]; Suter, 1996; EPA/USACE, 1998; Environment Canada, 1998, 1999). For this framework, sediments with less than a 20% difference between controls and test/reference sediments are not considered to be toxic, even if the difference is statistically significant.

2.2.6 Step 4c: Is the benthic community impaired?

Determine whether the benthic community is significantly different from appropriate reference sites. Two questions need to be addressed. First, is it appropriate or realistic to assess the benthic community? There may be situations where benthic community structure assessments relative to possible sediment contaminant effects are not appropriate or realistically possible (e.g., shallow harbours where propeller scour, dredging or other habitat disturbances alter benthic communities independent of any contaminant effects; dynamic sediment bedflow that may alter the biological zone as a result of deposition or scour). Benthic community structure assessments will also not be possible for sediments deeper than about 10 cm because the vast majority of the sediment-dwelling organisms live in shallower depths than 10 cm although some organisms (e.g., some bivalves) can burrow much deeper. Second, is the benthic community at the site significantly different from the benthic community in reference areas? Benthic community structure is often described in terms of the diversity, abundance, and dominance of different invertebrate species living in or on the sediment. Assessment of the benthic community could include multimetric and/or multivariate analysis (as appropriate) to properly characterize it. Data interpretation using multivariate approaches are strongly recommended; however, the use of other metrics may have merit (Reynoldson *et al*., 1995, Hawkins *et al*., 2000, Barbour *et al*., 1999, Bailey *et al*., 2004, Env. Canada 2002, USEPA 2002c).

Decision Point 3c: Determine benthic community impairment.

Comparison	Decision
It is inappropriate to assess the benthic community.	***PROCEED TO STEP 5***
Benthic community is not significantly different from reference areas.	***PROCEED TO STEP 5***
Benthic community is significantly different from reference areas.	***PROCEED TO STEP 5***

Rationale: Assessing the benthic community at a site, and comparing results to the community at appropriate reference areas, provides valuable information on the cumulative effect of multiple stressors on the invertebrate species that live in or on the sediment. Typically, benthic organisms reside at a site over most of their life span, and therefore integrate the effects of exposure to COPC as well as other biological and physical stressors. Alteration in the benthic community may be related to the presence of elevated substances in the sediment but may also be due to other factors either natural (e.g., competition/predation, habitat differences) or human-related (e.g., water column contamination). A properly conducted field study and selection of appropriate reference sites are crucial for accurately assessing potential adverse effects to the benthic community at the site.

2.2.7 Step 5: Develop decision matrix

Develop a decision matrix based on and ranking data from the available LOE (sediment chemistry, toxicity, benthos [if available and appropriate] and bioaccumulation potential) – Table 1 (adapted from Grapentine *et al*., 2002a). Samples for sediment chemistry and toxicity are collected synoptically (subsamples of the same samples); samples for benthos are collected coincidentally (i.e., at the same locations but not on the same samples). Samples for benthos and chemistry analyses can be collected during initial field sampling and archived until and unless needed, thus reducing field costs. However, samples for sediment toxicity cannot be archived for longer than 8 weeks and should ideally be tested as soon as possible following collection (EPA/USACE, 1998). If benthos studies are not reasonably possible, fit other LOE into Table 2 and use best professional judgement in Step 6.

Decision Point 4: At this point a definitive decision may be possible. Specifically, sufficient information has now been gathered to allow for an assessment of three possibilities: (1) the contaminated sediments pose an environmental risk (see Section 7 re Risk Management); (2) the contaminated sediments may pose an environmental risk, but further assessment is required before a definitive decision can be made; (3) the contaminated sediments pose a negligible environmental risk. See Table 2 – note that definitive determinations are possible in 4 of 16 possible scenarios (two determinations of negligible environmental risk requiring no further actions; two of environmental risk requiring management actions).

Rationale: At this point definitive determinations are possible in some cases with the proviso that sediment stability may still need to be assessed (Step 7); in other cases, further assessment is needed, but can be guided by the results of this data integration. As noted by Wong (2004), SQGs do not provide definitive information for decisions regarding contaminated sediments, including remediation; a weight of evidence (WOE) approach is required. In a WOE approach, sediment chemistry data are given the least weight (Section 2.1, "rules" 1 and 2); benthic community data are given the most weight (Section 2.1, "rule" 3).

The type of WOE integration of LOE shown in Table 2 is usually applied on a station-by-station basis. Thus, although initial screening (Steps 1-3) is intended to screen out areas with relatively low contaminant concentrations, subsequent more detailed sampling of these areas may include stations with contaminant concentrations below levels of concern. Mapping of the results is one means to apply the findings on a large sample basis (i.e., to all sample locations), as a tool for expert/stakeholder groups to identify and focus on obvious problem areas/patterns.

Photo: Environment Canada.

Table 1

Ordinal Ranking For WOE Categorizations For Chemistry, Toxicity, Benthos And Biomagnification Potential.

	■	◘	□
Bulk Chemistry (compared to SQG)	**Adverse Effects Likely:** One or more exceedences of SQG-high	**Adverse Effects May or May not Occur:** One or more exceedences of SQG-low	**Adverse Effects Unlikely:** All contaminant concentrations below SQG-low
Toxicity Endpoints (relative to reference)	**Major:** Statistically significant reduction of more than 50% in one or more toxicological endpoints	**Minor:** Statistically significant reduction of more than 20% in one or more toxicological endpoints	**Negligible:** Reduction of 20% or less in all toxicological endpoints
Overall Toxicity	**Significant:** Multiple tests/endpoints exhibit major toxicological effects	**Potential:** Multiple tests/endpoints exhibit minor toxicological effects and/or one test/endpoint exhibits major effect	**Negligible:** Minor toxicological effects observed in no more than one endpoint
Benthos Alteration (multivariate assessment, e.g., ordination)	"different" or "very different" from reference stations	"possibly different" from reference stations	"equivalent" to reference stations
Biomagnification Potential (relative to reference)	**Significant:** Based on Step 6	**Possible:** Based on Step 4a	**Negligible:** Based on Steps 4a or 6
Overall WOE assessment	**Significant adverse effects:** elevated chemistry; greater than a 50% reduction in one or more toxicological endpoints; benthic community structure different (from reference) ; **and/or** significant potential for biomagnification	**Potential adverse effects:** elevated chemistry; greater than a 20% reduction in two or more toxicological endpoints; benthic community structure possibly different (from reference); **and/or** possible biomagnification potential	**No significant adverse effects:** minor reduction in no more than one toxicological endpoint; benthic community structure not different from reference; **and** negligible biomagnification potential

SQG = Sediment Quality Guideline; EC = Effective Concentration. Note That The Overall Definition Of "No Significant Adverse Effects" Is Independent Of Sediment Chemistry.

Table 2.

Decision Matrix for WOE Categorization. Based on Table 1, see text for explanation; a dash means "or". Separate endpoints can be included within each LOE (e.g., metals, PAHs, PCBs for Chemistry; survival, growth, reproduction for Toxicity; abundance, diversity, dominance for Benthos).

Scenario	Bulk Sediment Chemistry	Overall Toxicity[1]	Benthos Alteration[2]	Biomagnification Potential[3]	Assessment
1	□	□	□	□	No further actions needed
2	■–◘	□	□	□	No further actions needed
3	○	□	■–◘	□	Determine reason(s) for benthos alteration (Section 5.3)
4	□	■–◘	□	□	Determine reason(s) for sediment toxicity (Section 5.3)
5	□	□	□	◘	Fully assess risk of biomagnification (Section 4.3)
6	■–◘	■–◘	□	□	Determine reason(s) for sediment toxicity (Section 5.3)
7	□	□	■–◘	◘	Determine reason(s) for benthos alteration (Section 5.3) **and** fully assess risk of biomagnification (Section 4.3)
8	■–◘	□	■–◘	□	Determine reason(s) for benthos alteration (Section 5.3)
9	■–◘	□	□	◘	Fully assess risk of biomagnification (Section 4.3)
10	■–◘	■–◘	□	◘	Determine reason(s) for sediment toxicity (Section 5.3) **and** fully assess risk of biomagnification (Section 4.3)
11	■–◘	□	■–◘	◘	Determine reason(s) for benthos alteration (Section 5.3) **and** fully assess risk of biomagnification (Section 4.3)
12	□	■–◘	□	◘	Determine reason(s) for sediment toxicity (Section 5.3) **and** fully assess risk of biomagnification (Section 4.3)
13	□	■–◘	■–◘	□	Determine reason(s) for sediment toxicity **and** benthos alteration[2] (Section 5.3)

14	□	■–◘	■–◘	◘	Determine reason(s) for sediment toxicity **and** benthos alteration (Section 5.3), **and** fully assess risk of biomagnification (Section 4.3)
15	■–◘	■–◘	■–◘	□	Management actions required[4]
16	■–◘	■–◘	■–◘	◘	Management actions required[4]

[1] Overall toxicity refers to the results of laboratory sediment toxicity tests conducted with a range of test organisms and toxicity endpoints. A positive finding of sediment toxicity may suggest that elevated concentrations of COPC are adversely affecting test organisms. However, toxicity may also occur that is not related to sediment contamination as a result of laboratory error, problems with the testing protocol, or with the test organisms used.

[2] Benthos alteration may be due to other factors, either natural (e.g., competition/predation, habitat differences) or human-related (e.g., water column contamination). Benthos alteration may also be related to sediment toxicity if a substance is present that was not measured in the sediment or for which no sediment quality guidelines exist, or due to toxicity associated with the combined exposure to multiple substances.

[3] Per Table 1, significant biomagnification (■) can typically only be determined in Step 6; Step 3 only allows a determination that there either is negligible biomagnification potential or that there is possible biomagnification potential. However, there may be site-specific situations where sufficient evidence is already available from fish advisories and prior research to consider biomagnification at a site significant; this would be determined in Step 1 (examination of available data). Thus, for example, if significant biomagnification were indicated in Scenario 5, above, management actions would be required. The other three LOE do allow for definitive determinations in prior Steps of this Framework.

[4] Definitive determination possible. Ideally elevated chemistry should be shown to in fact be linked to observed biological effects (i.e., is causal), to ensure management actions address the problem(s). For example, there is no point in removing contaminated sediment if the source of contamination has not been addressed.. Ensuring causality may require additional investigations such as toxicity identification evaluation (TIE) and/or contaminant body residue (CBR) analyses (see Section 5.3). If bulk sediment chemistry, toxicity and benthos alteration all indicate that adverse effects are occurring, further assessments of biomagnification should await management actions dealing with the clearly identified problem of contaminated and toxic sediments adversely affecting the organisms living in those sediments. In other words, deal with the obvious problem, which may obviate the possible problem (e.g., dredging to deal with unacceptable contaminant-induced alterations to the benthos will effectively also address possible biomagnification issues).

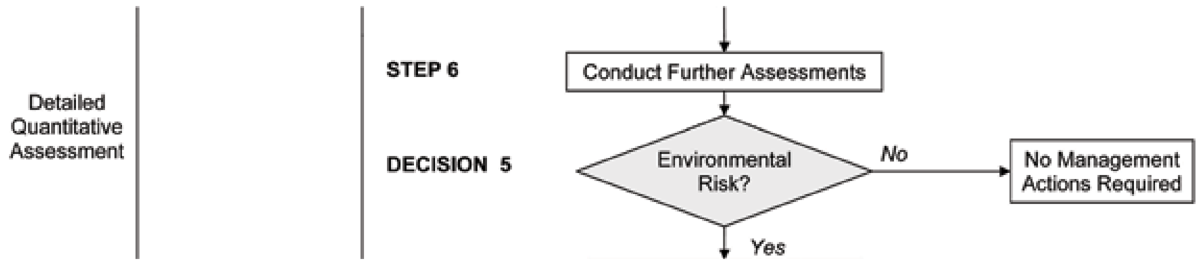


Figure 4. Detailed Quantitative Assessment (Step 6, Decision 5). See also Sections 4.3, 5.3, 6.1, 6.3 and 6.4. Decisions can be made regarding management actions for specific situations. In other situations, additional, focused investigations will be required.

2.2.8 Step 6: If necessary, conduct further assessments

As per the 16 possible scenarios in Table 2, 4 result in definite decisions and twelve possible scenarios result in a determination that the contaminated sediments may pose an environmental risk, but further assessment, outlined in Table 2, is required before a definitive decision is made.

Decision Point 5: Based on additional investigation, determine whether or not an environmental risk exists. ***This is where, in particular, and as noted in Section 2.2., it is critical that the study team include scientists with strong expertise in sediment chemistry (chemical fate, transport and speciation), sediment toxicity testing, benthic community assessment, food chain effects and environmental statistics for the design, implementation, and interpretation of both the previous and any additional investigative studies required.***

Rationale: (1) If there is no clear link between elevated chemistry (i.e., sediment contaminant concentrations > SQG-low) and biological effects (i.e., sediment toxicity and/or benthos alteration), there may be no point to sediment remediation as, if the sediment contaminants are not causative, sediment remediation will not ameliorate the biological effects. It is necessary to conduct more detailed studies to determine the cause of biological effects. (2) Observed toxicity and/or benthos alteration in the absence of elevated chemistry may be due to unmeasured contaminants or non-contaminant-related factors; either way, certainty as to causation is required (e.g., toxicity identification evaluation, TIE). (3) Modeling biomagnification only indicates whether there is no problem or may be a problem; if there is a potential biomagnification problem, more definitive assessments involving field measurements (e.g., contaminant body residue [CBR] analyses), laboratory studies, and/or more realistic modeling scenarios are required (see Section 4.3).

Photo: Environment Canada.

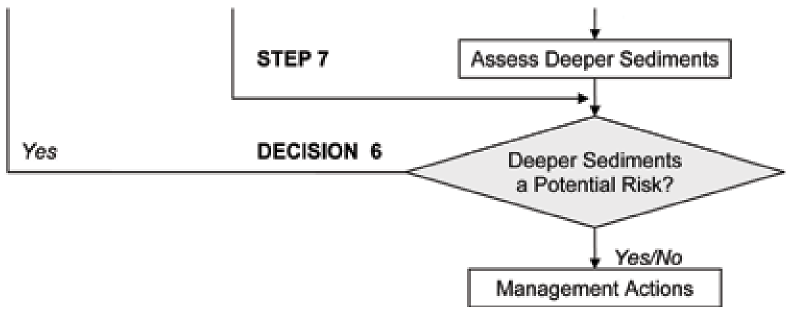


Figure 5. Assessment of Deeper (Below Surficial) Sediments (Step 7, Decision 6). If deeper sediments may pose a risk and could be exposed, the risk posed and need for management actions need to be determined.

2.2.9 Step 7: If necessary, assess deeper sediments

The previous assessments typically focus on surficial sediments (about 10 cm depth). Surficial sediments effectively cover deeper sediments, which may be similarly or differently contaminated. If so, there is a need to determine whether, under unusual but possible natural or human-related circumstances, these deeper sediments may be uncovered. Such studies involve an assessment of both sediment stability and sediment deposition rates.

Decision Point 6:

Comparison	Decision
Levels of COPC in deeper sediments below SQG-low and no substances present that can biomagnify, ***or*** deeper sediments very unlikely to be uncovered under any reasonably possible set of circumstances	No further assessment or remediation required. ***STOP.*** Management options for polluted surficial sediments should be determined.
Levels of COPC in deeper sediments above SQG-low and/or one or more substances present that can biomagnify, ***and*** these sediments may be uncovered under one or more reasonably possible set of circumstances	Potential risk; further assessment may be required (See Guidance, Section 1, "rule" 1). ***FOLLOW THE FRAMEWORK FROM STEP 1 (IF NECESSARY).*** Necessary information will probably already have been gathered for some initial steps.

Rationale: If deeper sediments are contaminated, and could be uncovered, they could pose an environmental risk, which needs to be evaluated. If the sediments are not likely to be uncovered, i.e., to become surface sediments, under any reasonably likely set of circumstances (e.g., a 100-year flood), then they do not require further assessment as any contaminants they contain will remain buried and there will be no exposure routes to biota.

3.0 ERA Components of the Framework: Problem Definition (Screening Assessment)

The following sections of this document provide information regarding key components of the ecological risk assessment (ERA) approach upon which the decision-making framework is explicitly based. The information provided is not and is not intended to be exhaustive (i.e., this document is not a "cook book"); rather, it is intended to provide readily understandable supporting information.

A Screening Assessment (Figure 2, Sections 2.2.1 to 2.23) involves simple, qualitative and/or comparative methods, with heavy reliance on literature information and previously collected data (CCME, 1996). Uncertainty (cf. Section 6.4) is highest at this level of ERA due to the use of conservative methodology and assumptions. Screening on both a conservative and a less conservative basis can provide a range of possible outcomes (which thus need to be investigated). Note that there is no single correct way to conduct this or other levels of an ERA. Subsequent ERA levels or tiers are conducted in an iterative approach, which generally means testing of hypotheses and conclusions and re-evaluating assumptions as new information is gathered.

3.1 Site Definition

Prior to initiating any investigations, spatial and temporal scales need to be explicitly defined. Sites typically comprise samples from multiple stations, and can be delineated based on ecologically defined scales (cf Section 6.1), on contaminant concentrations, or on other site-specific conditions. Within such delineations, species at risk and their habitats need to be considered, including the minimum home range of fish feeding on benthic invertebrates. Two additional determinations are needed: (1) does the site have a high level of environmental sensitivity based on habitat (not land use), e.g., is it a wetland used by migrating waterfowl or a feeding ground for shellfish or bottomfish; (2) is it contaminated only from off-site sources, which themselves need to be evaluated? These determinations will affect the design and implementation of subsequent investigations.

Further, the energy of the aquatic system should be considered in determining site boundaries. In a high energy system sediments may be washed downstream and deposited distal to the site. Likewise, evaluations of scour and deposition may show that sediments at depth may or may not be of concern or that the study area is potentially impacted from upstream sites.

3.2 Contaminants of Potential Concern (COPC)

Two classes of COPC need to be considered:

1. Contaminants that can cause acute (short-term, e.g., death) or chronic (longer-term, e.g., effects on growth and/or reproduction) effects to biota. The potential risk from these contaminants is assessed based on comparisons to SQG-low. Where SQG-low

are not available for particular contaminants, it may be possible to derive similar values using numerical methods from compilations of toxicity test data, such as species sensitivity distributions (SSDs). Note that SQGs of any sort are, by definition, preliminary, due to data limitations (O'Connor, 2004).

2. Contaminants that can biomagnify up food chains. Biomagnification is restricted to organic substances, e.g.: methyl Hg; PCBs; DDT; 2,3,7,8-TCDD.

3.3 Receptors of Potential Concern (ROPC)

Primary receptor species must both be potentially exposed to sediment contaminants (the COPC), and be relevant to the area being assessed (i.e., live or be expected to live primarily in that area). Secondary receptor species are the consumers of the primary receptor species. Agreement among stakeholders is required *a priori* regarding which receptor species to use for assessments and what surrogate species (if necessary) to use for toxicity testing.

3.4 Assessment Endpoints and Measures of Effect

An assessment endpoint is defined as the explicit expression of the environmental value that is to be protected. Examples of assessment endpoints include survival, growth and reproduction of major aquatic communities (e.g., aquatic plants, benthic invertebrates (bottom-dwelling animals without backbones), fish, aquatic-dependent birds and mammals). Generic ERA assessment endpoints are provided in USEPA (2003). A measure of effect is defined as the measurable ecological characteristic that is related to the assessment endpoint. Measures of effect comprise the actual measurements (e.g., actual determinations of survival, growth and reproduction via laboratory or other tests and/or field observations).

3.5 Reference Areas/Locations

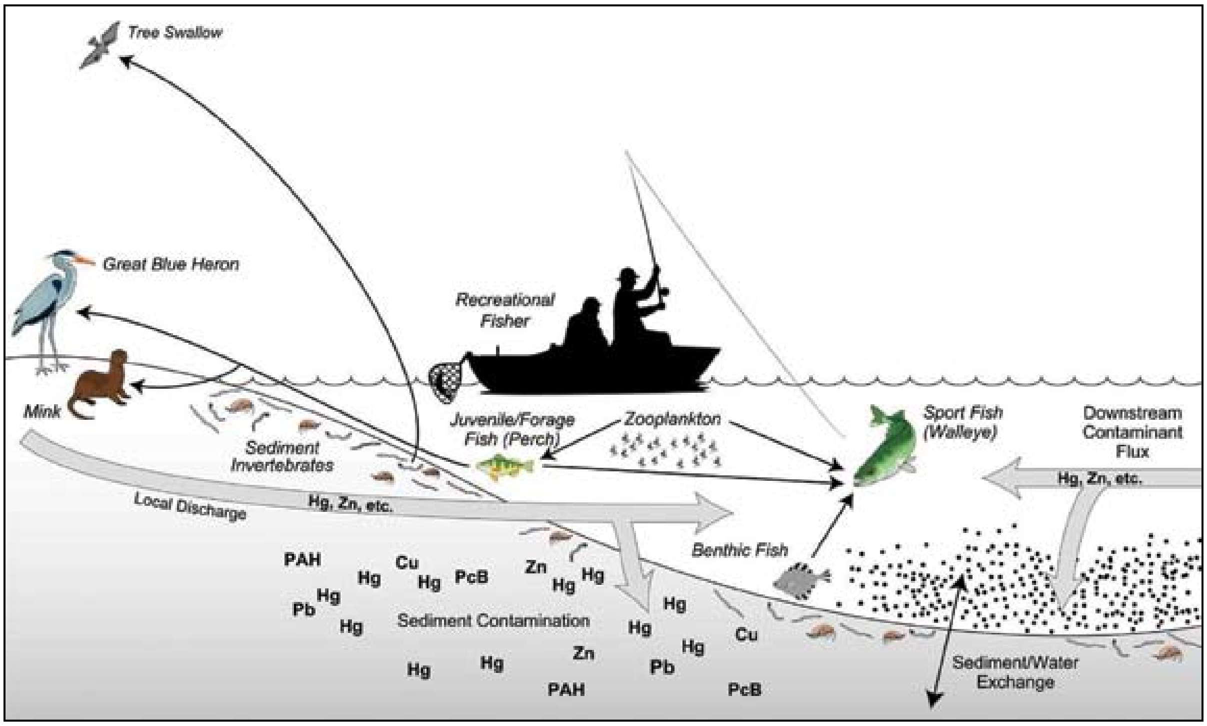
Reference areas/locations serve as the benchmarks against which to compare the contaminated sites. Typically, reference areas/locations represent "the optimal range of minimally impaired conditions that can be achieved at sites anticipated to be ecologically similar" and should be acceptable by local stakeholders and appropriately represent reference conditions (Krantzberg *et al.*, 2000). Ideally the same number of reference sites would be assessed as exposed sites; realistically, a smaller number can be used provided reference conditions are adequately quantified. However, some study areas may provide few or no suitable reference sites, and would be better sampled with a gradient array of sites.

Environment Canada has developed reference conditions for Great Lakes sediments based on a large data set of stations for three groups of parameters: physico-chemical attributes; toxicity; and, benthic community structure. Thus, exposed areas/locations can be compared to appropriate reference conditions by a variety of statistical methodologies (Reynoldson and Day, 1998; Reynoldson , 2002a).

Reference areas/locations can be used for three main applications (Apitz , 2002): to determine whether or not a contaminated area may require remediation; to determine incremental risk (between an exposed and reference site); and, in a post-remedial monitoring program.

3.6 Conceptual Site Model (CSM)

The conceptual site model (CSM) is a critical component of any sediment (or other) ERA assessment. It should involve both temporal and spatial components and be reviewed by regulatory agencies and other stakeholders prior to commencing field or laboratory studies to ensure there is agreement. It comprises "a three-dimensional description of a site and its environment that represents what is known (or suspected) about the contaminant source area(s), as well as, the physical, chemical, and biological processes that affect contaminant transport from the source(s) through site environmental media to potential environmental receptors. The CSM identifies assumptions used in site characterization, documents the relevant exposure pathways at the site, provides a template to conduct the exposure pathway evaluation and identifies relevant receptors and endpoints for evaluation. CSM development is an on-going, iterative process that should be initiated as early as possible in the investigative process. The CSM should be as simple or as complex as required to meet site objective(s). The CSM is also an important communication tool to facilitate the decision-making processes at the site" (Apitz, 2002). Work done at similar sites can assist in identifying potential shortcomings and pitfalls, and help focus the CSM to the extent possible.



Example of a Conceptual Site Model (CSM). Credit: Peter Chapman (Golder Associates Ltd.).

3.7 Sampling and Analysis Plan (SAP)

The Sampling and Analysis Plan (SAP) is developed based on all of the previous considerations (Sections 3.1 to 3.6). Its initial goal is to identify potential contaminant sources and to delineate areas of contamination (their full nature and their spatial – vertical and lateral – distribution) for subsequent investigation. Subsequent goals involve other LOE as per Figure 1. If a detailed quantitative assessment is conducted where PCBs, dioxins, and/or furans are COPC, congener specific information may be required to fully assess the potential risk of these compounds.

Photo: Environment Canada.

4.0 ERA Components of the Framework: Exposure Assessment

The decision-making framework is specific for environmental concerns associated with contaminated sediment, including not only ecological, but also human health concerns related to biomagnification. However, there may be situations where potential human health concerns are associated with dermal contact to contaminated sediment (e.g., swimming, wading), or by other exposure routes (e.g., flooding resulting in aquatic sediments contaminating residential soils or gardens, unacceptably high levels of contaminants that do not biomagnify such as Cd, Pb, PAHs in shellfish or fish). In such situations, a screening level HHRA should be considered to assess potential risks and inform the public.

4.1 Sediment Chemistry – Preliminary Quantitative

Preliminary quantitative assessment of sediment contaminants (Figure 2, Sections 2.2.1 to 2.2.3) can be done on the basis of individual contaminants or by using specific groups of contaminants as surrogates (Grapentine , 2002b). Combining information on different contaminants (e.g., Marvin , 2004) is not recommended due to information loss. However, where the mode of action and target effect of a toxicant are the same, additivity of contaminants can be considered. In addition, in some circumstances, an examination of integrated information from several types of contaminants (i.e., use of a Sediment Quality Index) could contribute to the overall interpretation of the data. Relying solely on such integrated information is not advised. Ancillary information required includes, but is not limited to, sediment particle size and total organic carbon (TOC) data. The extent of contamination can be characterized using techniques such as grids, random and stratified random sampling; the decision regarding which particular method to use will be site-specific.

4.2 Biomagnification Potential – Preliminary Quantitative

Uptake, bioaccumulation and biomagnification of chemicals through the food chain, which is restricted to a very few organic chemicals (e.g., methyl mercury; DDT; PCBs; 2,3,7,8-TCDD) should be considered on a case-by-case basis (Figure 3, Section 2.2.4). Fish advisories can provide useful information regarding issues (chemicals and species) related to biomagnification. Guidance in initial modeling efforts is provided in Grapentine (2003a,b). Essentially, “this approach relies on the application of conservative (i.e., protective) assumptions regarding BMFs and tissue residue criteria (TRC) to screen for potential toxicological effects to receptor species at higher trophic levels as the result of biomagnification from benthic invertebrate tissue through the food web” (Duncan Boyd, pers. comm.). Benthic invertebrate tissue concentrations are used to predict concentrations in higher trophic levels.

4.3 Detailed Quantitative

Detailed quantitative assessment within the framework is outlined in Figure 4, Sections 2.2.7 and 2.2.8). Because fish are mobile, their entire feeding area needs to be considered in order to fully assess the potential for some organic contaminants to biomagnify (e.g., through area curve modeling - Freshman and Menzie, 1996). Factors such as site-and species-specific BMFs, lipid content, age/size, and receptor food preference can also be incorporated. Utilizing more realistic assumptions than those used for preliminary quantitative assessment should allow for a better determination regarding the toxicological outcome for upper trophic level receptor species. Whereas the preliminary quantitative assessment is solely a modeling exercise based on sediment and benthos, this more detailed quantitative assessment involves other food chain measurements including fish and possibly plankton.

Natural fate and transport processes affecting sediment contaminants must also be considered, and could include: in-bed fate processes, including irreversible adsorption and chemical or biological reactions; in-bed transport processes, including diffusion and advection; interfacial transport processes, including sediment deposition and resuspension and bioturbation. Potential contaminant sources from groundwater should also be considered. Direct field evidence will be required in some cases. In other cases, reasonable assumptions may be possible based on scientific knowledge and best professional judgement.

More detailed sediment chemistry exposure assessment related to determination of causation could, in some cases, involve the use of biomarkers. Multiple biomarkers can be used in their own WOE assessment as part of the overall ERA (Galloway , 2004).

Photo: Ontario Ministry of the Environment.

5.0 ERA Components of the Framework: Effects Assessment

5.1 Toxicity Testing – Preliminary Quantitative

The magnitude of any toxicity (Figure 3, Section 2.2.5) associated with exposure to contaminants in the sediments is assessed. Such information is typically determined from sediment toxicity tests with well-established, standard test organisms. The possibility of toxicity due to factors other than the COPC (e.g., grain size, ammonia, sulfides) is typically considered as part of standardized test procedures. Various approaches are possible for integrating multiple toxicological endpoints into a single LOE, however the results of laboratory toxicity tests do not reliably predict effects to field populations (Suter, 1996; Reynoldson *et al.*, 2002a; Chapman *et al.*, 2002).

5.2 Benthos Alteration – Preliminary Quantitative

Benthos alteration (Figure 3, Section 2.2.6) is assessed by identifying and enumerating benthic assemblages, and using both univariate (e.g., species richness, abundance, dominance) and multivariate analyses (e.g., ordination, principle component analysis [PCA]) to determine similarities and differences from reference areas and/or conditions (Chapman, 1996; Simpson *et al.*, 2005).

5.3 Detailed Quantitative

Detailed quantitative toxicity assessment (Figure 4, Table 2, Section 2.2.8) involves additional or more extensive studies as appropriate to site-specific circumstances, for example: spiked sediment toxicity tests; TIE; CBR analyses; tests with resident organisms; in situ bioassays.

Spiked sediment toxicity tests involve adding increasing concentrations of one or more suspected toxicants to a reference sediment and determining concentrations at which effects occur compared to exposed sediments. This procedure can also be applied to exposed sediments. It assists in identifying causative agents for observed toxicity and/or benthic community alterations. Similar information can be provided by TIE and CBR.

TIE were originally based on water or effluent toxicity tests and involve manipulating the chemical composition of toxic samples to remove specific substances (e.g., metals, ammonia) followed by retesting (Burgess, 2000). When an expected toxic effect is not observed as a result of removing specific substance(s), those substance(s) are added back, and the toxic effect is reassessed to confirm that those substances are indeed responsible for the initially observed toxicity, and that toxicity recurs at about the same levels as initially. TIE were subsequently applied to sediment pore waters (assuming that most of the toxicity observed in sediments was due to aqueous exposure routes) (Ankley and Schubauer-Berigan, 1995). They have recently been applied to whole sediments in the marine environment, and although procedures are not yet available to perform full TIE on whole sediments, those procedures that are available show good promise (Burgess *et al.*, 2000; Pelletier *et al.*, 2001; Burgess *et al.*, 2003; Ho *et al.*, 2004).

A chemical fractionation scheme has been used together with toxicity testing, to attempt to determine causation in whole sediment freshwater toxicity tests in Lake Ontario (McCarthy *et al.*, 2004).

CBR determinations are based on the fact that, for a contaminant to cause toxicity to an organism, that contaminant has to contact a biological receptor, which generally means the contaminant must be bioaccumulated (taken up) by the organism. Though this remains an active area of research, contaminant concentrations in organisms have been linked to effects (Jarvinen and Ankley, 1999), and used to determine causation in WOE determinations (e.g., Borgmann *et al.*, 2001).

Testing the responses of resident organisms may be appropriate to determine, for instance, why laboratory tests with standard organisms indicate toxicity, but there are no alterations to resident benthic communities. It is entirely possible that resident organisms are more tolerant to sediment contaminants than naïve, laboratory organisms (Chapman *et al.*, 2003). If tolerance has been established, then whether or not there are also costs in terms of the loss of intolerant species or energetic costs to the tolerant organisms should be determined.

In a similar manner, in situ bioassays (toxicity and/or bioaccumulation) can be used to test for differences between responses in the laboratory and in the field. Laboratory bioassays are conducted under controlled conditions that will not mimic field conditions to which resident populations are exposed. Conducting bioassays in situ and comparing the results to laboratory tests can assist in determining why differences in responses occur, and whether or not resident populations are at risk (laboratory bioassays tend to be conservative).

Photo: Environment Canada.

6.0 Risk Characterization

The basic approach of starting with chemical hazard assessment (i.e., the use of SQGs – Figure 2), then adding toxicity tests, followed by receiving environment evaluations (Figure 3), matches current practices in the Great Lakes and other parts of Canada as well as the USA (Krantzberg *et al*., 2000; Appendix II), and international trends (Power and Boumphrey, 2004; Apitz *et al*., 2005). The Framework contained herein can be applied to both large and small sites in terms of both preliminary and more detailed assessments. It fits within the ERA paradigm, and provides information necessary for the protection of both local aquatic communities and endangered species. The framework also differentiates between those scenarios where elevated concentrations of contaminants are associated with adverse biological effects and those scenarios where they are not (since the presence of substances in sediments where they would not normally be found, or at concentrations above natural background levels, does not necessarily mean that adverse biological effects are occurring). The following documents provide additional detailed information regarding various LOE mentioned herein and their eventual use in risk characterization: MacDonald *et al.* (2002a,b); Ingersoll and MacDonald (2002); Suter *et al.* (2002).

6.1 Issues of Scale

Issues of scale need to be considered on a site-and situation-specific basis, and are an important factor in choosing between management actions and further study. Estimated exposure from a large area is usually much lower than exposure from a specific, localized site. Under the Contaminated Sites process, the Ontario Ministry of Environment (OMOE) does not allow the relatively high risks of small “hot spots” to be “averaged down” by the relatively small risks of the less contaminated surrounding area. Further, ERA should not be used to avoid addressing an extreme, local “hot spot”. However, considerations of biomagnification potential at a Detailed Quantitative level need to consider the feeding ranges (area use) and preferences of fish and waterfowl (i.e., the measured or assumed fraction of a predator’s diet that is represented by a particular prey species). Area use represents the proportion of a prey species’ home range associated with a particular area of contaminated sediments, and can include seasonal exposure during critical life stages or diminished exposure of migratory species.

6.2 Preliminary Quantitative

A Preliminary Quantitative ERA (Tables 1 and 2, Section 2.2.7) provides more quantitative information than a Screening Assessment, reduces uncertainty, and is more extensive and expensive (CCME, 1996). Exposure and effects assessments are integrated to determine whether or not significant effects are occurring or are likely to occur. In addition, the nature, magnitude, and areal extent of effects on the selected assessment points are described. The substances that may be causing or substantially contributing to such effects (the contaminants of concern (COCs)) are identified to the extent possible.

The results for each LOE are compiled and interpreted separately. Subsequently, they are combined and integrated, including uncertainty and best professional judgement, to establish a WOE for assessing risks (e.g., Chapman *et al*., 2002; Reynoldson *et al*., 2002). WOE approaches need to be: as quantitative as possible; transparent; and, draw on a broad range of interdisciplinary expertise (Burton *et al*., 2002). Risks of adverse effects can generally be considered in four categories:

- Negligible – similar to those for reference conditions
- Moderate – minor or potential differences compared to reference conditions
- High – major or significant differences compared to reference conditions
- Uncertain – requiring further study (e.g., a Detailed Quantitative assessment).

6.3 Detailed Quantitative

A detailed quantitative assessment (Table 2, Section 2.2.8) is the most extensive form of ERA, relying on site-specific data and predictive modeling; information is as quantitative as possible (CCME, 1996). It is intended to reduce key uncertainties in a transparent and scientifically sound manner such that final decisions can be made for all potential contaminated sediment scenarios. Typically, lower ERA tiers involve conservative or "worst case" assumptions. This higher tier of ERA typically involves more realistic assumptions.

Detailed quantitative assessment also generally involves determination of causation, specifically answering the question as to whether or not any observed biological effects are due to sediment contaminants and, if so, which contaminant(s) and at what concentration(s) (e.g., Suter *et al*., 2002). Although sediment stability issues can be addressed initially in a Preliminary Quantitative ERA, they are conclusively addressed here.

Risks will generally be considered in three categories:

- Negligible – similar to those for reference conditions
- Moderate – minor or potential differences compared to reference conditions
- High – major or significant differences compared to reference conditions.

6.4 Uncertainty

Scientific investigations do not always result in easy answers. Uncertainty is inherent in any and all ERA. However, the ERA process is designed to accommodate the relationship between scientific uncertainty and the ability of risk managers to make risk management decisions. The goal in progressing from screening to more quantitative assessment is to diminish key uncertainties and improve confidence in the decision-making process.

In the case of biomagnification assessments, site-specific data and locally relevant food-web structure will diminish the uncertainty associated with extrapolations from literature-based models. However, food-web modeling and predictions will still be required to evaluate possible effects related to biomagnification. Thus, uncertainty cannot be totally eliminated.

There are two general types of uncertainty. Stochastic uncertainty refers to the inherent randomness of the system being assessed, and can be described and estimated but cannot be reduced. Uncertainty arising from human error or from imperfect knowledge can, however, be reduced. In the case of biomagnification assessments, the major sources of the latter type of uncertainty are variability in model inputs (empirically observed variation and/or lack of data for key parameters, and the assumptions and simplifications which are inherent to the structure of any particular model).

Stochastic uncertainty results in intrinsic model limitations that are not the result of a lack of data or computational power. For example, food web model predictions are considered good if they are within a factor of five of observed concentrations for upper trophic level receptors. This leaves a considerable measure of uncertainty for decision-makers to deal with, since this margin of error will frequently exceed the scale of the relative improvement in ecosystem outcome which is desired.

CCME (1996) requires the identification of “key uncertainties”, a management decision as to whether they are acceptable or not, and an evaluation as to whether a preliminary quantitative ERA exposure assessment would significantly reduce uncertainty. The USEPA (1988) identifies the importance of quantitative uncertainty analysis and has published a policy for use of probabilistic analysis in risk assessment.

Three common methods for dealing with sources of uncertainty are sensitivity analysis, Monte Carlo simulation, and the use of monitoring data for model calibration. Sensitivity analysis is a fundamental requirement of any model application and geared to ensuring that the level of effort applied to improving the accuracy of model input parameters is commensurate with their effect on the accuracy of modeled output. Input parameters which have only a small effect on the accuracy of modeled output can be estimated by less accurate and costly methods. Once sensitivity analysis has identified the critical input parameters, a Monte Carlo analysis provides a stochastic approach to generating probabilistic model output through repetitive model runs using the distribution characteristics of uncertain model input parameters. The probability distributions associated with this approach provide an excellent means of quantifying model uncertainty. However, unless the input parameter distribution characteristics are derived from actual data, the uncertainty in outputs is purely a function of assumptions made about the uncertainty of input parameters. Model calibration using monitoring data is an obvious and necessary means of diminishing uncertainty, but good modeling practice requires that model calibration and validation use independent data to avoid assuming that which is to be predicted.

Progression from a screening level assessment, to a more quantitative assessment incorporating site-specifically derived values such as biomagnification factors (BMF), area use factors, and food preference factors for receptor species may result in some reduction of uncertainty compared with the use of literature values. It may also improve the ability to quantify and partition uncertainty. However, the achievable reduction in uncertainty requires careful evaluation before the decision is made to proceed with a more quantitative risk assessment, since it may not diminish uncertainty to the point where decision-making becomes any more straightforward. If the analysis demonstrates that the potential for significant reduction in uncertainty is limited, then the risk manager must evaluate whether the benefits of the ensuing marginal decrease in uncertainty justify the corresponding time and costs. It may prove more expedient to proceed to an examination of risk management options, particularly in cases where socioeconomic or technological constraints may limit these options.

In order to ensure that the allocation of time and resources to a quantitative ERA will sufficiently diminish uncertainty for risk management decision-makers, a quantitative uncertainty analysis must be applied at all sites as a prerequisite for proceeding from a screening level ERA to a quantitative ERA. This requirement is generic and not specific to biomagnification assessment.

In the specific case of biomagnification assessment, the accuracy of model predictions of tissue residues in third or fourth trophic level receptor species cannot be quantitatively validated using site-specific data due to the complexity of such food chain transfers, and hence site-specific tissue residue data should only be used to qualitatively ground-truth model predictions. Because sensitivity analysis will generally identify benthic invertebrate tissue concentrations as the most critical measurable input parameter in food chain models, measurement of invertebrate tissue residues should be used as the primary means of assessing biological exposure.

Photo: Environment Canada.

7.0 Risk Management

Risk management is distinct from risk assessment; the latter is primarily scientific, the former includes risk assessment along with other non-scientific considerations such as societal and economic concerns. Good science alone does not yield good management, but is an essential prerequisite for good decision-making. For example, the "range and significance of natural processes… must be adequately assessed prior to the selection, design and optimization of any management options for contaminated sediments" (Apitz *et al.*, 2002).

Application of the framework will assist in the eventual delisting of AOCs. Delisting criteria for AOCs can include: no consumption advisories for public health or wildlife (i.e., guidelines and objectives not exceeded); healthy benthos, fish and wildlife populations (i.e., self-sustaining communities at the expected level of abundance when compared to reference conditions or, in the absence of community structure data, no significant water or sediment toxicity); normal rates of fish tumours, deformities and reproductive problems in fish, birds and mammals (i.e., rates not elevated above reference conditions); and, no restrictions on dredging activities (i.e., guidelines and objectives not exceeded). Delisting will also require monitoring to ensure that any necessary management actions have been effective.

Photo: Ontario Ministry of the Environment.

Appendix 1 - Annotated Bibliography

Anderson J, Boyd D. 2003. Ecological Risk Assessment in Sediment Management Decision-Making. Unpublished document, November 3, 2003 draft.

This brief document describes the forthcoming preparation of a formal guidance document for sediment management decision-making in the Great Lakes. This guidance document is to describe a "standardized approach and process to follow when assessing contaminated sediments and provide a decision framework for management actions. The objective is to have a consistent approach to sediment decision-making in the Great Lakes that is objective, transparent and scientifically rigorous." Key issues including considerations and recommendations are provided regarding: defining the site; dealing with uncertainty; estimating hazard; and, statistical confidence.

Apitz SE, Davis JW, Finkelstein K, Hohreiter DL, Hoke R, Jensen RH, Jersak JM, Kirtay VJ, Mack EE, Magar V, Moore D, Reible D, Stahl R. 2002. Critical Issues for Contaminated Sediment Management. US Navy, Space and Naval Warfare Systems Center, San Diego, CA, USA. MESO-02-TM-01. http://meso.spawar.navy.mil/docs/MESO-02-TM-01.pdf.

This document emphasizes the importance of the conceptual site model, and discusses in detail: sediment ERA tools; characterizing the spatial extent of contamination; use or rapid sediment characterization tools; reference area conditions; sediment toxicity testing; contaminant bioavailability; in situ bioaccumulation tests; natural processes determining contaminant and sediment fate; monitoring natural recovery; monitoring remedial effectiveness. The importance of using different LOE in a WOE approach is emphasized.

Borgmann U, Norwood WP, Reynoldson TB, Rosa F. 2001. Identifying cause in sediment assessments: bioavailability and the Sediment Quality Triad. Can J Fish Aquat Sci 58: 950-960.

The authors combined bioaccumulation data with the SQT to determine not only environmental impacts, but also the most probable cause of those impacts. Adding bioaccumulation and overlying water measurements to the SQT allowed them to identify nickel as the major metal of concern in their study area.

Burton GA Jr, Chapman PM, Smith EP. 2002. Weight-of-evidence approaches for assessing ecosystem impairment. Human Ecol Risk Assess 8:1657-1673.

This review summarizes different approaches to WOE including: advantages; limitations; and, uncertainties. Critical issues involved in executing different LOE and their subsequent integration into WOE to characterize the likelihood of impairment are discussed. WOE does not remove uncertainty, but should reduce uncertainty in a transparent and scientifically sound manner. It is noted that WOE approaches need to be: as quantitative as possible; transparent; and, draw on a broad range of interdisciplinary expertise.

Burton GA Jr, Batley GE, Chapman PM, Forbes VE, Schlekat CE, Smith PE, den Besten PJ, Barker J, Reynoldson T, Green AS, Dwyer RL, Bertin WR. 2002. A weight-of-evidence framework for assessing sediment (or other) contamination: Improving certainty in the decision-making process. Human Ecol Risk Assess 8: 1675-1696.

This paper recommends a generic, technically defensible, widely usable basic framework for WOE for sediment assessments, focusing on stable sediments (e.g., sediments unlikely to be disturbed by physical means). The framework comprises a series of steps adapted from the ecological risk assessment (ERA) paradigm: identify critical receptors, define ecosystem quality, and identify potential stressors and associated exposure dynamics; develop a conceptual model; determine measurement endpoint responses; select reference sites and comparison methods; select appropriate LOE combinations and a method to integrate the LOE into a WOE; finalize study design including quality assurance/quality control (QA/QC); collect and verify data; analyze each LOE; integrate LOE into a WOE matrix, evaluating against the conceptual model and, if necessary, revisiting the conceptual model and/or collecting additional data; draw conclusions.

CCME. 1996. A Framework for Ecological Risk Assessment: General Guidance. Canadian Council of Ministers of the Environment. Winnipeg, MB, Canada. EN 108-4-10-1996E.

This document provides general guidance for the conduct of ERA in Canada. The ERA framework is iterative with three tiers: Screening Assessment (simple, qualitative and/or comparative methods, heavy reliance on literature information and previously collected data); Preliminary Quantitative (provides more quantitative information, reduces uncertainty, more extensive and expensive); Detailed Quantitative (most extensive and expensive; relies on site-specific data and predictive modeling; information is as quantitative as possible).

Chapman PM. 1996. Presentation and interpretation of Sediment Quality Triad data. Ecotoxicology 5: 327-339.

This paper updates previous papers by this author to provide general guidance on the use of three different LOE (sediment chemistry, sediment toxicity, and benthic community structure), and their combination into the WOE framework that comprises the SQT. The use of indices, in particular previously used ratio-to-reference comparisons, is not encouraged due to loss of information. Examples are provides of both a tabular decision matrix and methods to visually present SQT data.

Chapman PM. 2004. Modifying Paracelsus' Dictum for sediment quality (and other) assessments. Aquat Ecosyst Health Manage 7: 1-6.

This publication argues that Paracelsus' Dictum ("The right dose makes the poison"), though it correctly separates pollutants (toxicity occurs) from contaminants (toxicity does not occur), neither includes nor considers two critical modifying factors: bioavailability and realistic exposure scenarios. These two modifying factors can greatly affect whether or not, and to what extent, sediment contaminants are toxic to biota. Examples from sediment investigations are provided. It is argued that the Dictum needs to be rephrased: "All substances are poisons; there is none which is not a poison. The right dose of a bioavailable substance, administered under realistic exposure conditions, differentiates a poison."

Chapman PM, McDonald BG. 2004. Risk assessment using the Sediment Quality Triad. In: Blaise C, Férard J-F (eds.), Small-Scale Freshwater Environment Toxicity Test Methods. Kluwer Academic Press, Netherlands (in press).

This forthcoming book chapter provides a summary of the SQT including: a historical overview; reported applications; advantages; and, a full description of the procedure. Factors capable of influencing interpretation of the SQT are outlined and a detailed case study is presented. The chapter concludes with lessons learned, future prospects, and conclusions.

Chapman PM, McDonald GC, Lawrence GS. 2002. Weight of evidence frameworks for sediment quality and other assessments. Human Ecol Risk Assess 8: 1489-1515.

This paper summarizes different WOE frameworks, broadly divided into five different general categories: indices, statistical summarization, scoring systems, logic systems, and best professional judgment (BPJ). It is concluded that all categories are potentially useful with the exception of the first category. Specifically, development and use of indices is not recommended as indices result in information compression that, in the particular case of biological data, can negate full use of WOE. A tiered approach to WOE is recommended, congruent with ERA approaches. Three examples of sediment assessments involving WOE are detailed. Figure 1 provides the framework for a tiered sediment assessment congruent with ERA. It is argued that sediment WOE is based on correctly answering six specific questions:

1. Are contaminants present at levels of concern? (sediment chemistry)
2. Are the contaminants capable of causing toxicity? (laboratory toxicity tests)
3. Are resident biotic communities altered? (community structure analyses)
4. Are the contaminants causing the observed toxicity and/or community alterations (manipulative/investigative studies, e.g., TIE [toxicity identification evaluation], CBR determinations)
5. Are any contaminants of concern capable of and likely to biomagnify? (sediment chemistry and tissue analyses, food chain modeling)
6. Is the sediment stable or is it liable to erosion resulting in exposure of deeper, more contaminated sediments and/or contamination down-current? (shear stress and cohesion measurements relative to possible and unusual events)

CSMWG (Contaminated Sites Management Working Group). 1997. A Risk Management Framework for Contaminated Sites. A Discussion Paper. Soil Section, Guidelines Division, Science Policy and Environmental Quality Branch, Environment Canada.

This document deals with both risk evaluation and risk management. Risk evaluation is defined as using either risk-based environmental quality guidelines or risk assessment to establish site-specific remediation objectives. A separation of the roles of risk evaluation and management is noted as appropriate, however interaction and communication between the two is also noted as appropriate. Chemical analyses are noted as a method to determine whether or not further investigations/actions are required. Three different tiers of ERA are explained: screening assessment; preliminary quantitative; and, detailed quantitative. Types of uncertainty are described. The importance of multiple, independent LOE serving to provide WOE for supporting decisions is emphasized.

Dillon Consulting Ltd. 1999. A Federal Approach to Contaminated Sites. Contaminated Sites Management Working Group, Ottawa, ON, Canada.

This document was prepared for managers and other operational personnel responsible for managing contaminated sites on federal lands. Figure 1 provides useful Steps for Addressing a Contaminated Site. The document includes both ERA and HHRA in a presentation that is detailed but relative simplistic. There is no explicit reference to LOE or WOE.

Environment Canada/Ontario Ministry of the Environment COA Sediment Decision-Making Task Group. Undated. Conditions for Application of Ecological Risk Assessment (ERA) to Management of Sediments Containing Substances which Biomagnify.

This brief document outlines general conditions for the application of ERA to sediments contaminated with potentially biomagnifying substances related to a forthcoming Guidance Document. It is noted that guidance must be "sufficiently prescriptive to standardize the decision-making process throughout the Great Lakes basin" but without "cook book" assessments. The conceptual model should include "both temporal and spatial components [at appropriate scales] and site definition should be locally relevant and receptor driven." Key issues are: site definition; uncertainty and food web models; estimation of hazard; decision rules and Type I and Type II errors.

Environmental Response Team. 1997. Superfund Program Representative Sampling Guidance. Volume 3: Biological. Interim Final Draft. Office of Emergency and Remedial Response, Office of Solid Waste and Emergency Response, USEPA, Washington, DC, USA.

This guidance document is based on the SQT approach. LOE are: chemical residues; population/community studies; toxicity testing. There is no discussion of WOE, rather the focus appears to be on independent interpretation of each LOE.

Forbes VA, Calow P. 2002. Applying weight of evidence in retrospective ecological risk assessment when quantitative data are limited. Human Ecol Risk Assess 8: 1625-1640.

This paper suggests a WOE approach for use where quantitative data are limited. Although based on BPJ, the approach is logical, transparent and systematic. Specifically, the authors use human epidemiological criteria as the basis for seven questions similar to those posed by Chapman *et al.* (2002, above) whose answers are then weighted (a scoring system - very likely, likely, possibly, unlikely, don't know) to assign a likelihood of involvement of putative agents. On this basis, 15 possible scenarios are described, including the possibility of multiple causation, and illustrated using three real-world case studies involving declines of benthos, fisheries and molluscs. Effectively, this WOE approach is synonymous with the problem formulation or screening level stage of an ERA. This paper is updated in a forthcoming publication that, as demonstrated by two case studies, proposes "a method that aims to guide interpretation of various combinations of answers to the questions so that conclusions about the likelihood that identified agents have caused the observed effects in sediment systems can be consistently drawn:

Forbes VE, Calow P. 2004. A systematic approach to weight of evidence in sediment quality assessments: challenges and opportunities. J Aquat Ecosyst Health Manage (in press).

Galloway TS, Brown RJ, Browne MA, Dissanayake A, Lowe D, Jones MB, Depledge MH. 2004. A multibiomarker approach to environmental assessment. Environ Sci Technol 38: 1723-1731.

This paper demonstrates the use of multiple biomarkers together with chemical analyses to assess effects of contaminant exposures to invertebrates living on or above the sediments (e.g., clams, crabs). The multiple biomarkers are used in a WOE assessment within this LOE to reveal the existence of "environmental stress". The relevance of this approach to ERA is discussed.

Grapentine L, Marvin CH, Painter S. 2002a. Development and application of a sediment quality index for the Great Lakes and associated areas of concern. Human Ecol Risk Assess 8: 1549-1567.

Marvin C, Grapentine L, Painter S. 2004. Application of a sediment quality index to the Lower Laurentian Great Lakes. Environ Monit Assess 91: 1-16.

These two publications deal with the development of a sediment quality index (SQI) for summarizing and integrating sediment chemistry data for different contaminants. The SQI incorporates both the number of SQG exceedences and the extent of such exceedences to provide what is essentially a hazard index for the sediment chemistry LOE. The SQI was used to categorize different areas of the Great Lakes in terms of sediment quality based solely on sediment chemistry.

Grapentine L, Anderson J, Boyd D, Burton GA Jr, DeBarros C, Johnson G, Marvin C, Milani D, Painter S, Pascoe T, Reynoldson T, Richman L, Solomon K, Chapman PM. 2002b. A decision-making framework for sediment assessment developed for the Great Lakes. Human Ecol Risk Assess 8: 1641-1655.

A rule-based WOE approach that expands the SQT to four LOE is described. The four LOE are: sediment chemistry, sediment toxicity, benthic community structure, and biomagnification. A total of 16 outcome scenarios are possible; risk management is required for 9 of these scenarios. Ranking of individual LOE is also described. The framework is intended to be transparent, comprehensive and minimize uncertainty, but is not prescriptive. It is correctly noted "observations of elevated concentrations of contaminants in sediments alone are not indicators of ecological degradation. Rather, it is the biological responses of those contaminants that are of concern." Figure 1 and Table 1 are particularly relevant.

Grapentine L, Milani D, Mackay S. 2003. A Study of the Bioavailability of Mercury and the Potential for Biomagnification from Sediment in Jellicoe Cove, Peninsula Harbour. Environment Canada, Burlington, ON, Canada.

This study evaluated the biomagnification LOE for a specific site in the Great Lakes contaminated with mercury. It involved: comparisons of total to methyl mercury in sediments from the exposed and reference locations; analyses of the relationships of total and methyl mercury concentrations in invertebrates and sediments; and, predictions of the concentrations of total and methyl mercury in representative consumers of benthic invertebrates and their predators using screening-level trophic transfer models. Application of an areal averaging exposure model resulted in the conclusion that removal of mercury from an area of 7 contiguous sites reduced the predicted risk of adverse effects for the whole to negligible. The results of such studies provide one of two possible answers: either trophic transfer is not an issue, or trophic transfer may be an issue. Definitive answers are not possible from such studies "due to uncertainties associated with predicting receptor mercury concentrations."

Griffith MB, Lazorchak JM, Herlihy AT. 2004. Relationships among exceedences of metals criteria, the results of ambient bioassays, and community metrics in mining - impacted streams. Environ Toxicol Chem 23: 1786-1795.

This paper applies the primary methods (the SQT) used by USEPA for ecological assessment of contaminated sediments to mining-impacted streams: sediment criteria; ambient toxicity assessments; and, bioassessment of macroinvertebrates. USEPA LOE are described as follows. Chemical criteria are derived using numerical methods from compilations of toxicity test data, such as species sensitivity distributions. Ambient toxicity assessments involve tests with standard species such as, in freshwater, the amphipod *Hyalella azteca* and/or the chironomid *Chironomus riparius*. Bioassessments enumerate benthic macroinvertebrate assemblages, calculate metrics that describe the assemblages, and sum the metric scores to produce indices of biotic integrity.

Hollert H, Heise S, Pudenz S, Brüggemann R, Ahlf W, Braunbeck T. 2002b. Application of a Sediment Quality Triad and different statistical approaches (Hasse diagrams and fuzzy logic) for the comparative evaluation of small streams. Ecotoxicology 11: 311-321.

Hollert H, Dürr M, Olsman H, Halldin K, van Bavel B, Brack W, Tysklind M, Engwall M, Braunbeck T. 2002. Biological and chemical determination of dioxinlike compounds in sediments by means of a Sediment Triad approach in the catchment area of the River Neckar. Ecotoxicology 11: 323-336.

These two companion papers detail the use of different statistical methods for the evaluation and presentation of SQT data. The SQT comprised the traditional three LOE, with the toxicity LOE augmented by mutagenic, genotoxic, teratogenic, dioxin- and estrogen- like responses. A ranking procedure and Hasse diagram technique were suitable WOE integrators, but required expert knowledge to interpret the results. In contrast, the application of fuzzy logic allowed development of site-specific expert systems.

Jaagumagi R, Persaud D. 1996. An Integrated Approach to the Evaluation and Management of Contaminated Sediments. Ontario Ministry of the Environment, Standards Development Branch, Environmental Standards Section.

This document describes a stepwise approach to sediment assessment and also discusses choice of remediation option. The role of social and economic considerations in addition to scientific considerations is discussed. The report includes: relevant (to that date) legislation; data gathering; data evaluation and findings; sediment remediation and options; implementation of a cleanup plan; and, post-remedial monitoring. Different LOE are mentioned but there is no discussion of WOE.

Keegan RE, Anderson PD, Alsop WR, Samuelian JH. 1999. Risk-Based Management Principles for Evaluating Sediment Management Options. Ogden Environmental and Energy Services and Sediment Management Work Group.

This unpublished document presents a tiered approach to risk assessment and risk management applicable to both ERA and HHRA. It focuses on selection of receptors, development of exposure scenarios, and specification of assessment endpoints; endpoint identification is primary. The document considers both direct toxicity and bioaccumulative effects of COCs in sediments. It identifies three potential outcomes: no further action required; additional analyses required; action required. RAs are described on two levels: screening, quantitative and site-specific. Figure B-1 is particularly relevant to sediment assessment decision-making.

Kemper JF, Kindzierski W, Gaudet C, Moore D. 1997. Evaluation of Risk-Based Approaches in Environmental Guideline and Standard Setting. Part 1. Executive and Policy Summary. Kemper & Associates Inc. and CCME. Ottawa, ON, Canada.

This document attempts to provide guidance on common terminology for RA, an overview of how risk factors have been used in guideline development, and recommendations for further incorporation of risk into standards developed by CCME. The document notes that “there is no one correct way to conduct a risk assessment or to manage risk.” It provides positives and negatives of risk-based approaches. It notes that “uptake and bioaccumulation of chemicals through the food chain are considered on a case-by-case basis”, and that most sediment guidelines are “interim” due to data limitations.

Krantzberg G. 1995. Using the burden of evidence approach for sediment management; Case study: Collingwood Harbour. pp. 365-395 In: Munawar M, Edsall T, Leach J. (eds.), The Lake Huron Ecosystem: Ecology, Fisheries and Management. Ecovision World Monograph Series, SPB Publishing, Amsterdam, Netherlands.

The LEL (Lowest Effect Level) was used to separate areas that are contaminated and require further assessment (one or more sediment contaminants above the LEL) from those that did not (all sediment contaminant concentrations below the LEL). A WOE approach was used incorporating the following LO E: sediment chemistry, toxicity, benthic community structure, bioaccumulation.

Krantzberg G, Zarull MA, Hartig JH. 2000a. Sediment management: Deciding when to intervene. Environ Sci Technol 34: 22A-27A.

This publication notes that “knowledge of chemistry alone is insufficient” for assessing contaminated sediments. Additional LOE include: benthic community structure; laboratory toxicity tests; bioaccumulation and biomagnification information; knowledge of site stability; and, physico-chemical sediment properties. A matrix of data interpretations tools relating to different ecological threats associated with sediment contaminants is provided. A decision-making matrix based on different LOE is also provided. Effectively this paper espouses the SQT approach including decision matrix, but including additional LOE. A more detailed version of this publication can be found at: http://www.ijc.org/php/publications/html/sedwkshp/

Krantzberg G, Hartig J, Maynard L, Burch K, Ancheta C. 1999. Deciding When to Intervene: Data Interpretation Tools for Making Sediment Management Decisions Beyond Source Control. Sediment Priority Action Committee, Great Lakes Water Quality Board).

Krantzberg G, Reynoldson T, Jaagumagi R, Bedard D, Painter S, Boyd D, Pawson T. 2000b. SEDS: setting environmental decisions for sediment, a decision-making tool for sediment management. Aquat Ecosyst Health Manage 3: 387-396.

This publication builds on the senior author's publication that same year in Environ Sci Technol and the associated 1999 report, to recommend a "pragmatic decision-making framework". A stepwise approach to sediment management decisions is provided (Figure 1) along with a decision matrix (Table 2, above). A WOE approach is encouraged incorporating: sediment chemistry; bioaccumulation; sublethal or lethal toxicity; community structure; and, sediment stability. Recommendations are provided regarding: survey design and sampling; reference sites; and, data interpretation.

Lackey RT. 1997. Ecological risk assessment: Use, abuse, and alternatives. Environ Manage 21: 808-812.

This paper argues that ERA has a legitimate, appropriate, but limited role in science, policy analysis, and policy implementation. Its misuses and abuses are discussed and other possible approaches are suggested: benefits analysis; and, ecological alternatives assessment. The fact that ERA is "merely a tool in the decision-making process" is emphasized repeatedly.

Landis WG, Duncan PB, Hayes EH, Markiewicz AJ, Thomas JF. 2004. A regional retrospective assessment of the potential stressors causing the decline of the Cherry Point Pacific Herring run. Human Ecol Risk Assess 10: 271-298.

This paper focuses on a single ecological receptor but a variety of stressors: chemical, biotic, and abiotic. It proposes a different approach to retrospective relative risk than conventional regional risk assessments. It is based on the relative risk model previously used by the authors to integrate impacts due to a variety of stressors on a regional scale (as documented in various literature publications). This retrospective analysis is compared to WOE and other approaches to establish causality.

Luftig SD. 1999. Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites. Memorandum to Superfund National Policy Managers. October 7, 1999, Office of Solid Waste and Emergency Response, USEPA, Washington, DC, USA.

This document provides managers with "six principles to consider when making ecological risk management decisions": reduction of ecological risks; coordination with Trustees (federal, tribal and state stakeholders); using site-specific RA to support cleanup decision; characterizing site risk; communicating risk to the public; remediating unacceptable ecorisks. Four key questions are explained: What ecological receptors should be protected? Is there an unacceptable ecological risk at the site? Will the cleanup cause more ecological harm than the current site contamination? What cleanup levels are protective?

MacDonald DA, Matta MB, Field LJ, Munn MD. 1997. The Coastal Resource Coordinator's Bioassessment Manual. Report No. Hazmat 93-1. National Oceanic and Atmospheric Administration, Seattle, WA, USA.

This document outlines NOAA's approach to assessing contaminated sediments. Effectively, this is an SQT approach that includes the following LOE in addition to sediment chemistry: benthic infaunal community structure, bioaccumulation, biomarkers, and toxicity tests. This WOE approach fits within an ERA framework, though ERA is not explicitly mentioned in the document.

MacDonald DD, Ingersoll CG. 2002a. A Guidance Manual to Support the Assessment of Contaminated Sediments in Freshwater Ecosystems. Volume I: An Ecosystem-Based Framework for Assessing and Managing Contaminated Sediments. EPA-905-B02-001-A, USEPA Great Lakes National Program, Office, Chicago, IL, USA.
http://www.cerc.usgs.gov/pubs/sedtox/guidance_manual.htm.

MacDonald DD, Ingersoll CG. 2002b. Guidance Manual to Support the Assessment of Contaminated Sediments In Freshwater Ecosystems. Volume II: Design and Implementation of Sediment Quality Investigations. EPA-905-B02-001-B, USEPA Great Lakes National Program Office, Chicago, IL, USA. http://www.cerc.usgs.gov/pubs/sedtox/guidance_manual.htm.

Ingersoll CG, MacDonald DD. 2002. Guidance Manual to Support the Assessment of Contaminated Sediments in Freshwater Ecosystems. Volume III: Interpretation of the Results of Sediment Quality Investigations. EPA-905-B02-001-C, USEPA Great Lakes National Program Office, Chicago, IL, USA. http://www.cerc.usgs.gov/pubs/sedtox/guidance_manual.htm.

These three companion reports together provide an integrated ecosystem-based framework for assessing and managing sediment quality in freshwater ecosystems. Effectively these documents provide for WOE assessments within an ERA framework. Four separate LOE are "commonly used to assess contaminated sediments": sediment and pore water chemistry data; sediment toxicity data; benthic invertebrate community structure data; and, bioaccumulation data. An ancillary LOE is fish health assessment. The approach begins with sediment quality assessment guidelines (SQAGs) to determine when adverse ecological effects are unlikely or likely. It is noted that SQAGs "should be used together with other assessment tools to support comprehensive assessments of sediment quality conditions."

Menzie C, Henning MH, Cura J, Finkelstein K, Gentile J, Maughan J, Mitchell D, Petron S, Potocki B, Svirsky S, Tyler P. 1996. Special report of the Massachusetts weight-of-evidence workgroup: a weight-of-evidence approach for evaluating ecological risks. Human Ecol Risk Assess 2: 277-304.

The authors approach WOE in an ERA context, defining WOE as the approach by which measurement endpoints are related to assessment endpoints based on weight, magnitude and concurrence, to determine risk of harm. Measurement endpoints are weighted by stakeholders based on best professional judgment, relative to the assessment endpoint, the study's quality and design, and the confidence in the measurement. Specifically, 10 separate judging attributes are used, which may be equal or weighted: degree of association; stressor/response; utility of measure; data quality; site specificity; sensitivity; spatial representativeness; temporal representativeness; quantitative measure; and, standard measure. The measurement endpoint weight is derived by summing the scored, weighted scaling values and dividing by 5. Results are presented in a tabular decision matrix format for evaluation and interpretation. This approach also allows for varying degrees of BPJ, as the quantitative matrix approach for weighting attributes can also be replaced by a more qualitative weighting system. A subsequent paper by Johnston *et al.* (2002. Weighing the evidence of ecological risk from chemical contamination in the estuarine environment adjacent to the Portsmouth Naval Shipyard, Kittery, Maine, USA. Environ Toxicol Chem 21: 182-194) found that the Menzie *et al.* (1996) approach, with a few "improvements", allowed for appropriate LOE weightings to arrive at reasonable WOE conclusions.

OMEE. 1996a. Guidance on Site Specific Risk Assessment for Use at Contaminated Sites in Ontario. Ontario Ministry of Environment and Energy, Standards Development Branch.

This document provides general guidance for both ERA and HHRA for site clean- ups in Ontario. It provides: a general introduction to RA; general guidance for HHRA: and, a basic framework for ERA. The CCME (1996) framework for conducting ERAs is accepted with its three levels of investigation: screening level; preliminary quantitative; and, detailed quantitative.

OMEE. 1996b. Guideline for Use at Contaminated Sites in Ontario. Ontario Ministry of Environment and Energy. ISBN 0-7778-4052-9.

This document focuses on background contaminant levels, and site specific RA. Effectively it recommends tiering beginning with chemical guideline comparisons and proceeding to RA. There is no mention of WOE.

Porebski LM, Doe KG, Zajdlik BA, Lee D, Pocklington P, Osborne JM. 1999. Evaluating the techniques for a tiered testing approach to dredged sediment assessment – a study over a metal concentration gradient. Environ Toxicol Chem 18: 2600-2610.

This publication documents the use of the SQT approach for dredged sediment assessment in New Brunswick. SQGs at the threshold effect level (TEL) performed well as "levels below which unacceptable biological effects were unlikely to occur." Predicted effect level (PEL) SQGs were less predictive. A WOE approach based on three lines of evidence (chemistry, toxicity, benthic community structure) is recommended for contaminated sediment assessments. This work comprised a demonstration project for the Canadian Disposal at Sea Program.

Reynoldson TB, Smith EP, Bailer AJ. 2002a. A comparison of three weight-of evidence approaches for integrating sediment contamination data within and across lines of evidence. Human Ecol Risk Assess 8: 1613-1624.

This paper compared three strategies for combining information within and among different sediment assessment LOE: multivariate clustering; meta-analysis to pool empirically derived P-values; quantitative estimation of probability derived from odds ratios. Critical issues in all cases were: defining appropriate reference conditions; defining an “impact” relative to reference conditions; use of distance from the reference distribution to define effect measures. Each of the three strategies had advantages and disadvantages; presently there is not a single ideal method for WOE integration. The authors urge researchers to ensure transparency and completeness in their WOE integrations.

Reynoldson TB, Thompson SP, Milani D. 2002b. Integrating multiple toxicological endpoints in a decision-making framework for contaminated sediments. Human Ecol. Risk Assess. 8: 1569-1584.

This paper used toxicity data for four different test organisms exposed to sediments from 220 Great Lakes reference sites to establish three categories of responses (mean ± SD). It then examined three different approaches for integrating toxicity information within that LOE, both score based and multivariate statistical. The most appropriate method was multivariate ordination: least subjective; quantitative; and provided appropriate weighting based on the variation observed within reference sites.

Risk Assessment Forum. 1998. Guidelines for Ecological Risk Assessment. Federal Register 63(93): 26846-26924. EPA/600/R-95/002F, Washington DC, USA.

These Guidelines are primarily intended for EPA personnel, but may be used outside the Agency. They replace the earlier 1992 ERA Framework document, expanding on and modifying that original document. The three primary phases of ERA are discussed in detail: problem formulation, analysis, and risk characterization. There is no description of WOE; the focus is on interpreting independent LOE.

Risk Assessment Forum. 2003. Framework for Cumulative Risk Assessment. USEPA, Washington, DC, USA. EPA/630/P-02/001F.

This document marks the start of USEPA’s attempts to address cumulative risk from complex exposures. It lays out broad areas for possible analyses, with particular focus on pesticides. There are three differences from the typical ERA and HHRA framework: the focus is on multiple, combined stressors rather than individual stressors; there is increased focus on specific populations affected rather than on hypothetical receptors; and, nonchemical stressors are considered to a much greater degree.

Stronkhorst J. 2003. Ecotoxicological Effects of Dutch Harbour Sediments. The Development of an Effects-Based Assessment Framework to Regulate the Disposal of Dredged Materials in Coastal Waters of the Netherlands. Ph.D. Thesis, Institute for Environmental Studies, Vrije Universiteit Amsterdam, Netherlands.

This Dissertation involved two individual LOE (toxicity tests and chemical analyses) integrated in a WOE approach comprising an effects-based framework to more accurately identify hazardous dredged material prior to disposal. A hazard quotient was defined that weights all LOE from the measured endpoints. The importance of WOE assessments for dredged materials in the Netherlands and in the rest of Europe is emphasized. There is no mention of ERA.

Suter GW II. 1996. Risk Characterization for Ecological Risk Assessment of Contaminated Sites. Office of Environmental Management, US Department of Energy, Oak Ridge, TN, USA. ES/ER/TM-20.

This document emphasizes the fact that ERA "is performed by weight of evidence". LOE described include "chemical analyses, toxicity tests, biological surveys, and biomarkers". An approach for estimating risks based on individual LOE and then combining them in a WOE assessment is described. The importance of evaluating the relationship between measurement and assessment endpoints is documented. The LOE based on chemical analyses provides an initial screening. Toxicity in that LOE is not significant if "the effects relative to controls are less than 20%… and the effects are not significantly different from the controls." Biomarkers are used to support other LOE. Recommendations are provided for weighting different LOE in the event that there is not agreement among the different LOE. The point is made that LOE that contradict the findings of field surveys "are clearly incorrect".

Suter II GW, Norton SB, Cormier SM. 2002. A methodology for inferring the causes of observed impairments in aquatic ecosystems. Environ Toxicol Chem 21: 1101-1111.

This paper describes a methodology for inferring the cause(s) of impaired aquatic ecosystems. The methodology is based on BPJ within a formalized, logical WOE analysis (the authors term this a "strength-of-evidence" analysis), and includes reconsideration of the causal analysis when no clear cause is evident. The methodology has been successfully applied to a relatively simple situation in a river in Ohio, USA, although it has not been applied to more complex systems.

USEPA. 1997. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. Interim Final. Solid Waste and Emergency Response, Washington, DC, USA. EPA 540-R-97-006.

Guidance is provided for designing and conducting technically-defensible ERAs for the Superfund Program. This guidance is based on the Risk Assessment Forum's guidance documents but provides much greater detail applicable to Superfund Sites for use by risk managers at those sites. There is no mention of WOE.

USEPA. 1998. EPA's Contaminated Sediment Management Strategy. Office of Water, U.S. Environmental Protection Agency, Washington, DC, USA. EPA-823-R-98-001. http://www.epa.gov/waterscience/cs/stratndx.html.

This document describes USEPA's four strategic goals relative to contaminated sediments: (1) prevent increases in volume; (2) reduce existing volumes; (3) manage dredging and dredged material disposal in an environmentally sound manner; (4) develop scientifically sound sediment management tools. USEPA uses the following tools for sediment assessment in a tiered testing, WOE framework: numerical sediment quality criteria and biological testing methods (toxicity [for freshwater - *Hyalella azteca* and *Chironomus tentans* 10-d tests] and bioaccumulation tests [for freshwater - *Lumbriculus* variegates 28-d test]). Benthic community structure data are also considered. Sediment quality criteria can be used for ranking contaminated sediments and identifying hotspots. If impacts of a remedial alternative will "cause more environmental harm than leaving the contaminants in place, EPA may not proceed with a cleanup at that time."

USEPA. 2003. Generic Ecological Assessment Endpoints (GEAEs) for Ecological Risk Assessment. Risk Assessment Forum, U.S. Environmental Protection Agency, Washington, DC, USA. EPA/630/P-02/004F. http://cfpub1.epa.gov/ncea/cfm/recordisplay.cfm?deid=55131.

This document describes a set of generic ecological assessment endpoints (GEAEs) that can be considered and adapted for specific ERAs. Table 2-1 provides a list of the GEAEs. GEAEs of particular relevance to sediment assessments are as follows: individual organisms – survival, fecundity, growth; populations - extirpation, abundance; communities - taxa richness, abundance, physical structure. Though this document is intended primarily for use within USEPA, it clearly has utility outside the Agency.

US Policy Committee. 2001. Restoring United States Areas of Concern: Delisting Principles and Guidelines. http://www.epa.gov/glnpo/aoc/delist.html.

This document outlines principles for delisting as well as requirements for removal of a beneficial use impairment including: restoration of the beneficial use; demonstration that impairment is due to natural causes; the impairment is of much wider geographic scope than the AOCs; the impairment is caused by sources outside the AOCs. The importance of monitoring data to ensure success is emphasized. Appendix 4 summarizes delisting guidelines and criteria for: fish and wildlife consumption (standards/objectives/guidelines not exceeded, no public health advisories); degraded fish and wildlife populations (healthy, self-sustaining communities at the expected level of abundance or, in the absence of community structure data, no significant water or sediment toxicity); fish tumors or other deformities (not elevated above background); bird or animal deformities or reproductive problems (not elevated above background); degradation of benthos (not significantly different from comparable reference sites or, in the absence of community structure data, no significant sediment toxicity); restrictions on dredging activities (sediment contaminants do not exceed standards, criteria or guidelines).

Wenning RJ, Ingersoll CG (eds). 2002. Use of Sediment Quality Guidelines and Related Tools for the Assessment of Contaminated Sediments. Executive Summary Booklet of a SETAC Pellston Workshop. SETAC Press, Pensacola, FL, USA.
http://www.setac.org/files/SQGSummary.pdf.

A SETAC (Society of Environmental Toxicology and Chemistry) Pellston Workshop held August 2002 summarized the state of the science regarding the development and use of SQGs for assessing contaminated sediments. This Workshop Summary Document summarizes: the scientific underpinnings of SQGs; their predictive ability; other sediment assessment tools; the use of SQGs in sediment assessment frameworks; and, the use of SQGs and other tools for assessing sediments in different aquatic environments. There was agreement at the Workshop, which involved all major developers of SQGs and a wide range of international practitioners, that:

- SQGs have two primary roles – to address relative spatial and/or temporal patterns of contamination, including probable no effect and possible effect concentrations; and, for primary decision-making "in cases of simple contamination where adverse biological effects are likely… when the costs of further investigation outweigh the costs of remediation, and there is agreement to act further instead of conducting further investigations."
- SQGs have secondary roles as part of an ecological risk assessment (ERA) and/or in a tiered assessment scheme, in conjunction with other tools. Such secondary roles include: determining the condition of populations and communities; estimating ecological risks; screening the suitability of a proposed use or development; assessing impacts of sediment dredging or management; remediation and restoration objectives; long-term post-remediation monitoring.
- SQGs have no role in evaluating human health risks or biomagnification, nor in determining sediment stability and transport.
- SQGs are generally appropriate for lakes and ponds or low gradient rivers and streams.
- Site-specific SQGs may be required for estuaries.
- SQGs may not be appropriate for depositional wetlands or highly modified systems.
- SQGs are not appropriate for non-depositional and erosional environments.

Appendix 2 - State of the Science Overview and Jurisdictional Scan (2004)

1.0 Introduction

This Appendix provides both a critical "State-of-the-Science" summary of ERA (ecological risk assessment) processes specific to contaminated sediment assessment, and a jurisdictional (primarily Canadian but also some US jurisdictions) scan of existing practices. The jurisdictional scan was done to ensure that the proposed Framework would not significantly conflict with the ways other Canadian jurisdictions deal with contaminated sediments.

2.0 State of the Science Summary of ERA Processes Specific to Contaminated Sediment Assessment

2.1 Sediment Quality Guidelines (SQGs)

CCME have established Canadian ISQGs (interim sediment quality guidelines), which are numerical values recommended to support and maintain aquatic life, based on the biological effects of sediment-associated substances. Two levels are set: the Threshold Effect Level (TEL) – the value at which there is no toxicological effect; and the Probable Effect Level (PEL) – the value at which an adverse biological effect is expected. Other jurisdictions in Canada and the United States that have developed their own SQGs, have set two levels of SQGs similar to CCME, specifically a lower level below which adverse effects are unlikely, and a higher level above which adverse effects are likely (but not certain). The region between these two levels is considered an area within which adverse effects may or may not occur.

Highest reliance and reliability is placed on the lower SQG level, predicting the absence of adverse effects. Typically, failure to exceed this lower level SQG does not require further investigations, and a determination of negligible risk can be made. However, exceedences of this lower level SQG require additional investigations up to and including ERA.

2.2 Ecological Risk Assessment (ERA)

Ecological risk assessment (ERA) is a process that evaluates the likelihood or probability for adverse ecological effects occurring as a result of exposure to contaminants or other stressors. It comprises a framework for gathering data and evaluating their sufficiency for decision-making. And, it recognizes, considers, and reports uncertainties in estimating adverse effects of stressors (because of the diverse and complex nature of ecosystems, uncertainty can never be fully characterized or estimated).

Adverse ecological effects occur only when a receptor is exposed (e.g., via inhalation, ingestion, or contact) to a stressor under realistic conditions at a sufficient concentration and after a sufficient duration. In the case of a chemical contaminant, risk of an adverse effect can only result with such exposure to the bioavailable fraction of that contaminant. Thus, the risk of an adverse effect is determined not only by the presence of a stressor, but also by its form, concentration and the duration of exposure under realistic conditions.

A critical component of the ERA is the development of a conceptual site model (CSM). A CSM is a three-dimensional description of a site and its environment that represents what is known (or suspected) about contaminant sources as well as exposure routes to aquatic and other biota. The CSM is developed at the beginning of the ERA and is further developed as the ERA proceeds. It can be simple or complex, depending on the site and site-specific objectives.

A typical ERA process consists of three steps in the U.S.: problem formulation, analysis (exposure and effects characterization), and risk characterization (U.S. EPA, 1998), and four steps in Canada and in the European Union (EU): hazard identification, dose-response assessment, exposure assessment, and risk characterization (CCME, 1996). Key information required for an ERA includes: the emissions, pathways and rates of movement of contaminants in the environment; and, information on the relationship between contaminant concentrations and the incidence and/or severity of adverse effects.

The first step (i.e., Problem Formulation/Hazard Identification) involves articulating the ERA purpose, defining the problem, identifying stressors that could cause adverse effects, and determining a plan for the subsequent analysis and risk characterization. Environmental quality guidelines (EQGs) such as sediment quality guidelines (SQGs) are typically used in this step to identify stressors that could cause an adverse effect (i.e., stressors or contaminants of concern). ERA then moves on to the next step (i.e., Exposure and Effects Characterization) to collect and assess data to determine how exposure to stressors is likely to occur (i.e., exposure characterization) and under this exposure what are the potential and types of adverse effects that may occur (i.e., effects characterization). Different types of information, various assumptions and their uncertainties, and different types of models or data interpretation may be required in this step. The exposure and effects profiles are then integrated in the final Risk Characterization step to estimate the incidence and severity of adverse effects likely to occur. To reduce the uncertainty, risk characterization generally builds the final risk estimates upon different lines of evidence (LOE), using a weight-of-evidence (WOE) approach. Lines of evidence may include evidence from: (1) laboratory studies; (2) field observations; (3) model predictions; and (4) professional judgment.

The final ERA product is an estimate of the probability of ecological effects that may occur or are occurring. In addition, the uncertainty (or degree of confidence) in the risk estimates is indicated, LOE supporting the risk estimates are cited, and the significance of observed/predicted adverse ecological effects is discussed. Stakeholders can then use the report for decision-making.

ERAs of contaminated sediments should, at a minimum, include the following information:

- Identification and quantification of sediment contaminants including their vertical and horizontal distribution;
- Sources of historical and current contamination to the sediment (both point and nonpoint sources);
- Physical, chemical and biological processes affecting fate, transport and bioavailability of sediment contaminants including sediment characteristics (e.g., proportion of sand, silt and clay);
- Non-contaminant stressors (biotic or abiotic);
- Ecological exposure pathways for the contaminants and other stressors via one or more CSMs;
- Current and potential future ecological risks;
- Baseline data that can be used for monitoring of future conditions.

2.3 Brownfields in Ontario – Sediment Assessment

The new Ontario Brownfields Draft Regulation – Relating to the Filing of a Record of Site Condition (http://www.e-laws.gov.on.ca/DBLaws/Source/Regs/English/2004/ R04153_e.htm - OMOE, 2004a), which came into force October 1, 2004 and the associated Soil, Ground Water and Sediment Standards dated March 9, 2004 (http://www.ene.gov.on.ca/envision/techdocs/4697e.pdf - OMOE, 2004b) provide an excellent example of the use of both SQGs and ERA for assessing contaminated sediments in a tiered framework. [Note that the Environmental Protection Act, Ontario Regulation 153/04 "Records of Site Condition – Part XV.1 of the Environmental Protection Act" and any amendments to the RSC Regulation or Act should be referred to in order to understand what is in the legislation and regulations and where necessary, seek legal counsel.] Dealing with contaminated sediments (defined as "the soil, to a maximum depth of 0.15 meters, located at the base of a water body") requires the following (from the Regulation and Terri Bulman, pers. comm., July 12, 2004 and December 15, 2004):

1. The site is initially assessed based on existing knowledge including review of records, a site visit and interviews - a Phase 1 Environmental Site Assessment conducted according to Ontario Regulation 153/04 with reference to CAN/CSA Z768-01 published by the Canadian Standards Association and dated November 2001, as it may be amended from time to time.
2. If there is evidence that the site may be contaminated as a result of industrial or commercial activities, then a Qualified Person (QP, as defined in the Regulation) may require sediment sampling as part of a Phase 2 Environmental Site Assessment conducted according to Ontario Regulation 153/04 with reference to CAN/CSA Z769-00 published by the Canadian Standards Association and dated March 2000, as it may be amended from time to time. There are no explicit requirements to sample sediment in the Regulation, but if sediment is sampled the results must meet the standards. There are no requirements in the Regulation to sample off site. The sediment standards in

the Regulation are based on OMOE's 1993 Sediment Guidelines and are contained in OMOE (2004b). Proponents can also develop site-specific standards for sediments using the risk assessment process described in the Regulation.

3. A risk assessment is explicitly required if a standard is exceeded, and must include an assessment of both human health and ecological risk. A risk assessment is not explicitly required if a contaminant is found but not listed, at the QP's discretion, however the QP must certify that there is no evidence of contaminants that are likely to interfere with the proposed property use. If the site is risk assessed, the QP for risk assessment must make a statement about whether off site receptors are likely to be exposed to concentrations above the generic standards. Statements relating to off site receptors could involve sediment, even if the on-site investigations/risk assessment did not.
4. Information from points 1 and 2 above would be used in developing the Problem Formulation component of an ERA and determining the level of ERA required. This information must be provided to OMOE prior to submission of a risk assessment in a pre-submission form (PSF) together with diagrams and explanatory text as prescribed in Ontario Regulation 153/04 Schedule C Section 3 and Table 1 sections 4 (for human health) and 5 (for ecological risk). All ERA should follow the CCME framework.Within that framework, some approaches to conducting ERAs have been identified by OMOE as being eligible for special treatment under the OMOE review timelines. OMOE provides comment on the general approach and scope of the proposed risk assessment after considering the information provided in the PSF.
5. An ERA approach that is eligible for the "short" (8 week) review timeline is termed "limited scope" and is defined as one of the following:

   - An assessment for an area which would normally have to meet full depth background site condition standards due to classification under Ontario Regulation 153/04 Section 41 as an environmentally sensitive area, but which meets, and can be shown to be adequately protected by, the full depth generic potable site condition standards.
   - A property which is subject only to groundwater contamination from offsite sources (flow-through contamination).
   - An assessment conducted using the same models and assumptions used by the Ministry to develop the full depth generic site condition standards. This would include using the component values for individual pathways published by OMOE in the document "Rationale for the Development and Application of Generic Soil, Groundwater and Sediment Criteria for Use at Contaminated Sites in Ontario" (December 1996).

6. Estimation of natural background conditions when natural background site conditions exceed the full depth background site condition standards is also considered a form of risk assessment under the Regulation which is eligible for the "short" (8 week) review timeline. In this case, natural local background data may be shown appropriate to replace the full depth background site condition standards at that location provided

they meet the effects-based full depth generic site condition standards. This assessment is effectively a sampling program.

7. An ERA approach that is eligible for the "long" (22 week) review timeline is termed "new science" and is defined as one in which a standard needs to be developed using new toxicity data, (for example for one or more unlisted contaminants), proprietary computer modeling is required, or probabilistic modeling is used for exposure assessments. These additional components require detailed review by OMOE and will require a longer timeline for review.
8. A final ERA approach that is eligible for the "long" (22 week) review timeline is termed ERA is termed "wider area of abatement", which means that the site is part of a larger geographic area of contamination.
9. Distinctions between ERA types as noted above arise from the need to describe types of risk assessment that fall outside the 'norm' and therefore outside the typical review timeline of 16 weeks. The limited scope risk assessments described in item 5 will take less time for OMOE to review because they will either qualitatively justify the existing standards or they will consider a very limited set of pathways. The new science and wide area of abatement risk assessments, described in items 7 and 8, will take more time to review either because of the need to consider additional material (toxicity data, models) or the need to involve the district and community.
10. All ERAs should still make use of, and refer to, the CCME hierarchy (screening, preliminary, detailed) related to either individual components of a risk assessment (individual contaminants or pathways) or the entire ERA. The Regulation does not explicitly reference the CCME document but provides for qualitative and quantitative approaches. A forthcoming Guidance Document will make explicit reference to the CCME approach.

3.0 Jurisdictional Scan of Existing Practices (Canada and US)

3.1 US

The US National Oceanic and Atmospheric Administration (NOAA) uses a bioassessment approach to assess contaminated sediments (MacDonald *et al.*, 1997). Effectively, this is an SQT (Sediment Quality Triad) approach that includes the following LOE in addition to sediment chemistry: benthic infaunal community structure; bioaccumulation; biomarkers; and, toxicity tests. The approach fits within an ERA framework.

Three primary methods (the SQT) are used by USEPA for ecological assessment of contaminated sediments (Griffith *et al.*, 2004): sediment criteria; ambient toxicity assessments; and, bioassessment of macroinvertebrates. Chemical criteria are derived using numerical methods from compilations of toxicity test data, such as species sensitivity distributions. Ambient toxicity assessments involve tests with standard species such as, in freshwater, the amphipod Hyalella azteca and/or the chironomid *Chironomus riparius*. Bioassessments enumerate benthic macroinvertebrate assemblages, calculate metrics that describe the assemblages, and sum the metric scores to produce indices of biotic integrity.

The USEPA Superfund Program encourages the use of multiple lines of evidence in assessing risks and in making cleanup decisions (Stephen Ells, pers. comm., July 12, 2004). They presently have no guidance on a specific WOE approach, but are considering developing a fact sheet on WOE for use at contaminated sediment sites. The USEPA Great Lakes National Program Office (Scott Ireland, pers. comm., July 8, 2004) is involved in non-regulatory assessments and generally conducts SQT analyses (sediment chemistry, toxicity and benthic community structure). They are developing a framework that can be applied to those US AOCs with contaminated sediment-related impairments. For AOCs in commercial ship harbors there may be no rationale for collection of benthic invertebrates "when there is essentially no habitat for them" (Scott Ireland, pers. comm., July 8, 2004). Their generic guidance delisting criteria for AOCs are available at: http://www.epa.gov/glnpo/aoc/delist.html, and give the States flexibility to work in community goals. As the guidance states, "this document does not define explicit targets (beyond any articulated jurisdictional values incorporated here by reference), it does offer some criteria for target-setting. The development of specific targets is a separate process, and is beyond the scope of this paper. Delisting targets should be premised on local goals and related environmental objectives for the watershed; they should be consistent with the applicable federal and state regulations, objectives, guidelines, standards and policies, when available, and the principles and objectives embodied in Annex 2 and supporting parts of the GLWQA." The IJC SEDPAC group decided that (Dave Cowgill, pers. comm., July 12, 2004) "describing the decision-making processes that are actually used on both sides of the border (noting commonalities) was more useful than developing one process that no one actually uses."

For reports are available related to an integrated ecosystem-based framework for assessing and managing sediment quality in freshwater ecosystems (MacDonald and Ingersoll 2002a,b; Ingersoll and MacDonald 2002; MacDonald *et al.* 2002). Effectively these documents provide for WOE assessments within an ERA framework. Four separate LOE are "commonly used to assess contaminated sediments": sediment and pore water chemistry data; sediment toxicity data; benthic invertebrate community structure data; and, bioaccumulation data. An ancillary LOE is fish health assessment. The approach begins with sediment quality assessment guidelines (SQAGs) to determine when adverse ecological effects are unlikely or likely. SQAGs "should be used together with other assessment tools to support comprehensive assessments of sediment quality conditions."

The Minnesota Pollution Control Agency (MPCA; Judy Crane, pers. comm., July 6, 2004 and August 30, 2004) has prioritized metrics for a number of ecosystem health indicators as part of a WOE approach for assessing sediment quality conditions in the St. Louis AOC. Complete details of these indicators are available in Table 5 at http://www.pca.state.mn.us/water/sediments/sqt-tables.pdf (Crane *et al.* 2000). Specifically, their focus is on: the SQT; sediment chemistry including sediment quality target values (mean probable effect concentration [PEC] quotient, % incidence of predicted toxicity); acute and chronic sediment toxicity tests (primarily 10-d survival and growth with *Hyalella azteca* and *Chironomus tentans*); benthic invertebrate community status (diversity, evenness, indicator species); physical characteristics; water chemistry; tissue residues (in resident populations and in laboratory bioaccumulation tests with *Lumbriculus variegatus*); porewater toxicity; biomarkers in fish; and water column and elutriate toxicity. They have identified Level I and Level II sediment quality targets as follows (Crane *et al.* 2000, 2002): Level I sediment quality targets are intended to identify contaminant concentrations below which harmful effects on sediment-dwelling organisms are unlikely to be observed; Level II sediment quality targets are intended to identify contaminant concentrations above which harmful effects on sediment-dwelling organisms are likely to be frequently or always observed. Crane and MacDonald (2003) provide recommended applications for these sediment quality targets. MPCA do not find benthic community data to be a driving factor in moving a site to remediation where other factors (sediment chemistry, physical parameters such as particle size, sediment toxicity, mean PEC-Qs, and sediment bioaccumulation) have been more useful for assessing and delineating the boundaries of contaminated sediment areas, but they and others have conducted SQT investigations in the lower St. Louis River AOC. Though they use a WOE approach, they find the sediment and toxicity LOE to be "more useful for assessing and delineating the boundaries of contaminated sediment areas". Additional information is available at: http://www.pca.state.mn.us/water/sediments/studiesstlouis.html.

3.2 Canada

3.2.1 Nationwide

The Government of Canada's Contaminated Sites Management Working Group (CSMWG) defines a contaminated site as one at which substances occur at concentrations: (1) above background levels and pose, or are likely to pose, an immediate or long-term hazard to human health or the environment; or, (2) exceed levels specified in policies and regulations (http://www.ec.gc.ca/etad/csmwg). For sediments, the CCME ISQGs provide the basis for determining whether or not a site is contaminated. Since the CCME has no authority to implement or enforce legislation, it is up to each province and territory to decide whether or not to adopt the CCME ISQGs.

Environment Canada's Marine Protection Division (Linda Porebski, pers. comm., July 7, 2004) administers Canada's international obligations under the London Convention and the 1996 Protocol on Disposal at Sea and controls a permit system under CEPA (the Canadian Environmental Protection Act) that follows international waste assessment guidance. They have traditionally said no to the disposal of any contaminated sediments at sea but have examined WOE approaches for use in future decision-making (Porebski *et al.*, 1999) and are now considering their incorporation into a risk assessment framework. They have completed a small workshop on tools for contaminated sediment management (proceedings not yet available), and are planning a second one in 2005 on decision-making for contaminated sediments. They favor a unified decision-making framework for both marine and freshwater sediments.

3.2.2 Great Lakes

The Great Lakes Regional Office of the IJC (Gail Krantzberg, pers. comm., July 7, 2004) espouses a WOE approach based on the SQT LOE plus bioaccumulation. Effectively this is a risk assessment approach. Examples of work done in the Great Lakes that has involved the different LOE in a WOE approach, but which should not be considered formally endorsed by the IJC, are available in a series of publications: Krantzberg (1995); Zarull *et al.* (1999, 2002); Krantzberg *et al.* (2000a,b; 2001).

3.2.3 British Columbia

The BC Ministry of Water, Air and Lands Protection (WLAP) uses the SQT approach augmented with bioaccumulation testing (Mike Macfarlane, pers. comm., June 17, 2004). As of July 2004, certificates of compliance can be obtained for both criteria-based and risk-based approaches. Previously, risk-based approaches only provided conditional certificates of compliance. As a result, the use of ERA for contaminated sediment assessment is expected to increase greatly in BC. WLAP has specific guidance for screening-level ERA (BCE, 1998). For more detailed ERA, they recommend the model used in the Calcasieu Estuary in the US (MacDonald *et al.*, 2002). WLAP has developed their own sediment quality criteria (SQC) for sensitive and typical sites, for both freshwater and marine/estuarine sediments. These SQC were developed to correspond with the narrative goals of a 20% or 50% probability of an amphipod EC50 and are consistent with the risk thresholds advocated in provincial ERA guidance. They have also developed tissue residue guidelines for: methyl mercury; total PCBs; DDD, DDE, DDT; and, 2,3,7,8-TCDD. These SQC and guidelines are applicable to provincial and joint provincial/federal sites. WLAP expects increased involvement of trained biologists in the field on sediment projects, rather than sole reliance on sediment chemistry. Specific examples of where risk assessment/biology are required include:

- Detailed site investigations require the preparation of a problem formulation to examine routes of exposure, appropriate receptors, etc. If listed bioaccumulative substances (i.e., substances that biomagnify) are present, then the sediment assessment will require explicit consideration of bioaccumulation in the food chain. It is not acceptable to ignore bioaccumulation, even if there is no direct toxicity to benthos.
- The decision to classify a site as "sensitive" or "typical" is dependent on the habitat available at the site, not the land use. For example, an industrial site located on a wetland is sensitive, irrespective of its history. Site classification should be conducted by a biologist, based on site-specific information.
- There will be increasing reliance on the toxicity testing and benthic community LOE. Reliance on chemistry alone, leading to a decision to dredge everything out is not likely to be approved. Very few sites actually require dredging for remedial purposes; it would be necessary to make a convincing case that the impacts from dredging would be less than the impacts from leaving the sediment in place.

- The use of chronic toxicity tests (e.g., 28-d amphipod survival, growth and reproduction tests) is recommended over acute toxicity tests.

WLAP would like to find a practical way to equate impacts from development related to habitat impacts with those from chemical impairment to obtain approval for the use of Confined Aquatic Disposal (CAD) facilities for some contaminated sediments. They plan (Mike Macfarlane, pers. comm., June 17, 2004) to update their screening-level ERA guidance (BCE, 1998) to include a sediment-specific module; for benthos, the P20 and P50 (probabilities) for an EC20 would be their risk benchmarks.

Supporting documentation related to the WLAP Director's Criteria for Managing Contaminated Sites in British Columbia are available at:
http://wlapww.gov.bc.ca/edp/endpa/contam_sites/standards_criteria/sed_criteria.pdf and
http://wlapww.gov.bc.ca/edp/endpa/contam_sites/standards_criteria/sed_criteria_tech_app.pdf.

The following four volumes of guidance are available at:
http://wlapww.gov.bc.ca/epd/epdpa/contam_sites/guidance/index.html

- Volume I – An Ecosystem-Based Framework for Assessing and Managing Contaminated Sites
- Volume II – Design and Implementation of Sediment Quality Investigations in Freshwater Ecosystems
- Volume III – Interpretation of the Results of Sediment Quality Investigations
- Volume IV – Supplemental Guidance on the Design and Implementation of Detailed Site Investigations in Marine and Estuarine Waters.

With the recent passage of the 4th stage amendment to the Contaminated Sites Regulation, the Director's Criteria for sediment per se have now been replaced by Schedule 9 of the Regulation: http://www.qp.gov.bc.ca/statreg/reg/E/EnvMgmt/EnvMgmt357_96/375_96.htm.

Environment Canada reviews sediment quality at contaminated sites pursuant to their mandate under the Fisheries Act subsection 36(3). They recommend an initial comparison of sediment chemistry with the CCME Environmental Quality Guidelines and/or applicable BC Sediment Guidelines or Criteria. If conditions for application of these criteria are met (e.g. TOC is between 1% and 5% for BC criteria), there are no bioaccumulative substances present, and if no parameter concentrations are exceeded, then no further assessment of sediment is required. Otherwise, proponents are encouraged to complete further assessment. This may include investigation of other LOE, consideration of guidelines/criteria from other jurisdictions, or development of site specific toxicity reference values (TRVs) (Jo-Ann Aldridge, pers. comm., September 9, 2004).

3.2.4 Prairie Provinces

Alberta and Saskatchewan have no specific policies or SQGs related to contaminated sediments and thus default to the CCME approach of any exceedences of ISQGs resulting in site-specific assessments (ERA) (SAIC Canada, 2002; Pritam Jain, pers. comm., July 23, 2004; Sam Ferris, pers. comm., August 20, 2004). Saskatchewan is presently working on "developing both a quantitative and qualitative approach to risk assessment of various environmental stressors, but have not published any document as yet" (Pritam Jain, pers. comm., July 23, 2004). Saskatchewan Environment has not "encountered instances of contaminated sediment clean-up in recent times" but, if they do so, they may use SQGs alone or other LOE such as sediment toxicity and/or biological surveys of affected bottom dwelling organisms; to date there has been no need to integrate different LOE into a WOE approach (Sam Ferris, pers. comm., August 20, 2004).

If CCME ISQGs are not available, Alberta will "investigate the existence of guidelines from other jurisdictions (e.g., USEPA) and/or literature on sediment toxicity testing (Anne-Marie Anderson, pers. comm., August 3, 2004). They rely solely on SQGs "when ambient conditions meet available guidelines" (see for example a recent report on PCBs in river sediments, at http://www3.gov.ab.ca/env/info/infocentre/PubDtl.cfm?ID=2129) and/or when "sediment quality falls within the natural regional background range (Anne-Marie Anderson, pers. comm., September 1, 2004). They recently used the SQT in Wabamum Lake, a suspected case of sediment contamination, where CCME ISQG, and in some cases PEL guidelines for several metals and PAH were exceeded. They also noted (Anne-Marie Anderson, pers. comm., August 3, 2004) "Depending on the nature of the contaminants (likelihood of biomagnification), body burdens along the food chains would be assessed" but "So far, contaminated sediments have not been a big issue in Alberta and ERA have been carried out seldom by Alberta Environment although they have, in some cases, been required from industry." Reports produced by Alberta Environment on Wabamum Lake are at http://www3.gov.ab.ca/env/water/reports/wabamunsum.html.

Manitoba has adopted a three-tier approach to assessing contaminated sediments (SAIC Canada, 2002). Tier 1 involves the CCME ISQGs. Tier 2 involves limited modification to the CCME ISQGs based on site-specific situations. Tier 3 involves a full ERA for cases where Tiers 1 and 2 do not provide definitive information.

3.2.5 Yukon, NWT and Nunavut

The NWT and Nunavut have adopted the CCME ISQGs and assessment approach (SAIC Canada, 2002). The Yukon does not currently have sediment standards, although they are investigating whether to incorporate such standards in the next amendments of their Contaminated Sites Regulation (Shannon Jensen, pers. comm., July 13, 2004). In the 2-3 cases where sediments have been an issue thus far, they have used their soil standards for comparison. Environment Canada has conducted chemistry and benthic invertebrate population studies in the Yukon (www.ec.gc.ca/bisy), including an SQT. They do not rely on the CCME ISQGs because they provide too many false positives for metals given the mineralized nature of many parts of the Yukon (Benoit Godin, pers. comm., August 13, 2004). Environment Canada relies on benthic invertebrate assessments to determine whether or not impacts are occurring in Yukon sediments, generally related to metals (though petroleum is also a contaminant of potential concern), and have investigated sequential extractions to assess metal bioavailability in sediments as well as attempting to develop regional reference sites; most of their information is collected “under the Control/Impact, Before and After Scenario” (Benoit Godin, pers. comm., August 13, 2004).

3.2.6 Ontario

The Ontario Ministry of Environment (OMOE) has developed its own SQGs. The Lowest Effect Level (LEL) is based on the concentration of a particular substance that is toxic to less than 5% of the population rather than the CCME Threshold Effect Levels (TEL) at which there is no toxic effects. However, while some LEL are higher than their corresponding TEL, many are similar to or lower. Exceedence of the SQGs typically results in the use of other LOE, up to and including an SQT. Recently, biomagnification has been added as an additional LOE to the SQT. OMOE explicitly identifies the need to consider the environmental fate of contaminants, their potential transport pathways, and potential toxic effects on aquatic organisms, including their potential to bioaccumulate and biomagnify. However, OMOE does not provide specific guidance on the incorporation of ERA principles into sediment assessment.

3.2.7 Quebec

The Environment Canada St. Lawrence Centre classifies sediments into three management categories based on SQGs used by OMOE and CCME (http://www.slv2000.qc.ca). The No Effect Level (NEL) corresponds to natural background. At the Minimal Effect Level (MEL) some effects are noticeable to some organisms, but not to the majority. At the Toxic Effect Level (TEL), 90% of the benthic organisms may be affected. Exceedence of SQGs results in implementation and examination of other LOE, often implemented in an ERA format. The Ministère de l’Environnement and the Ministère d’Environnement Canada (region Québec) provided extensive information (Lise Boudreau, pers. comm., July 19, 2004 and August 25, 2004; Caroll Bélanger, pers. comm., August 26, 2004). The OMOE SQGs are being replaced with values based on CCME ISQGs, which apply to both fresh and marine environments. Additional guidelines will be calculated based on the CCME TEL and PEL, with the same database, and will be used as management tools for dredged material disposal or as clean-up levels. Background contaminant levels will no longer be combined with SQGs, but will be part of the management tools. Three types of background levels will be used: a generic level; a level for post-glacier sediments; and, regional ambient levels. SQGs will continue to be used for initial screening, with sediment toxicity tests used to further discriminate sediments that may pose risks to the environment. The deadline for SQG revision and publication of interpretation and management options is fall 2004. The SQT (sediment chemistry, toxicity, biological evaluations) is a component of the revised ecotoxicological approach to sediment assessment in the region, which is expected to be finalized in 2007.

3.2.8 Atlantic Provinces

The Atlantic Provinces follow the generic CCME approach (SAIC Canada, 2002). Their approach involves two tiered steps and emphasizes a risk-based approach rather than sole reliance on generic SQGs. In the first step, measured levels of contaminants are compared to CCME ISQGs (or those of other jurisdictions where CCME ISQGs do not exist). If there are no exceedences, no further actions are required. However, if there are exceedences, then the site may require additional investigation which could include a site-specific ERA. The ERA involves various LOE including sediment chemistry (including bioavailability), toxicity, and benthic community status. These LOE are integrated in a WOE approach. There are no fixed rules on how to integrate the various lines of evidence, however Ken Doe (pers. comm., July 8, 2004) believes that "biological effects should be given more weight than chemical contamination". In Newfoundland, reliance is placed on consultants for choosing and integrating different LOE (Haseen Khan, pers. comm., July 14, 2004). Nova Scotia does not currently have provincially derived SQGs and defers to the CCME SQGs; in specific cases, the derivation of site specific guidelines based upon human health/ecological risk assessment is an option, however there is no technical reference nor prescribed approach for site specific ERA (Paul Currie, pers. comm., July 20, 2004). Recent examples of ERA conducted in the Atlantic provinces include: the Sydney Tar Ponds in Nova Scotia, PCB-contaminated sediment at the former Irving Whale site; and, a site in Dalhousie, New Brunswick, which involved the SQT (Kok-Leng Tay, pers. comm., July 16, 2004).

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Glossary

Acute Toxicity – Toxicity having a sudden onset, lasting a short time and severe enough to induce a response rapidly. The duration of an acute aquatic toxicity test is generally on the order of days and mortality is the response measured.

Adsorption – The adhesion of one substance on the surface of another.

Advection – The horizontal movement of a mass of water that causes changes in temperature or in other physical properties of the water.

Area Use – The extent to which an area is used (e.g., for feeding, rearing) by organisms such as fish.

Aroclor – A component of mixtures of polychlorinated biphenyls (PCBs), containing a large number of isomers and identified by a number reflecting the average degree of chlorination.

Assessment Endpoint – The undesired effect whose probability of occurrence is estimated in a risk assessment. The explicit expression of the environmental value that is to be protected. Examples include extinction of an endangered species, eutrophication of a lake, or loss of a fishery.

Benthic – Referring to organisms living in or on the sediments of aquatic habitats.

Benthos – The sum total of organisms (including plants and animals) living in, or on, the sediments of aquatic habitats.

Bioassay – The use of an organism or part of an organism as a method for measuring or assessing the presence or biological effects of one or more substances under defined conditions. A bioassay test is used to measure a degree of response (e.g., growth, or death) produced by exposure to a physical, chemical or biological variable (a toxicity test) or uptake of a chemical into an organism (a bioaccumulation test).

Bioavailability – Refers to the fraction of the total chemical in the surrounding environment which can be taken up by organisms. The environment may include water, sediment, suspended particles, and food items.

Biomagnification – Uptake of a contaminant through a food chain resulting in increasing concentrations through three or more trophic levels.

Bioturbation – The movement and relocation of bottom sediments by the activities of bottom-dwelling organisms.

Chronic Toxicity – A biological response of relatively slow progress and long continuance, usually associated with lower concentrations of chemicals than would cause an acute toxicity response.

Coincidental Sampling – Different field-collected samples from the same area/station are used for different analyses.

Conceptual Site Model – A three-dimensional representation of a site and its environment that represents what is known or suspected about contaminant sources as well as the physical, chemical and biological processes that affect contaminant transport to potential environmental receptors.

Diffusion – The random movement and scattering of water-soluble contaminants in the interstitial waters of sediments and into the overlying water column.

Distal – Situated away from the point of origin.

Ecological Risk Assessment – The process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. This definition recognizes that a risk does not exist unless: (1) the stressor has an inherent ability to cause adverse effects, and (2) it is coincident with or in contact with the ecological component long enough and at sufficient intensity to elicit the identified adverse effect(s).

Empirical – Derived from or depending on experience or observation/experimentation rather than theory or logic.

Human Health Risk Assessment – The process that evaluates the likelihood that adverse human health effects may occur or are occurring as a result of exposure to one or more stressors. This definition recognizes that a risk does not exist unless: (1) the stressor has an inherent ability to cause adverse effects, and (2) it is coincident with or in contact with the one or more humans long enough and at sufficient intensity to elicit the identified adverse effect(s).

Infauna – Invertebrate organisms living within the bottom sediment of fresh, estuarine or marine waters.

Interfacial – Having a common boundary; point of connection.

Invertebrate – Animal lacking a dorsal column of vertebrae or a notochord.

Line of Evidence – A component of Weight of Evidence determinations (e.g., toxicity, benthos alteration, biomagnification, chemical contamination).

Measurement Endpoint – An expression of an observed or measured response to a hazard; it is a measurable environmental characteristic that is related to the valued characteristic chosen as the assessment endpoint.

Receptor – The entity (e.g., organism, population, community, ecosystem) that might be adversely affected by contact with or exposure to a substance of concern.

Reference – A designated site, or set of conditions, used for comparison when evaluating sediment for contamination or pollution.

Remediation – An activity undertaken to correct an unacceptable existing condition (e.g., treating or moving polluted sediment).

Sediment – Material, such as sand or mud, suspended in or settling to the bottom of a liquid. Sediment input to a body of water comes from natural sources, such as erosion of soils and weathering of rock, or as the result of anthropogenic activities, such as forest or agricultural practices, or construction activities.

Sediment Quality Guideline – A numerical value for one or more chemicals related to a level of probability (but not of certainty) that adverse environmental effects may or may not occur above or below the guideline value.

Sensitivity Analysis – Analysis undertaken to determine what data or information are primarily responsible for an assessment.

Species Sensitivity Distribution – A graphical representation of the different sensitivities of different species to the same stressor. Used to determine the concentration or level of a stressor protective of most species in the environment.

Stochastic Uncertainty – The inherent randomness of a system being assessed; can be described and estimated but cannot be reduced.

Surficial – On the surface.

Synoptic Sampling – Subsamples for analyses are taken from the same, generally composite, sample.

Toxicity Identification Evaluation – A methodology for determining the causative agent(s) for toxicity identified in toxicity tests. Specific contaminants are removed and the sample retested until toxicity has been removed, then the presumed causative agent(s) are added back in and the sample retested to confirm that they are indeed the causative agent(s).

Trophic Level – Functional classification of organisms in a community according to feeding relationships – e.g., the first trophic level includes green plants, the second level includes herbivores (plant eaters), etc.

Weight of Evidence – A determination related to possible ecological impacts based on multiple Lines of Evidence.

List of Acronyms

AOCs	Areas of Concern
BMF	Biomagnification Factor
CBR	Contaminant Body Residue
COA	Canada-Ontario Agreement (Respecting the Great Lakes Ecosystem)
COC	Contaminant of Concern
COPC	Contaminant of Potential Concern
CSM	Conceptual Site Model
EC	Effective Concentration
ERA	Ecological Risk Assessment
HHRA	Human Health Risk Assessment
IJC	International Joint Commission
LEL	Lowest Effect Level
LOE	Line of Evidence
OMOE	Ontario Ministry of the Environment
PCA	Principle Component Analysis
ROPC	Receptor of Potential Concern
SAP	Sampling and Analysis Plan
SSD	Species Sensitivity Distribution
SQG	Sediment Quality Guideline
TEL	Threshold Effect Level
TIE	Toxicity Identification Evaluation
TOC	Total Organic Carbon
TRC	Tissue Residue Criterion/Criteria
WOE	Weight of Evidence

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