REPORT ON:

DETAILED ECOLOGICAL RISK ASSESSMENT (DERA) IN BRITISH COLUMBIA TECHNICAL GUIDANCE

Submitted to:
The Ministry of Environment
September 2008

Submitted by:
Science Advisory Board
For Contaminated Sites in British Columbia

DRAFT

Preface and Acknowledgements

The report herein on *Detailed Ecological Risk Assessment t (DERA) in British Columbia, Technical Guidance* including three Appendices is presented for the information and benefit of the Contaminated Sites community in British Columbia. It is hoped that it will be of interest to practitioners in other jurisdictions.

The work builds on the earlier review of detailed ecological risk assessment which is posted on the SABCS documents page. The Task Group for both the review and the technical guidance that follows has been chaired by Beth Power of the SABCS and Azimuth Consulting Group Inc, and her skill and excellent leadership in bringing the project to completion is great fully acknowledged by the SABCS. The contributions of the members of the task group to the development and review of the work are also much appreciated.

The development of the technical guidance was undertaken by Golder Associates for the SABCS and the high level of expertise and thoroughness that was demonstrated in the work is acknowledged with appreciation by the SABCS.

The SABCS acknowledges with appreciation grant funding from the government of British Columbia through the Ministry Of Environment that has made this work possible.

The Science Advisory Board for Contaminated Sites in British Columbia is soliciting comment on the documents which together constitute a report to the BC Ministry of Environment on Detailed Ecological Risk Assessment. Comments will be reviewed and compiled by the SABCS, and will be much appreciated.

Practitioners and others with interests in contaminated sites should be aware that this report has not been adopted in whole or in part by the Ministry of Environment of British Columbia. While every effort has been made to incorporate the best available science, it should be used solely as scientific review and commentary by the reader and applied in practice solely at the readers discretion and responsibility. This disclaimer is consistent with SABCS Policy.

Please send your comments to the Science Advisory Board for contaminated Sites by email or email attachment to pwest@uvic.ca. Comments received by December 31, 2008 will be most useful in further refinement of this work. However comments at any time on SABCS work are always appreciated.

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DETAILED ECOLOGICAL RISK ASSESSMENT (DERA) IN BRITISH COLUMBIA – TECHNICAL GUIDANCE

Submitted to:

Science Advisory Board (SAB) for
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NOTE TO READER

Role of Provincial Policy – Methods and acceptance criteria to assess ecological risk are tied strongly to provincial policy decisions. This DERA guidance focuses on the technical and scientific methods of ecological risk assessment. Technical decision points that may be affected by provincial policy are identified throughout the document. To accommodate future updates or revisions to Ministry of Environment (MOE) policy, this document does not provide detailed discussions of individual policy determinations by MOE. Rather, the practitioner **must** consult the most recent MOE policy (*Ecological Risk Assessment Policy Decision Summary*) located on the Ministry of Environment Land Remediation website. It is essential that DERA guidance be applied in conjunction with current policy decisions.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

APPENDIX I – DIRECT MEASUREMENT TOOLS

APPENDIX II – MODELLING TOOLS

APPENDIX III – INTERPRETIVE TOOLS

LIST OF ACRONYMS

AEC	area of environmental concern
AOU	American Ornithologist's Union
APEC	area of potential environmental concern
ASTM	American Society for Testing and Materials
AVS	acid volatile sulphides
B-IBI	Benthic Index of Biotic Integrity
BAF	bioaccumulation factor
BC	British Columbia
BCF	bioaccumulation factor
BCMELP	British Columbia Ministry of Environment, Lands and Parks (now MOE)
BCWLAP	British Columbia Ministry of Water, Land, and Air Protection (now MOE)
BIEAP	Burrard Inlet Environmental Action Program
BLM	biotic ligand model
BMF	biomagnification factor
BSAF	biota-sediment accumulation factor
CA	correspondence analysis
CCA	canonical correspondence analysis
CCME	Canadian Council of Ministers of the Environment
CDC	Conservation Data Centre
CEPA	*Canadian Environmental Protection Act*
COC	contaminant of concern
COPC	contaminant of potential concern
COSEWIC	Committee on the Status of Endangered Wildlife in Canada
CREM	Council on Regulatory Environmental Modelling (USEPA)
CSAP	Contaminated Sites Approved Professionals
CSR	Contaminated Sites Regulation
CSST	Contaminated Sites Standards Taskgroup
CWS	Canadian Wildlife Service
CYP1A	cytochrome P4501A protein expression
DCA	detrended correspondence analysis
DERA	detailed ecological risk assessment
DFO	Fisheries and Oceans Canada
DQO	data quality objective
DSI	detailed site investigation
EC_X	effect concentration, with percent effect of X
EC_{20}	effect concentration, with percent effect of 20% for non-lethal endpoint
EC_{50}	effect concentration, with percent effect of 50% for non-lethal

	endpoint
ECOTOX	USEPA ecotoxicology database (Release 4.0)
EEC	expected environmental concentration
EEM	environmental effects monitoring
EMA	*Environmental Management Act*
EPT	mayflies (Ephemeroptera), stoneflies (Plecoptera), caddisflies (Tricoptera)
ERA	ecological risk assessment
ERAGT	Ecological Risk Assessment Guidance Team
EROD	ethoxyresorufin-O-deethylation
FA	factor analysis
FAC	fluorescent aromatic compounds
FREMP	Fraser River Estuary Management Program
FSP	field sampling plan
GIS	global positioning system
HC_p	hazardous concentration, with p percent of species affected
HQ	hazard quotient
HWR	Hazardous Waste Regulation
ILMB	Integrated Land Management Bureau
LC_{50}	lethal effect concentration, with 50% mortality
LC_p	lethal effect concentration, with p% mortality
LDA	linear discriminant analysis
LOE	line of evidence
LOEC/LOAEL	lowest observed effect concentration/lowest observed adverse effect level
MANOVA	multivariate analysis of variance
MDS	multidimensional scaling
MOE	British Columbia Ministry of Environment
MSD	minimum significant difference
MSIM	multiple species inventory and monitoring
NMDS	non-metric multidimensional scaling
NOEC/NOAEL	no observed effect concentration/no observed adverse effect level
NRC	National Research Council
OMEE	Ontario Ministry of Environment and Energy
ORNL	Oak Ridge National Laboratory
PAH	polycyclic aromatic hydrocarbon
PBET	physiologically-based extraction test
PBPK	physiologically-based pharmacokinetic model
PCA	principal components analysis
PCB	polychlorinated biphenyl
PF	problem formulation

PF/SAP	problem formulation/sampling and analysis plan
PSI	preliminary site investigation
QAPP	quality assurance project plan
QA/QC	quality assurance/quality control
QSAR	quantitative structure-activity relationship
RDA	redundancy analysis
ROC	receptor of concern
ROPC	receptor of potential concern
SAB	Science Advisory Board
SALES	sequential analysis of lines of evidence
SAP	Sampling and Analysis Plan
SARA	*Species at Risk Act*
SCS	sensitive contaminated site (CSR)
SedQCscs	sediment quality criterion (standard) for sensitive contaminated sites
SedQCtcs	sediment quality criterion (standard) for typical contaminated sites
SEM	simultaneously extracted metals
SLRA	screening level [ecological] risk assessment
SSCW	Salem Sound Coastwatch
SSD	species sensitivity distribution
SSI	supplemental site investigation
TCS	typical contaminated site (CSR)
TIE	toxicity identification evaluation
TG	technical guidance (CSR)
TL	trophic level
TOC	total organic carbon
TRV	toxicity reference value
TTF	trophic transfer factor
TWINSPAN	two-way indicator species analysis
UCL	upper confidence limit
UF	uncertainty factor
USDOE	United States Department of Energy
USEPA	United States Environmental Protection Agency
UV	ultraviolet radiation
WDNR	Washington Department of Natural Resources
WHAM	Windermere Humic Aqueous Model
WOE	weight of evidence

1.0 INTRODUCTION

Detailed ecological risk assessment (DERA) is a process used to evaluate the probability and magnitude of harm to non-human organisms that is attributable to anthropogenic stressors. In British Columbia, DERA can be applied for a range of site conditions (type, size, and complexity), regulatory drivers (ranging from voluntary assessments to formal remedial orders), and environmental conditions and pathways. The purpose of this document is to provide technical guidance for practitioners of DERAs in British Columbia, in support of the provincial process for assessment and management of contaminated sites.

1.1 Framework for Ecological Risk Assessment

The classical definition of ecological risk assessment (ERA) is the determination of the probability of an effect occurring to an ecological system. USEPA (1992) defines ERA as "a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors." Therefore, the two critical components of a risk assessment are probability and magnitude of harm resulting from exposure to a stressor.

The basic approach to ecological risk assessment (ERA) in Canada was developed by Environment Canada (1993) and elaborated upon in numerous books (Suter, 2006; Landis and Yu, 1995). Additional publications cover the use of ERA for the remediation of contaminated sites (OMEE, 1996; CCME, 1996; Environment Canada 1994). In April 1998, the U.S. Environmental Protection Agency published *Guidelines for Ecological Risk Assessment* (USEPA, 1998), which supersede the USEPA (1992) guidance and provide a systematic procedure for risk assessment that is mirrored in numerous other jurisdictions.

A common element of risk assessment guidance documents is that they seek to evaluate the interaction between exposure, receptor and hazard for each stressor. A stressor, for the purposes of this document, is a condition or environmental factor that causes one or more responses, either positive or negative, to a biological system. Risk is a combination of these three attributes expressed as a probability:

1. *Exposure* – Exposure is the interaction of an organism with a stressor. Exposure often entails measurement of concentrations in environmental media (*e.g.*, water, soil, air, tissue). Exposure can also be expressed in terms of an intake or ingestion rate (amount of contaminant encountered or ingested by the receptor per unit of time per unit body weight), which often provides a more accurate representation of the potential for toxic effects.

2. *Hazard* – Hazard is the potential of a stressor to cause particular deleterious effects to an organism or ecosystem. Hazard may be determined by considering literature reviews of concentration-response, environmental quality guidelines, or direct assessment of site-specific data. Hazard determination requires specification of an effects endpoint and organism of interest, an effect size of interest, and a means of quantifying the endpoint.

3. *Receptor* – Receptors are the organisms of concern or ecosystem component(s) that are being investigated. Receptors may be individual organisms (*e.g.*, threatened and endangered animals), local populations of a single species, or communities consisting of multiple individuals of multiple species.

The interaction between exposure, receptor and hazard in the risk assessment framework leads to three principles.

- *Receptor principle* – Without presence of organisms considered to be important (either for their inherent value or their linkage to other ecological entities) there is no ecological risk.

- *Pathway principle* – Without a mechanism for site stressors to interact with a biological entity, there is no ecological risk. Risk requires the existence of a pathway from the source of the stressor to a site of toxic action in an organism.

- *Hazard principle* – A stressor poses no ecological risk unless exposure occurs at a level sufficient to elicit toxic effects under a worst-case condition. The 16th century alchemist Paracelsus noted that "all substances are poisons; there is none which is not a poison." He further commented that: "The right dose (or exposure) differentiates a poison and a remedy". Thus, if the dose or exposure to a stressor is sufficiently low, even a highly toxic substance will cease to cause a harmful effect, and on this basis contaminants can be screened based on their hazard potential.

1.2 Why Apply Risk Assessment?

The practice of ecological risk assessment, in its broadest sense, includes a wide range of tools and frameworks designed to inform decisions related to the management of ecological resources. The most common application of the ERA process is to assist with site management and related decision-making. Often this decision-making is linked to a specific regulatory framework (*e.g.*, in BC, obtaining a Certificate of Compliance for a contaminated site), whereas in other cases the decision-making relates to broader site management objectives.

The Regulatory Driver

British Columbia's Contaminated Sites Regulation (CSR; MOE, 2007a) provides technical support for the *Environmental Management Act* (*EMA*). The regulation includes a process for determining whether a site is contaminated, whether a site requires cleanup, and a process to establish remediation requirements. Site management relies on: (a) numerical standards that include generic and matrix (scheduled standards) and site-specific standards, or (b) risk-based standards. Where the latter option is chosen, remediating a site to meet risk-based standards requires a quantitative ERA. The general framework for linking ERA to the site assessment and remediation process specified in the CSR is illustrated in Figure 1, and is discussed in detail in Section 2 of this document. Section 2 also describes other regulatory drivers, such as federal statutory triggers for ecological risk assessment.

The Decision-Making Driver

Not all environmental issues that incorporate risk assessment are driven by a specific regulatory mechanism. Increasingly, site managers and responsible parties use risk assessment to facilitate decision making and optimize use of resources. Risk assessment can be used proactively to evaluate the benefits and limitations of potential management actions. In this context, risk assessment is often linked to the field of decision analysis. Decision analysis is a management technique in which statistical tools such as decision trees, multivariate analyses, and/or probabilistic methods are applied to the practical issues of managing contaminants and other stressors. The objective of a decision analysis is to identify the most advantageous alternative under the site-specific circumstances.

Examples of risk assessment as a decision-making tool include:

- Assessment of the environmental constraints associated with a sediment maintenance dredging project;
- Determination of the need for environmental controls associated with an effluent discharge;
- Determination of relative risks associated with various upland remediation alternatives for a contaminated site (*e.g.*, pump-and-treat, physical containment, hydraulic containment, *in situ* treatment); and,
- Determination of the environmental effects associated with a mining project during project planning, operation and/or closure.

Although all of the above examples would include various aspects of regulatory permitting and compliance, they are driven by a need to incorporate information on environmental risks in a broader decision-making framework. To this end, many of the procedures and principles described in this document can be harmonized with the requirements of legislation such as the federal Metal Mining Effluent Regulations or the provincial Municipal Sewage Regulation.

The focus of this document is on the provincial regulatory application of risk assessment, particularly in relation to the CSR (see Section 2). However, many of the processes and tools described in this report are relevant to the broader decision-making application of ERA.

As risk assessment is a decision-making framework, it should proceed to the point that an informed risk management decision can be made. The underlying role of DERA is to support involved parties (site owners and regulators) to make informed site management decisions. At many sites, risk assessment and remediation are applied concurrently. Site managers may opt to remediate the site (or parts of it) at any point in the risk assessment process.

1.3 Purpose of DERA Guidance

Detailed ecological risk assessment (DERA) for a contaminated site provides a framework for the assessment of risks to non-human organisms associated with environmental stressors. Whereas Sections 1.1 and 1.2 of this document describe the generic role of ERA in site management, DERA encompasses a subcomponent of the broad ERA process in British Columbia, as illustrated in Figure 2. A DERA may be conducted as a follow-up to a screening level ecological risk assessment (SLRA[1]), or may be conducted independently from SLRA. The connection between DERA and SLRA is represented in Figures 2 and 3. Where initial assessment or SLRA indicates that any pathway is operable, the ERA process progresses to a DERA problem formulation. It is possible to apply SLRA to screen out individual pathways; however, where pathways are operative, DERA must be used. DERA in British Columbia applies to a wide range of sites and levels of complexity.

[1] Under the CSR, an SLRA consists of a pathway-based assessment using simple, highly constrained administrative rules and/or simple constrained models to identify situations where risk is clearly acceptable despite exceedances of numerical standards. The SLRA is available as Protocol 13 on the MOE Land Remediation website: http://www.env.gov.bc.ca/epd/remediation/policy_procedure_protocol/index.htm

Whereas ecological risk assessment processes are based on science, apply adaptive approaches to decision-making, and have well-defined analytical stages (Power and McCarty, 2002), the term "detailed ecological risk assessment" requires clarification to avoid confusion with other types of quantitative risk assessments that are not the focus of this document. Some clarifications include:

- *Engineering versus Toxicology Risk Paradigm* – The conventional engineering definition of risk is the "expected loss from failure" (Todinov, 2006). This approach to risk assessment, which often includes quantitative tools of statistical decision analysis and probabilistic risk assessment, is designed to evaluate the consequences (often financial) of alternative management decisions. In contrast, the toxicology-based paradigm expresses risk as the probability and magnitude of adverse effects. Although the engineering-based risk assessment methodology is frequently applicable to ecological systems (fish stock assessment, biological hazards, wildlife population modelling), it is not the focus of this guidance document.

- *Stressor Types* – Stressors may be chemical, biological, and/or physical in nature, but DERAs in British Columbia are most frequently applied to the evaluation of chemical stressors on ecosystems. Although many of the principles discussed in this document apply to all stressor types, this document emphasizes contaminant-induced responses and issues related to contaminated sites. Other types of stressors (*e.g.*, habitat modification) can often be handled using a risk assessment framework, but may also trigger an environmental impact assessment process under the provincial *Environmental Management Act* or federal laws.

- *Definition of "Detailed"* – In this document, DERA applies to a broad range of levels of complexity. The term "detailed" applies to all levels of complexity and analysis (usually beyond the pathway screening conducted in an SLRA). Whereas other jurisdictions apply nomenclature to distinguish iterations within a risk assessment (*e.g.*, Level 1, Tier 2, or other), the term DERA conveys no specific stage of a phased risk assessment and in fact may include multiple iterations or tiers.

DERA guidance applies to sites that cannot be resolved through SLRA and that require the application of more sophisticated assessment methods to evaluate environmental risk. DERA guidance is also intended to provide the greatest degree of uncertainty reduction per unit cost, such that DERAs are practical and feasible, while also satisfying regulatory requirements. DERAs are iterative, often starting with "conservative" assumptions which, if risks are predicted, are then replaced with best estimates of exposure, effects and reported with associated uncertainties (Dearfield *et al.*, 2005).

This DERA framework facilitates a transparent and consistent approach in terms of both assessing and prioritizing risks. Although each site is unique, and the tools and approaches applied (including the use of professional judgement) will vary among sites, it is desirable to maximize consistency among DERAs. The role of this DERA guidance is to streamline the following aspects:

- **Completeness:** DERA guidance increases our confidence that key risk components are not ignored (*e.g.*, relevant pathways, stressors, receptors are all considered);

- **Relevance:** DERA guidance increases our confidence that the tools chosen are biologically relevant to the environmental values of interest;

- **Compliance:** DERA guidance increases the probability that the risk assessment deliverable will satisfy pertinent regulatory requirements, remain consistent with MOE policy and guidance, and receive positive review by interested parties; and,

- **Consistency:** DERA guidance, although not prescriptive, will encourage practitioners to address a common set of risk issues, and to provide rationales for the approaches selected. Identification of key decision points and provision of guidance on commonly encountered risk assessment challenges will also facilitate third party reviews of DERAs.

Site managers contemplating DERAs face differences in management settings (ranging from voluntary site assessments to remediation orders), data availability, time constraints, and other limiting factors. The challenge of DERA is to facilitate consistent practice, while allowing the process to be customized to site-specific factors. In this respect, DERA is best viewed as process (Figure 3) rather than a prescriptive "cookbook". DERA guidance is targeted to identification of the appropriate risk-based questions at each site; the specific tools and approaches will vary depending on the scale and management objectives of the site.

1.4 Scope of DERA Guidance

This guidance document was prepared using the following guiding principles:

Intended Audience – This document was prepared for an audience of experienced risk assessment practitioners. Readers are assumed to be well-versed in risk assessment terminology and concepts, and to have familiarity and experience with the practical application of ERA in British Columbia (including existing contaminated sites legislation such as MOE [2007a]).

Guidance Manual versus Code of Practice – This document is not intended to be a formal code of practice for conducting detailed ecological risk assessments in BC. It is a technical guidance manual, and as such, does not contain guidance on several aspects of administrative and/or policy-related process (*e.g.*, legal and formal regulatory reporting requirements). A separate protocol document will be prepared by MOE that will establish mandatory procedures for DERA.

Role of Provincial Policy – Methods and acceptance criteria to assess ecological risk are tied strongly to provincial policy decisions. DERA guidance focuses on the technical and scientific methods of ecological risk assessment. Technical decision points that may be affected by provincial policy decisions are identified throughout the document. To accommodate future updates or revisions to MOE policy, this document does not provide a detailed discussion of each policy statement. Rather, the practitioner must consult the most recent MOE policy statements (Ecological Risk Assessment Policy Decision Summary) located on the MOE Land Remediation website[2]. It is essential that DERA guidance be applied in conjunction with current policy decisions.

Level of Prescription and Detail – The emphasis of this document is on higher level guidance to promote consistency among practitioners in terms of resolving key issues. The guidance is intended to be flexible (to accommodate the range of possible site conditions and evaluation methods) and aims to help practitioners consistently follow general procedures and thought processes. DERAs are sufficiently complex that they cannot be reduced to a standard "cookbook" that can be universally applied at all sites. However, there are many areas in which the risk assessment process can be harmonized among sites by focusing on similar risk questions. Furthermore, in the future, MOE may commission development of more prescriptive guidance for commonly applied risk tools; therefore, this document has a modular format to facilitate future additions. Certain aspects of specific risk assessment techniques were explored in greater detail herein, given the immediate need to supplement the existing provincial risk assessment guidance in high-priority areas.

The supporting technical appendices (Appendix I through III) contain supporting biological and toxicological details; however, they are not intended to replace a good library or other data repositories. Practitioners are encouraged to review the literature cited in the appendices and to support that information with emerging literature related to key subject areas.

Site Applicability – The DERA document is intended to apply to variety of sites in terms of complexity and size. However, the document is not intended to provide guidance for wide area sites (see policy decision comment below). These types of sites require specialized expertise and more consultation with MOE.

[2] http://www.env.gov.bc.ca/epd/remediation/index.htm

Policy Decision: Wide Area Sites – Wide area sites are sites that a MOE director may designate (after careful consultation with affected parties) if they cover an extensive geographic area and have multiple individual properties containing specified contaminants from specified sources. Amendments to the CSR address specific remediation standards/procedures for wide area sites. Practitioners must consult with MOE in evaluating these sites, and certain requirements and processes may differ from those sites relative to those discussed in this manual.

Unique Conditions – The DERA document emphasizes situations and conditions commonly encountered in British Columbia. Specific sites may include issues for which generic guidance will not be sufficient, and for which tools other than those described herein are required. This guidance is generally applicable, but not necessarily sufficient in all cases, and should be used only with a reasonable understanding of the limitations of the tools.

1.5 Document Organization

Section 2 provides information on the regulatory regime in relation to the process of DERA. The remainder of this guidance manual is organized in a hierarchical manner. The broadest hierarchy is based on the traditional framework for ERA (*i.e.*, problem formulation, exposure assessment, effects assessment, and risk characterization). The problem formulation (Section 3.0) is used to understand the site and frame the relevant risk issues. Following completion of the draft problem formulation (which can later be revised based on new information), the risk assessor should refer to the relevant ecosystem-specific sections of the exposure and effects assessments (Sections 4.0 and 5.0) and the applicable appendices. Risk characterization for all sites should reflect the guidance presented in Section 6.0.

The secondary hierarchy is organized around five broad ecosystem types (*i.e.*, deep aquatic, shoreline, rivers and streams, upland human-use, upland wildlands) (Figure 4). Exposure pathways, measurement endpoints, and risk assessment tools tend to differ substantially among these generic ecosystems. For some sites, multiple ecosystems may be present; however, it is rare that all ecosystems will be applicable at a single site.

Throughout the document, two types of text boxes are used:

1. Key issues and content have been summarized as shown below (single line around the text box).

2. Technical decision points affected by provincial policy are shown below (double line around the text box).

Key Issues for the DERA Practitioner: This box type highlights major information needs or "state-of-the-science" issues that may be encountered during the risk assessment process. The guidance manual does not provide prescriptive guidance on these topics. The practitioner should determine if the issue is appropriate on a site-specific basis and proceed accordingly.

Content for the DERA:

- Highlights specific items for inclusion or consideration in the DERA document.

Policy Decision: This box type identifies a technical decision point that may be affected by provincial policy. It is essential that the practitioner consults with MOE to obtain up-to-date summaries of policy decisions.

2.0 DERA ADMINISTRATIVE PROCESS

Because of the number of potential scenarios contemplated within the DERA process, application of this guidance document requires an understanding of the overall regime (regulatory and procedural) overarching the technical and scientific components of DERA. This section provides a broad introduction to these process-based elements, such that risk assessment tools and procedures described in the remaining chapters are integrated into the overall site management strategy.

The specific objectives of this section include:

- Discussion of how DERA fits into the overall site assessment and management regime in BC (*i.e.*, regulations, guidance, and contaminated site assessment procedures).

- Presentation and explanation of an overall guidance process for DERA, based conceptually on existing risk assessment guidance from other North American jurisdictions, but also customized to suit the BC regulatory regime.

- Identification of checkpoints for review and feedback in DERA (*i.e.*, when is stakeholder consultation, formal review, and/or opportunity for program adaptation recommended?).

- Identification of the respective roles of risk assessment practitioners, regulators (*e.g.*, Ministry of Environment [MOE] staff), and Contaminated Sites Approved Professionals (CSAP), and their linkages to the DERA process.

2.1 Provincial Regulations

In Canada, the division of powers under the *Constitution Act* places the authority and jurisdiction for the management of land and wastes under the authority of the provinces (S. 92[13] and S. 92[16] *Constitution Act* [1867]). The British Columbia *Environmental Management Act* (*EMA*) is the primary provincial legislation under which contaminated sites in BC are managed.

It is the responsibility of a DERA practitioner to have full awareness of the applicable provincial laws, regulations, and codes of practice that govern the management of contaminants and biological resources. A summary of the most commonly referenced provincial legislation pertinent to DERA is provided below. It is strongly recommended

that practitioners review the applicable legislation for a given site prior to beginning a DERA. Amendments and revisions to provincial legislation are common, requiring frequent updates over time[3].

Environmental Management Act – This statute, brought into force in 2004, covers provincial aspects of permitting (authorization of the introduction of waste into the environment from prescribed uses under the Waste Discharge Regulation) and specifies limitations to other introductions of waste into the environment. A central theme of *EMA* is a prohibition against causing pollution[4]. Pollution Abatement Orders can require both remediation to numerical standards and the more general requirement to not cause pollution.

- Contaminated Sites Regulation (including amendments) - The Contaminated Sites Regulation (CSR), a statute promulgated under *EMA*, provides specific direction on how pollution is addressed at contaminated sites for the purposes of the *Act* (*EMA Section 6[4]*). The CSR defines a contaminated site as a site at which the numerical standards of the regulation have been exceeded. In addressing or remediating contamination at a contaminated site, a responsible person may elect to use either the numerical concentration standards or the risk based standards (*i.e.,* risk assessment approach) of the regulation. When the risk assessment option is selected, the risk assessor may also need to consider if additional substances than those prescribed in the regulation schedules could potentially result in risk or "cause pollution" as defined (and prohibited) under the *Act*.

- Hazardous Waste Regulation (HWR, including amendments) – Under *EMA*, this regulation addresses the identification, proper handling, treatment and disposal of hazardous wastes. It includes procedures for determining whether materials containing levels of specific contaminants (*e.g.*, polycyclic aromatic hydrocarbons, dioxin-like substances) warrant classification as hazardous wastes and therefore would be subject to the requirements of the HWR.

- Municipal Sewage Regulation – Under *EMA*, this regulation establishes requirements for local governments and private sewage dischargers for the treatment, reuse, and discharge of domestic sewage, wastewater or municipal liquid waste. As such it is relevant to the DERA evaluation of wastewater treatment plant discharges and associated receiving environments.

[3] Updates and summaries of applicable provincial environmental legislation are provided on the Internet site: http://www.env.gov.bc.ca/. Updates specific to *EMA* are currently located at the following address: http://www.env.gov.bc.ca/epdiv/env_mgt_act/ema_reg_amend.html

[4] Under *EMA* Section 1[1], pollution is defined as "the presence in the environment of substances or contaminants that substantially alter or impair the usefulness of the environment."

Water Act – The Ministry of Environment manages the *Water Act*, which aims to protect water resources in British Columbia. It includes provisions for protection of wells and ground water.

- Groundwater Protection Regulation – The primary purpose of the Ground Water Protection Regulation is protection of the quantity and quality of the province's ground water resource (integrity, efficient use, and environmental safety). In the DERA context, this regulation relates mainly to the site investigation process.

Wildlife Act – Under this *Act*, the province is authorized to designate threatened and endangered species and designate land in a wildlife management area as a critical wildlife area or wildlife sanctuary. This *Act* is relevant to the selection of receptors of concern in a risk assessment, and also to the level of protection afforded in the risk assessment[5]. Also, the *Act* establishes an offence if a person harms wildlife or their habitat by altering, destroying or damaging wildlife habitat, or by depositing on land or water a non-permitted substance or product. The latter provisions are relevant to the selection of assessment endpoints in an ecological risk assessment, and interrelate with Section 78 of *EMA*.

2.2 Linkage to Site Assessment Process

DERAs for contaminated sites in British Columbia are part of a larger contaminated sites regulatory regime that includes site characterization, remediation, and/or risk management (Figure 1). Part 4 of *EMA* (Contaminated Site Remediation) outlines the process for environmental site assessment in BC, including the development of site profiles, site investigations, site registries, and provision of contaminated site regulations (CSR and HWR).

2.2.1 Instruments

Often, the objective of a site assessment and remediation project is the issuance of a regulatory instrument, and DERAs are often central to the achievement of this goal. Examples of regulatory instruments include:

- *Determination* – a determination that a site is or is not contaminated, under Section 44 of *EMA*.

- *Approval in Principle* – an approval under Section 53(1) of *EMA* indicating that a remediation plan for a contaminated site has been reviewed, approved, and may be implemented in accordance with conditions specified by the director.

[5] For example, risk to a threatened or endangered species is generally considered at the level of individual organism, rather than the population level, to enhance the protection for the species as a whole.

- *Certificate of Compliance* – a statement under Section 53(3) of *EMA* that the contaminated site has been remediated in accordance with the human health and environmental protection standards specified by (a) numerical or risk-based standards; (b) any applicable orders; (c) any remediation plan approved by the director, and (d) any additional requirements imposed by the director.

- *Contaminated Soil Relocation Agreement* – a statement under Section 55 of *EMA* that the relocation of contaminated soil will not cause a significant potential for adverse effects on human health or pollute the environment.

2.2.2 Deciding When to Conduct a DERA

DERAs are initiated when a risk-based standards approach is chosen (as opposed to the numerical standards approach[6]) and as a result of one of the following scenarios (Figure 3):

- An SLRA was conducted for the site; however, one or more exposure pathways indicated potential risk that could not be eliminated from consideration (because there are potential risks and/or because of uncertainties or incomplete information). In these cases, DERA focuses on remaining exposure pathways for which potential risks remain and explores issues that require reconsideration and/or more detailed analysis.

- The site is ineligible for SLRA due to precluding conditions[7] (*e.g.,* contaminated sediment is present).

- Site investigations (or project scoping) suggest that a SLRA is unlikely to bring closure to the risk issues at the site. In this case, a DERA may be initiated directly.

Irrespective of the scenario that results in choosing a risk-based standards approach, DERAs must identify all relevant risk issues during the problem formulation phase, and systematically eliminate exposure pathways with negligible risk in order to focus on those remaining issues that require detailed analysis. The size, complexity, and cost of the DERA are determined by the number of issues that could not be eliminated from consideration during the screening phase, as well as the desired level of certainty in the risk estimates relative to site management goals.

[6] A risk-based approach may also be applied concurrently with a numerical standards approach (*i.e.*, to address pathways, stressors, or portions of the site that cannot be resolved through the numerical approach alone).
[7] Consult Protocol 13 for list of precluding conditions.

2.2.3 CSR Process for Addressing Contaminated Sites

The CSR process is the default framework for sites under provincial jurisdiction; it is noted that federal sites may follow federal guidance. The CSR process consists of five main stages (modified from MacDonald and Ingersoll, 2003a, 2003b) that are common to upland and aquatic environments:

1. *Site Identification and Screening* – site profiles are submitted and assessed by regulators pursuant to Section 40 of *EMA*[8], and a determination is made regarding need for follow-up activity. Suspected contamination based on a site profile can trigger a site investigation (Step 2, below).

2. *Site Investigation and Determination* – If information from the site profile or another source indicates that a site is potentially contaminated, preliminary and/or detailed site investigations (*i.e.*, PSIs and DSIs) are often conducted to determine if the site is contaminated, as defined under the CSR. A PSI assesses the present and historical site use and management practices, and includes a review of records, a site visit, and limited sampling of the relevant environmental media. A DSI characterizes a site with a reasonable degree of certainty, identifying distribution, depth and degree of contamination and extent of contaminant migration.

3. *Site Management Planning* – Remediation plans are developed at this stage (sometimes preceded by a review of remediation options), and the approvals process is initiated. This stage evaluates who is potentially responsible for the contamination and who is potentially liable for clean-up costs. If necessary the province can impose remediation requirements through a Remediation Order or Pollution Abatement Order.

4. *Risk Management and Remediation* – The remediation stage covers all of the activities that are associated with cleaning-up or securing a contaminated site. The overall goal of contaminated site remediation is to restore the environmental quality of the site to a level that does not pose unacceptable risks to humans or ecological resources. The regulations provide the responsible party with a choice of two approaches related to the determination of acceptable cleanup levels (*i.e.*, remediation standards). Specifically, the responsible party may: (1) use numerical standards provided in the regulation or derive site-specific numerical standards, or (2) use risk assessment to determine if a risk management scenario will meet risk-based remediation standards (Figure 1).

[8] Guidance for completion, submission, and review of site profiles is summarized in Administrative Guidance on the Ministry website: http://www.env.gov.bc.ca/epd/remediation/guidance/

5. *Monitoring and Evaluation* – Following the implementation of remedial measures, confirmatory sampling and analysis is typically required as a condition for issuance of a Certificate of Compliance. If the numerical standards have been used, the residual soil and water must be checked to ensure compliance with numerical standards. In cases where the risk-based standards have been used, long-term custom monitoring programs are often adopted, to ensure the continuing effectiveness of risk management works and no-action alternatives.

Risk assessment activities may be conducted at several of the above stages, depending on the context of the problem. For example, risk assessments are sometimes conducted voluntarily by a responsible party prior to formal determination as a contaminated site under CSR (*e.g.,* for due diligence). Alternatively, risk assessments are sometimes conducted as a confirmatory stage following active remediation or other site management. However, risk assessments in BC are most commonly conducted in conjunction with Step 2 (Site Investigation and Determination). Therefore, the discussion below emphasizes the site investigation process and corresponding linkage to DERA.

The site investigation process is well defined in BC under *EMA* and CSR. Generally, the process is staged and includes the Stage I Preliminary Site Investigation (PSI), Stage II Preliminary Site Investigation and a Detailed Site Investigation (DSI). The Stage I PSI includes a historical review of the Site to identify all areas of potential environmental concern (APECs) and contaminants of potential concern (COPCs) (CSR 58[1][a]). Stage II PSI targets sampling at each APEC identified in Stage I to confirm or refute suspected contamination (CSR 58[1][b]). If contamination is found, the APECs become areas of concern (AECs) and are delineated vertically and horizontally in a DSI. Although the process is defined by three stages, they are not always conducted as distinct stages. Depending on overall project objectives, two or more stages may be combined in one report (*e.g.* Stage I and II PSI), or contamination may be delineated during the remediation process, particularly if the responsible party is pursuing independent remediation.

According to 59(2) of the CSR, a DSI must provide information necessary for conducting a risk assessment, if applicable, for the Site. Therefore, in order for the necessary information to be collected in a DSI, the objectives of the risk assessment and data requirements to address the objectives must be defined and linked to the site assessment prior to the initiation of field work for the DSI.

In reality, risk assessment is an iterative process, particularly for more complex sites. This is reflected in Figure 3, which shows feedback loops (represented as two-direction arrows) during both the Problem Formulation and subsequent analysis phases of DERA. Additional sampling may be required in subsequent phases of a risk assessment based on

preliminary results. Furthermore, risk assessments often require data collection from more than one field season (*e.g.*, seasonal data). Consequently, data collection occurs in several stages, including:

- Stage I PSI – AECs and COPCs are identified;
- Stage II PSI or DSI – Confirmation of contamination, profiling of spatial extent, and primary field work to support risk assessment; and,
- Supplemental Site Investigations (SSI) – additional sampling for risk assessment purposes (may be tiered).

An SSI based on a formal sampling plan is often conducted for specific data requirements of a risk assessment. A "limited SSI" can be used prior to the formal ERA studies to evaluate methods/explore site issues at a reconnaissance level.

Many risk assessments are conducted outside of the classic site investigation process developed for soil and groundwater contamination of upland properties (*e.g.*, risk assessments of water lots). For these sites, the principles of site investigation (*e.g.,* historical review to identify APECs and COPCs) should still be included in the initial stages of the project to ensure that APECs are assessed in the risk assessment.

Sampling for the risk assessment at such sites should also consider policy decisions (*e.g.*, beneficial structures, storm drains) made by the MOE so that the risk assessment addresses the desired objectives. Protocol 13 (SLRA) provides guidance on eligible beneficial uses that are not considered to constitute an unacceptable risk.

Policy Decision: Beneficial Uses – MOE has acknowledged that certain physical or engineered features in urban settings provide specific societal values and that sampling conducted under the CSR framework should not target these features. Policy also exists for the radius of an eligible beneficial use and maximum applicable site size. The practitioner should consult SLRA (Protocol 13) and MOE policy for additional details.

2.3 Policies, Protocols, Procedures and Technical Guidance

In addition to the provincial statutes and supporting regulations, evaluation of contaminated sites is also subject to provincial policies, protocols, procedures and technical guidance that direct site assessment and remediation. Some of these are legally required under the CSR. The objective of this section is to indicate how these documents are aligned with DERA, as opposed to providing an in-depth summary of all applicable policies, protocols, procedures and technical guidance.

2.3.1 Policies

MOE has adopted policies for contaminated sites that focus on scientific, technical and legal policy decisions. Some of these policies relate to the Contaminated Sites Soil Task Group (CSST) protocol, liability provisions, and precedent established in regulatory standards development. However, the most pertinent policy considerations from a DERA perspective are updates to the Ecological Risk Assessment *Policy Decision Summary*[9], which summarizes the key science policy issues. MOE policy decision summaries originated during the development of the *Guidance and Checklist for Tier 1 Ecological Risk Assessment of Contaminated Sites in British Columbia* (ERAGT, 1998). The *Policy Decision Summary* is updated periodically.

Because MOE policy is not static, but rather reflects the evolution of risk assessment practice over time, understanding of the treatment of policy considerations within this document is important for applying DERA guidance. Rather than discuss policy decisions in detail within this document, the remainder of this document treats policy issues as follows:

- A brief introduction to each policy issue is presented at the pertinent stage of DERA (*i.e.*, identification of the issue and why it is important to DERA practice);
- The issue is clearly identified as being the subject of provincial policy using text boxes (double-line border); and,
- The practitioner is reminded to consult the relevant MOE policy statement for details of the policy decision.

Risk assessment practitioners are expected to routinely consult the most recent MOE policy decision summary for updates applicable to their risk assessments, particularly as new policies may emerge that are not contemplated in this document.

2.3.2 Protocols

Protocols are technical procedures that are legally required under the CSR. Protocols include matters of a technical nature (*e.g.*, derivation of background concentrations of soils and groundwater) or a legal nature (*e.g.*, role of Contaminated Sites Approved Professionals, site security). At the time of publication, the following protocols relevant to DERA were available or in preparation:

[9] The current Policy Decision Summary is located at: http://www.env.gov.bc.ca/epd/remediation/policy_procedure_protocol/index.htm.

- **Protocol #6** describes the role of Contaminated Sites Approved Professionals (CSAPs). Protocol 6 (under Section 42[1] of the *EMA*) designates risk assessment application types that may be performed by an Approved Professional, and also identifies risk assessment attributes that are considered ineligible for CSAP review. Protocol 6 determinations are therefore relevant to the Reviews and Approvals aspect of DERA process, discussed in Section 2.5, below.

- **Protocol #12** describes the site risk classification system, providing procedures for classifying sites where contaminants occur at concentrations (and under conditions) that pose high risks to the environment or human health, under both current and reasonably anticipated future uses.

- **Protocol #13** describes SLRA[10], which is distinct in regulatory intent from DERA, but the two are linked in purpose and function within a broad risk assessment framework (Figure 3). In brief, SLRA provides a simplified risk assessment procedure to identify sites where substances exist above the numeric standards, but do not represent an unacceptable risk due to the absence of operable exposure pathways. DERA guidance applies to all ecological risk assessment activities outside the scope of SLRA, and also is relevant to the framing of all ecological risk assessments prior to selection of SLRA as a relevant process.

- **Protocol #20 (under preparation by MOE)** describes key components of DERA; the technical guidance behind the DERA protocol is provided herein.

2.3.3 Procedures

Procedures are internal directives used by MOE staff to ensure consistent administration of legislation and regulations. They are rarely applicable to the technical execution of DERAs.

2.3.4 Guidance

Practitioners should be familiar with MOE guidance documents prior to conducting risk assessments. The MOE technical guidance is not binding on practitioners or responsible parties, as the guidance is provided to advise (not to compel) on technical and scientific matters. However, practitioners should consider the administrative advantages (efficiency of review and approval) when contemplating deviations from the default guidance. Practitioners should provide rationale for deviations from guidance documents.

There are various forms of guidance that MOE makes available through their website[10], including technical guidance, administrative guidance, interim guidance and external guidance. Guidance will continue to expand and be updated; therefore, DERA practitioners need to be aware of updates.

Of the existing guidance documents, the following warrant special mention for ecological risk assessment:

- *Detailed Ecological Risk Assessment Technical Guidance* (TG-32; this document[11]). DERA guidance replaces previous ERA guidance in BC (ERAGT, 1998; known as "Tier 1" or "Protocol 1" (to be repealed). References to ERAGT (1998) are used for documentation purposes but this previous guidance should not be used after it is repealed.

- *Supplemental Guidance for Risk Assessments* (TG-7) – The purpose of this technical guidance document is to provide risk assessment practitioners with MOE-recommended sources of procedural guidance related to the performance of risk assessment and toxicity assessment. Recommendations are provided for toxicity reference value derivations, exposure assumptions, and guidance for probabilistic risk assessment.

- *Assessing and Managing Contaminated Sediments* (TG-19) – Supplemental guidance includes a general framework, guidelines for study design, and guidelines for interpretation of sediment quality assessments. A fourth module emphasizes issues specific to marine and estuarine environments.

2.4 Conceptual DERA Framework

Sections 2.1 through 2.3 describe the protocols and guidance used to support the risk assessment process in BC. These are drawn together during the overall DERA process in a stepwise strategy that is consistent with: (1) legal and formal procedural requirements in BC; (2) common risk assessment practice in BC; and (3) a traditional ERA approach.

The DERA process was adapted from an eight-step ERA guidance process for United States Environmental Protection Agency (USEPA) Region 5, taking into consideration BC circumstances. The process was simplified, with fewer formalized review stages relative to the EPA framework. As such, the DERA process can accommodate relatively simple sites efficiently while remaining applicable to larger and more complex sites. In the latter case, the detailed stages of the USEPA Region 5

[10] http://www.env.gov.bc.ca/epd/remediation/guidance/index.htm#3
[11] TG-32 marks the technical guidance number reserved for DERA; the document is awaiting final MOE review prior to being adopted as formal MOE guidance.

guidance (*e.g.*, study design, data quality objective development, and field sampling plan verification) may be incorporated and would be included in the Sampling and Analysis Plan (SAP) and Supplemental Data Gathering stages of the conceptual framework for DERA (described below and depicted in Figure 3).

2.4.1 Overview

Figure 3 illustrates the DERA process as applied to contaminated sites evaluation in BC, showing three aspects:

1. The outputs in risk assessment from a procedural or process point of view (left side of the figure);

2. The traditional ERA framework (the middle of the figure); and,

3. The applicable sections of this document that provide guidance (right side of the figure).

An important attribute of the DERA framework is the incorporation of multiple checkpoints. Checkpoints are "intended to ensure that site management decisions are made quickly, without the need for repeated studies, and entail meetings between the risk manager and the multi-disciplinary risk assessment team" (USDOE, 1998). Checkpoints are applied at key stages in the ERA process to evaluate, approve, and/or redirect efforts. This can be accomplished by obtaining buy-in from stakeholders for the selected path forward, or by conducting a formal deliverable review for completeness and validity. Overall, the purpose of checkpoints is to provide a mechanism for feedback on the acceptability of ERA assumptions, endpoints, strategies, and/or conclusions. The key decision is generally the acceptability of the work conducted to date for the intended purpose. In addition, the site project manager and scientific advisors decide upon what additional steps are necessary. In some cases it is important for practitioners to involve other regulatory agencies (*e.g.,* federal agencies) and stakeholders at the appropriate times, and the checkpoints provide a vehicle for this interaction.

Some key elements of the DERA process (Figure 3) are:

- *Manageable Number and Complexity of Checkpoints* – Attention through review at checkpoints[12] is focused on key stages such as SLRA, problem formulation and study design, and risk characterization. The objectives are to maximize the value of external reviews through good timing, and minimize unnecessary delays in project implementation, especially for simple sites. More complex and higher risk sites may warrant a higher-level of review at various points in the DERA process.

[12] In the EPA paradigm, checkpoints are called Scientific Management Decision Points, and represent a more formal and onerous process of review and feedback than contemplated under DERA.

- *Feedback Loops* – The framework explicitly includes a tiered approach to implementation and evaluation of data, in which subsequent phases of data collection and analysis are conducted only as warranted by results of the previous phase. Inclusion of an optional limited SSI prior to full ERA sampling (discussed in Section 2.4.3) facilitates timely collection of data to avoid delays in project completion. Revision of a problem formulation[13] through inclusion of such data is acceptable but not mandatory. Other feedback loops, such as refinement of a DERA through tiered stages of analysis and uncertainty reduction, are also possible.

2.4.2 Scoping

Prior to preparation of any formal risk assessment deliverables, the BC process requires a scoping stage resulting in a choice of SLRA or DERA. A preliminary site visit, review of site characterization data, and consideration of potential stakeholders' interests are examples of tasks that would support the scoping effort. The outputs of the scoping effort are: (1) a framing of the key environmental risk issues at the site, (2) a determination of which regulatory approach (adherence to numerical standards, SLRA, and/or DERA) is appropriate, and (3) determination of whether existing site characterization data are adequate to support a defensible problem formulation. Depending on the outcome of the scoping exercise, a DERA problem formulation may be required, either in entirety or to address remaining pathways following an SLRA.

The scoping assessment should consider whether the site has conditions that preclude the evaluation of the site under SLRA. CSR Protocol 6 (Eligibility of Applications for Review by Approved Professionals) identifies site conditions and risk assessment attributes deemed ineligible by MOE for the SLRA review process. These precluding conditions currently relate to evaluation methods (probabilistic analyses, toxicity tests, effects threshold derivations, and weight-of-evidence approaches) and also to site conditions (wild lands and off-site impacts). The practitioner should consult Protocol 6 for potential updates and to ensure that an SLRA is not contemplated for a site with precluding conditions.

2.4.3 Problem Formulation

The problem formulation (PF) stage is described in detail in Section 3 of this document. Section 3 includes discussion of eight discrete PF steps (denoted PF-1 through PF-8). PF is considered to be the most important stage of DERA because decisions made (and tools selected) during the PF set the scope and direction for the remainder of the risk assessment.

[13] Obvious data gaps may also be identified during the initial scoping step, such that preparation of an incomplete problem formulation can be avoided entirely.

For sites classified by MOE as "high risk" sites, completion of a formal document review at the PF stage is mandatory. For other sites, formal reviews of PFs are also recommended, commensurate with the risk and complexity of the site. Although it is at the discretion of the risk assessor and their client to pursue formal PF document review by an appropriate party (see Section 2.5.1), this practice is strongly encouraged.

During problem formulation, the requirement for supplemental environmental sampling must be assessed. If the available information (*e.g.*, DSI characterization) is sufficient to conduct the DERA, the practitioner may proceed directly to the Exposure Assessment and Effects Assessment stages[14]. However, problem formulations often identify information gaps that must be addressed prior to the technical analysis phases. These information gaps may be addressed using a formal Sampling and Analysis Plan (SAP) and/or a limited supplemental site investigation (*i.e.*, reconnaissance level SSI).

Formal Sampling and Analysis Plan

An SSI based on a formal sampling plan is often conducted for specific data requirements of a risk assessment. This includes preparation of a Sampling and Analysis Plan (SAP) that documents details of the information to be collected in support of the risk assessment. The SAP should include discussions of study design elements and must include data quality objectives (DQOs) that are linked to the measurement endpoints identified in the problem formulation. At minimum, DQOs are formal data quality specifications that must be tabulated within a quality assurance section. DQOs can also be applied at a broader level (conceptual study design level) to develop performance and acceptance criteria. Such criteria clarify study objectives, define the appropriate type of data, and specify tolerable levels of potential decision errors to be used as the basis for establishing the quality and quantity of data needed to guide site management (USEPA, 2006). The degree to which a SAP follows a regimented DQO process such as USEPA (2006) is partly a function of the complexity of the site and the quantity of data required to fill the information gaps.

Depending on the size and complexity of the project, the SAP may require formal review and approval[15]. SAPs are often combined with problem formulations in a single document (PF/SAP) for streamlined review. In preparing a PF/SAP, practitioners should ensure that the DQOs are compatible with the DERA needs; the British Columbia Field Sampling Manual (MOE, 2003b) should be consulted for a more complete discussion of QA/QC related to field sampling.

[14] The SAP and supplemental data gathering phases of DERA are marked as "optional" on Figure 3 because they are not required at all sites.
[15] Similar to PF requirements, formal SAP reviews are mandatory for high-risk sites defined by MOE, and strongly recommended for other sites.

Limited Supplemental Site Investigation

A "limited SSI" can be used prior to the formal ERA studies to evaluate methods and/or explore site issues at a reconnaissance level. If the DERA problem formulation indicates that limited supplemental sampling (*e.g.*, pilot-testing of sampling methods, evaluation of organism availability/biomass for tissue sampling, evaluation of potential reference areas) is advisable, such can be conducted prior to the formal detailed planning stage. The decision of whether to implement a limited SSI prior to the detailed planning stage depends greatly on the specific conditions at a site and on the status of the existing environmental investigation. Some considerations in making the above determination are:

- *DERA needs relative to CSR site investigation needs* – Risk assessment needs often extend beyond strict requirements of the contaminated site investigation process. For example, the preliminary problem formulation may identify that bioaccumulative COPCs are present. In these cases, determination of whether sufficient tissue mass can be obtained from field samples is an important issue for study design (*i.e.*, *in situ* samples versus laboratory bioaccumulation tests or modeled concentrations). Obtaining such information early in the process may assist in streamlining the risk assessment. In addition, chemical and/or biological assessment of samples collected during a limited SSI can yield data of relevance to a refined problem formulation.

- *Test protocol requirements* – Some analytical tests require that samples be obtained within certain ranges of physical parameters, such that limited sampling in advance of a major sampling effort is necessary. For example, Environment Canada protocols for marine amphipod toxicity testing require that particle size distribution be known prior to test species selection (Environment Canada, 1998). In other situations, information on salinity, soil pH, or other environmental characteristics may be desired prior to preparation of a detailed PF/SAP to help guide the selection of assessment tools.

- *Modification of site conditions* – If the exposure conditions have changed significantly from the time that the detailed site investigation was prepared (*e.g.*, due to voluntary remediation or other physical alterations), it may be appropriate to conduct targeted SSI activities to update the understanding of current site conditions.

- *Logistical issues* – Practical considerations may create an incentive for limited SSI analysis prior to detailed planning. For example, if seasonal constraints to sampling (winter freeze-up, growing season issues) will significantly delay sampling to support site-specific risk-assessment, it may be appropriate to fast-track the supplemental sampling. At remote locations, remobilization costs or availability of sampling crews may create an incentive to collect additional data earlier in the process.

- *Client need* – In some cases, the urgency of client timelines for achieving a risk characterization dictates that additional data are collected prior to development of a formal Sampling and Analysis Plan.

Where limited SSI sampling is contemplated, the practitioner and client should recognize the limitations of studies conducted outside a formalized PF/SAP process. Data collected within a systematic framework (whereby measurement endpoints are linked to study goals identified in the PF, and determined *a priori*) carry additional weight in risk characterizations relative to exploratory measurements.

2.4.4 Exposure and Effects Assessment

Technical guidance is provided in DERA Sections 4.0 and 5.0 (Exposure Assessment and Effects Assessment, respectively). There are also various appendices that provide tools for this analysis phase of the ERA. The selections of tools, and the staging (tiering) of the analyses, are highly site-specific.

2.4.5 Risk Characterization

The risk characterization results in conclusions and interpretations reached based on the analysis of the study information, including discussion of uncertainties. Where uncertainties are considered too great to support required risk management decisions, the assessment is considered incomplete and additional work will be required.

Technical guidance for risk characterization is provided in DERA Section 6.0 (Risk Characterization) and is supplemented by Appendix III (Interpretative Tools). The reporting, risk management, remediation, and monitoring phases of site management are not specifically covered in this DERA guidance. However, some broader aspects of these elements are addressed in Section 6.7 (Linking Risk Assessment with Risk Management).

2.4.6 The DERA Toolbox

Risk assessments incorporate various "tools" selected to meet the needs of the project. The number and complexity of the tools reflects the level of detail required in the risk assessment. In BC, an SLRA is based on simple and highly constrained tools (rules and models), whereas DERAs often require multiple tools of higher complexity in order to evaluate all exposure pathways at the desired level of certainty. We have organized the DERA "tools" into four categories, spanning the range of DERA inputs from raw data to high-level interpretative methods:

1. **Direct Measurement Tools**: This category consists of direct measurements that contribute raw data to support the exposure and/or effects assessment. Appendix I summarizes a range of direct measurement tools that are commonly applied in DERAs.

2. **Modelling Tools**: This category consists of quantitative methods used to: (a) provide estimates of exposure and/or effects where field data are unavailable (*i.e.*, surrogates for direct measurements); or (b) simulate the fate, bioavailability, or toxicity of stressors in the environment, using field data as inputs. Appendix II summarizes modelling tools commonly applied in DERAs.

3. **Interpretative Tools**: This category includes techniques used to evaluate the ecological significance of environmental data and/or model simulations. Interpretative tools provide the linkage between data and information; they are usually (but not always) quantitative, and they provide information targeted to the evaluation of the measurement endpoints. Appendix III summarizes interpretative tools commonly applied in DERAs.

4. **Synthesis Tools**: This category includes techniques used to integrate findings from multiple interpretative tools. These tools are applied mainly during the risk characterization phase. Synthesis tools include techniques for weight-of-evidence evaluation (WOE) as well as approaches for assessing uncertainty. Synthesis tools can be either qualitative or quantitative. Section 6.0 summarizes synthesis tools commonly applied in DERAs.

Many tools are generic (*i.e.*, they remain relatively consistent in terms of implementation and interpretation irrespective of the site at which they are applied). Others are more specialized. Discussion of the first three types of risk assessment tools is provided in the Appendices. Discussion of different synthesis tools is provided in the risk characterization section of this document (Section 6.0). This modular format is intended to: (a) facilitate a document format that can be readily updated with additional tools; (b) facilitate future expansion of the technical guidance for some tools (*i.e.*, increased level of detail); and (c) minimize redundancy within the DERA manual and streamline the organization of tool descriptions. Many of these tools have multiple uses within ERA and the selection and use of any given tool is at the discretion of the risk assessor, chosen on a site-specific basis.

2.4.7 Incorporating Land Use

Incorporation of land use factors in DERA is important because: (a) many risk-based decisions are linked to land use categories, and (b) the role of land use in the ERA process is evolving. Numerical CSR soil and water standards for use at contaminated sites are organized by land use, and MOE policy regarding permissible levels of impact are also land-use dependent.

Policy Decision: Land Use in DERA – The original trigger to enter the risk assessment process is often by reference to exceedances of standards that are land use based. Land use will continue to be a consideration for risk assessment policy on level of protection (and thus toxicity reference values and other measures of effect) and selection of species of concern (and thus conceptual model and sampling program). The practitioner should consult MOE policy for additional details.

DERA guidance assigns greater importance to regional ecology than to land use in the site conceptual model. This is not to say that land use is not considered — in fact, the degree of anthropogenic disturbance at a site (as expressed through land development) is a primary factor in terms of habitat availability and quality, which in turn influences selection of receptors of potential concern (ROPCs) and exposure pathways.

It is important that DERA practitioners consider land use in context of the surrounding landscape (*e.g.*, a commercial property in a rural area surrounded by natural areas, or a former mine site in a wildlands setting). However, federal and provincial guidance make no distinction in terms of the level of protection afforded to aquatic organisms, irrespective of the surrounding land use. Macfarlane *et al.* (2003) notes that "differences in land use activities do not influence the importance of sediments to benthic organisms."

Key Issues for the DERA Practitioner:

- The practitioner should characterize a site based on both the formal land-use designation and the ecological context (ecosystem type) of the site setting.
- If there appears to be a disconnect between the formal land use designation and the habitat or ecological setting (*e.g.*, small developed parcel of a site zoned commercial or industrial within a backdrop of a more natural setting), the MOE should be consulted for assistance in establishing relevant receptors and protection goals.
- For wildlands settings, consult MOE policy on risk assessment procedures.

Content for the DERA:

- Provide contaminant screening results based on land use designations, using the most conservative land use applicable to a given site.
- Conduct an evaluation of habitat and ecosystem type for all sites and incorporate this information in the development of the conceptual model for the site (receptors, pathways, *etc.*).
- Document the role of policy versus ecological considerations in the determination of the conceptual model

The difficulty in assigning all conceivable sites to discrete land use categories has been noted by members of the SAB task group and others. Examples of where considering land use alone would be inappropriate for a DERA include:

- Remote wilderness areas can be highly disturbed by linear industrial developments such as survey lines and pipeline right-of-ways; and,
- Large contaminated sites, by virtue of their size, can contain very important habitat features;

2.5 Reviews and Approvals

2.5.1 CSR Review Pathways

The checkpoints described in Section 2.4.1 provide an important aspect of the DERA process. Because of the iterative nature of site assessment and review in BC, and the range of review methods available under the existing legislation, flexibility is required in terms of how reviews and approvals are linked to the framework.

Currently, there are three approaches to review/approval of ERAs in BC:

1. MOE Review – high risk sites only, as defined by Protocol 12 (Draft; MOE, 2008).

2. CSAP Review – use of Approved Professionals, applies to sites as detailed in Protocol 6 (MOE, 2007b).

3. External Review – MOE external contractual review (applies to non-high risk sites that are not eligible for Protocol 6).

The DERA guidance is applicable to all three of the above approaches.

Policy Decision: Ministry Review – The Ministry has made a policy decision to focus their reviews of ecological risk assessments on high risk sites. Low and medium risk sites with certain attributes qualify for CSAP and external review. The Ministry has provided guidance to define low, medium and high risk site in Protocol 12: Site Risk Classification System (Draft; MOE, 2008). The practitioner should consult MOE guidance for additional details.

In the overall ERA process, checkpoint reviews should be completed by one of the three types of eligible reviewers.

2.5.2 Reviewer Independence and Role

The independence and role of the reviewer are described in the following bullets:

- Guidelines for maintaining independence of reviewer from site practitioner have been developed by the MOE and SAB.

- MOE Review (high risk sites) – role depends on whether ERA is being conducted under a Pollution Abatement and/or Remediation Order. For sites not under an Order, formal MOE review is sought at Problem Formulation and/or Sampling Analysis Plan (PF/SAP) stage, and following risk characterization. For sites under an Order, MOE Review will be agreed on a case-by-case basis.

- Draft Practice Guidelines for CSAP practitioners are published by the Society http://www.csapsociety.bc.ca

- Administrative Guidance for external reviewers is published by the MOE http://www.env.gov.bc.ca/epd/remediation/guidance/index.htm#2

2.6 Role of Other Jurisdictions

The DERA manual is intended to apply to sites under provincial jurisdiction. These DERAs are commonly conducted by consultants on behalf of property owners in a client-consultant relationship. This relationship defines the primary liaison in the design and implementation of the risk assessment. However, the practitioner needs to be aware of when either different jurisdictions apply (*e.g.*, federal lands, First Nations reserves) or when statutory requirements of other levels of government (*e.g.,* federal) may apply to the site and/or adjacent environments. Importantly, because these requirements can influence the scope and scale of inquiry or the selection of endpoints, the definition of the population potentially impacted, and the viability of certain risk management measures, the risk assessor should address these requirements during the problem formulation stage.

Notwithstanding that land management is a provincial responsibility; it is encumbent on site managers and contaminated sites professionals to adhere to all applicable statutory requirements. The engagement of all affected parties for review and feedback is advisable and may be mandatory.

The identification and inclusion of other stakeholders in risk assessment projects is the responsibility of the site owner and the risk assessor. Some of these stakeholders may include, but are not limited to:

- Department of Fisheries and Oceans – *Fisheries Act* Section 35 (harmful alteration of fish habitat, such as might occur during physical works associated with risk management) may apply. The federal government has the sole authority for "seacoast and inland fisheries" including their habitat (S. 91[12] *Constitution Act* [1867]). The inclusion of federal requirements (*e.g.*, assessment endpoints, measures of effects) early in the risk assessment (Problem Formulation) could become a pre-requisite should eventual risk management measures require federal approvals or works within/adjacent to fish habitat. Although risk management is not the focus of this DERA document, there may be advantages to the risk assessor (and responsible party) in identifying such issues early in the DERA process;

- Environment Canada – There are at least four potential statutory linkages of Environment Canada to DERA:

    - *Fisheries Act* Section 36 (deleterious substances – although the Minister of Fisheries and Oceans is accountable for this section, the administrative lead role has been assigned to Environment Canada). S.36 of the *Fisheries Act* prohibits the deposit or a deleterious substance into fish-frequented waters or into a place or conditions where they may enter such waters (*e.g.*, groundwater at a site near a surface water);

- *Migratory Birds Convention Act,* through Section 35 of the *Migratory Birds Regulation* provides protection for migratory birds from oil pollution. This act has previously been applied to oiling from free product and thus projects that pursue a risk assessment approach are unlikely to be dealing with free product reporting into migratory bird habitat;

- *Canadian Environmental Protection Act (CEPA)* – *CEPA* is an act respecting pollution prevention and the protection of the environment and human health from risks posed by toxic and other harmful substances. Prohibitions in *CEPA* are generally for specific substances that have been identified as "toxic"; and,

- *Species at Risk Act (SARA)* – The *Act* is a federal government commitment to prevent wildlife species from becoming extinct and secure the necessary actions for their recovery. Sites with federally-listed rare or endangered species may trigger *SARA* and require input from Environment Canada and the Canadian Wildlife Service. This act primarily applies to federal land; however, risk assessors should be aware of its implications on non-federal lands when certain migratory bird habitat is present or if listed aquatic species[16] are present.

- Canadian Environmental Assessment Agency – *Canadian Environmental Assessment Act*. This act would apply where there is a "trigger" that initiates a review such as the exercise of a regulatory function (*e.g.*, granting a *Navigable Waters Protection Act* permit) or where the federal government provides funding, in whole or in part).

- Transport Canada – There are at least two potential statutory linkages of Transport Canada to DERA:

  - *Transportation of Dangerous Goods Regulation,* which may apply to the movement of materials at a contaminated site; and,

  - *Navigable Waters Protection Act,* which could apply to physical works associated with risk management.

- Inter-governmental partnerships – Partnerships and administrative processes may offer single-window review (*e.g.,* Burrard Inlet Environmental Action Program [BIEAP], Fraser River Estuary Management Program [FREMP]). Not all such partnerships/administrative processes will address issues pertaining to contaminants. In those cases, the risk assessor should also work directly with the agencies that are members of such partnership processes.

[16] Under SARA, aquatic species includes those species that are a fish or marine plant as defined in the *Fisheries Act*.

- Port Authorities – Port Authorities may have authority through tenancy agreements *etc.*, and these could influence the selection of assessment endpoints or measures of effect. Certain Port Authorities may also have jurisdiction over effluent discharge issues (*Canada Shipping Act, Port Authority Operating Regulations*).

- Municipal and regional governments have a role in land use planning, zoning, *etc.* and are frequently a party identified by the MOE as a necessary component of consultation programs.

- First Nations – There is often an obligation in trust by federal and provincial governments to consult with First Nations and that obligation may be assigned to a risk assessor, particularly when that assessor is working within the context of provincial (and possibly other) requirements. With the site owner, the Risk Assessor should consider the need for First Nations consultation. Although guidance for First Nations consultation is beyond the scope of this document, practitioners may be assisted by MSRM (2004) and Government of British Columbia (2002).

- Neighbouring properties, particularly if there is migration of contamination off-site (either on land or to adjacent water bodies).

- Other stakeholders (case-dependent).

It is recommended that where possible a "single window" approach be taken to regulatory consultation, so that issues can be reviewed in the context of all stakeholders at the same time. It is the responsibility of the site owner/proponent and their consultant(s) to ensure the work at the site fulfils all regulatory requirements.

3.0 PROBLEM FORMULATION

3.1 Introduction

The problem formulation (PF) is the most important phase of any risk assessment. The level of effort required for the problem formulation is dependent on the complexity of the site. This guidance can be applied at any site (although for simple sites, SLRA will be an alternative route) and therefore encompasses a wide range of site conditions (size, complexity, ecosystems).

Although this problem formulation guidance is organized in a sequential manner, problem formulations are not linear in construction; they often require simultaneous consideration of multiple steps and may entail iterative refinements as site knowledge is obtained (see discussion of tiering and feedback loops in Section 2.4.1).

3.1.1 Problem Formulation Definition

The problem formulation phase is a planning and screening process that defines the feasibility, scope, and objectives for the risk assessment and provides an opportunity for consensus building. This process includes examination of scientific data and data needs, regulatory issues, and site-specific factors. The problem formulation identifies the ecosystems potentially at risk, the stressors, and the measurement and assessment endpoints. This information is summarized in a conceptual model that hypothesizes how the stressor(s) might affect the ecological components (*i.e.*, the individuals, populations, communities, or ecosystems of concern). Problem formulations have been defined elsewhere as follows:

- "Problem formulation is a process for generating and evaluating preliminary hypotheses about why ecological effects have occurred, or may occur, from human activities. It provides the foundation for the entire ecological risk assessment" (USEPA, 1998);

- The problem formulation "documents the key issues, establishes the breadth and depth of the problem, and initiates the process of prioritization… it documents the background for the decision to conduct an ERA" (CCME, 1996); and,

- "Problem formulation is a process of defining the nature of the problem to be solved and specifying the risk assessment needed to solve the problem" (Suter *et al.,* 2000).

In this guidance document, all of the above definitions are applicable. From a practical standpoint, the problem formulation is defined to include the following steps:

- Step PF-1: Scoping and Planning;
- Step PF-2: Review of Historical Documentation;
- Step PF-3: Identification of Contaminants of Potential Concern;
- Step PF-4: Identification of Exposure Pathways of Concern;
- Step PF-5: Identification of Receptors of Potential Concern;
- Step PF-6: Definition of Study Endpoints and Risk Hypotheses;
- Step PF-7: Development of a Conceptual Model; and,
- Step PF-8: Risk Assessment Strategy (Tool Selection, Sampling Design, Study Tiering, and Review).

These steps are discussed in detail in Sections 3.2 through 3.9, respectively.

3.1.2 Importance of Problem Formulation

Reviews of ERA case studies have concluded that the majority of difficulties documented in the case studies might have been avoided had more attention been paid to the problem formulation stage of the ERA (USEPA, 1993a). A well-constructed problem formulation reduces the likelihood of the following pitfalls in the resulting DERA:

- *Incompleteness* – The risk assessor incorrectly excludes pathways, receptors, contaminants, or analyses that are required to produce a defensible ERA.

- *Incorrect Study Framework or Evidence of Study Bias* – The risk assessor chooses ERA methods based on what is readily available, personal and/or professional experience, or anticipated outcomes and then tries to build a risk assessment framework around them. In these cases, problem formulations appear (improperly) as constructed around the technical content of the ERAs, rather than as a means of genuinely guiding the scope and objectives of the ERA.

- *Inconsistency or Lack of Objectivity* – The risk assessor develops an appropriate conceptual model and study endpoints during the problem formulation, but fails to follow through in an objective manner in subsequent ERA phases (*e.g.*, cherry-picking of effects metrics). Major decisions about how to interpret the data are made *post hoc* or without concurrence from interested parties. In these situations, the measurement and assessment endpoints are poorly aligned, and therefore key issues identified during problem formulation remain unaddressed.

- *Lack of Transparency* – The risk assessor does not provide sufficient rationale for methods, interpretations, or conclusions, as required for external reviews or project audits.

- *Technical Errors* – The risk assessor chooses or applies a technical tool incorrectly, or interprets results in a manner inconsistent with the science.

3.2 Step PF-1: Scoping and Planning

Most existing ERA guidance emphasizes the scientific aspects of risk assessment; however, there are a number of non-scientific risk management factors that can influence the nature of an ERA, including environmental policy considerations, management constraints (*e.g.*, project timelines), and the interests of other parties. Risk assessment should not be conducted in a vacuum from risk management. Rather, the role of risk management issues should be explicitly addressed in a transparent manner during the problem formulation phase, rather than deferred to the exposure and effects assessments. The respective roles of science and policy should be clear within the document. The following subsections summarize risk management issues for consideration in the DERA problem formulation.

3.2.1 Definition of Management Goals and Link to Risk Management

USEPA (1993a) concluded that risk assessments were frequently deficient in their articulation of management goals. Management goals, within the risk assessment framework, are defined as "desired characteristics of ecological values that the public wants to protect" (USEPA, 1998). This definition often results in vague narrative statements (*e.g.*, "protect ecosystem integrity") that provide little meaningful direction to a contaminated site ERA. The term "management goals" is used in DERA for consistency with other guidance manuals[17].

The purpose of management goals is to act as a practical statement in the DERA problem formulation regarding the objectives of the ERA with respect to site management. Management goals should be defined with input from site manager(s), in collaboration with the risk manager relative to business objectives and the applicable regulatory requirements (see also Section 6.7). Site management goals provide the overall framework under which assessment endpoints and measurement endpoints are developed. A management goal should not be confused with a protection goal, which is the desired level of protection for ecological receptors.

[17] Notably, CCME (1996) uses the terms "objectives of the ERA" instead of "management goals"

Site management goals may be relatively generic and stated at a high level (*e.g.*, "maintain a sustainable aquatic community adjacent to a public marina", or "risk assessment to support planning for redevelopment of a commercial site). In other cases, more specific management goals may be identified depending on the driver for the ERA – site management may differ for a site where there is a regulatory driver compared to a site where ERA is part of due diligence by the site owner. Two examples of management goals that guide the development of a risk assessment are provided for illustrative purposes:

- **Terrestrial:** "Determine whether the magnitude of soil contamination at the site requires remediation, or whether the magnitude of soil contamination is amenable to *in situ* management because risks to relevant receptors are found to be acceptable (with a high degree of certainty) for a future industrial land use."

- **Aquatic:** "To determine if the concentrations of COPCs in surface sediments do (or will) adversely affect the survival, growth, or reproduction of sediment-dwelling organisms"

The difference between the roles of the risk assessor and risk manager is as follows:

- The risk manager serves as the primary decision maker for a site; he/she uses the result of the risk assessment along with information on technical feasibility and social, economic and political concerns to reach a decision regarding the need for and scale of any management actions (such as remediation) (CCME, 1996).

- The risk assessor is responsible for the design and implementation of an ERA that meets the overall management goals for the site.

An advantage to clarifying management goals in the Problem Formulation is that it results in early discussion of risk management options at a given site. Risk managers and risk assessors are often considered to be separate parties. In practice, this separation is difficult to maintain (see also Section 6.7) but this goal should be aspired to. The challenge of separation is driven by several factors. First, a strategic client-consultant relationship is common for contaminated site ERAs in BC. Clients and site owners frequently request input from the risk assessor on issues such as technical feasibility and typical regulatory concerns; risk assessments are often bundled within a larger site management or remedial action plan document. A single risk manager is also unlikely: risk manager responsibilities are spread across multiple parties, including site owners, lead consultants, and/or regulatory agencies. Risk assessors need to be clear in their documentation where the information being presented brings in aspects of risk management.

Questions that risk assessors may ask people with risk management responsibilities (especially clients) to help formulate practical management goals include:

- Is the ERA intended to determine only if an unacceptable risk is currently present? For former industrial sites with historical contamination, the present condition may be the worst-case condition.

- Will it be necessary to develop a site-specific risk based standard for a particular contaminant that will be used for remediation?

- What is the range of future potential land uses for which the risk assessment is intended to be applied? How will future site development affect risk estimates?

- What is the desired level of certainty in the risk assessment conclusions? Risk assessments that are linked to compressed development schedules typically require greater certainty earlier in the ERA process because they are less amenable to tiered evaluations. The potential for residual liability may also influence the desired level of certainty.

Dialogue with risk managers (or client) regarding these topics at the beginning of the project is recommended, because it provides an opportunity for:

- The risk manager (or client) to communicate their expectations regarding the risk assessment process, which facilitates an understanding of budgetary and timing constraints as well as the nature of the relationship (if any) between the client and other interested parties; and

- The risk assessor to communicate the regulatory expectations and ecological considerations involved in a detailed ERA. The risk manager (or client) may not be aware of jurisdictional issues (Section 2.6) or of the need for the ERA to fully document the decision making process (rather than simply focusing on the perceived issues of importance).

Multiple management goals are possible at the beginning of the risk assessment (*i.e.*, *in situ* management, remediation to numerical standards, remediation to risk-based standards, or a combination of multiple approaches).

Key Issues for the DERA Practitioner:

- Do I understand why the client is doing this risk assessment, and have these needs been incorporated in the risk assessment design?

- Is the client aware of the legal and regulatory constraints that apply to the site evaluation (Section 2.6), and have these requirements/limitations been incorporated in the risk assessment design?

- Does the study design (*e.g.*, level of tiering of study components) correspond to the project schedule, if timelines are a significant limiting factor for risk management?

3.2.2 Selection of Appropriate Risk Protocol

As indicated in Figure 1 and Section 3.2.1, there are multiple options available to a responsible party in managing a contaminated site. Some options entail management to achieve numerical standards whereas others require development of risk-based standards. Once a decision is made to pursue risk-based standards, the first decision is whether a site will be subject to SLRA or DERA (see Figure 3). This decision is based on the eligible and precluding conditions described in SLRA guidance (Protocol 13). Where an SLRA has already been conducted, there are two possible outcomes:

1. The site has no operable pathways as determined by SLRA and the site exits the ERA process. A site that is captured by SLRA does not have to undergo a formal problem formulation, but following the process would be beneficial.

2. As determined by SLRA, operable pathways exist for the site or there is too much uncertainty to exit the process with an SLRA and the site moves to DERA. Once it is identified that a site will be within the DERA process, then the Problem Formulation guidance described in this section applies. Problem formulation must be completed before exposure and effects assessments have been implemented.

The scoping/planning phase may include situations for which an SLRA has not yet been conducted, and for which no precluding conditions have been identified. In these cases, the client/practitioner may opt to proceed directly to DERA without preparing a formal SLRA. Similarly, there may be aspects of site management for which it is preferable to invoke remediation rather than risk-based management (*e.g.*, where risk assessment costs are predicted to exceed active management costs). Administrative costs and time delays associated with managing small volumes of material in a risk-based framework should also be considered.

Policy Decision: Minimum Site Size for ERA – MOE is considering a definition of a minimum size of site (or volume of contaminated material) that is considered suitable for an ERA. For very small sites, the benefits of risk assessment decline relative to a standards-based management. When dealing with small sites, the practitioner should consult MOE policy for additional details.

3.2.3 Obtaining Input from Interested Parties

Most contaminated sites ERAs are conducted by consultants on behalf of property owners in a client-consultant relationship. This relationship defines the primary liaison in the design and implementation of the risk assessment. In addition to client input, risk assessments benefit from interactions with other interested parties, ranging from formal regulatory agency direction and/or advice, informal discussion, or public consultation (Section 2.6). Potential interested parties include:

- Provincial regulatory agencies (*e.g.*, BC Ministry of Environment [MOE] or their representative[18]);
- Federal regulatory agencies (*e.g.*, Department of Fisheries and Oceans [DFO]; Environment Canada; Canadian Wildlife Service [CWS]);
- First Nations;[19];
- Administrative bodies (*e.g.*, Vancouver Port Authority, Transport Canada, Fraser River Estuary Management Program, Burrard Inlet Environmental Review Committee);
- Other levels of government (*e.g.*, municipal; regional); and,
- Non-governmental organizations (*e.g.*, environmental groups; local community organizations).

All DERAs involve liaison with one or more interested parties, although the magnitude and formality of these interactions tends to be commensurate with the size, scope, and complexity of the project. Not all interested parties are applicable to all sites. For many sites (but not all), it is often sufficient to solicit input only from applicable regulatory agencies. The context for determining the appropriate involvement of interested parties varies depending on the following factors:

Applicable Jurisdictions

In British Columbia, environmental matters pertaining to contaminated sites generally fall under the jurisdiction of the provincial Ministry of Environment (MOE). Specific regulations relating to the assessment and remediation of contaminated sites include the CSR (BC Reg. 375/96, last amended in 2007), and the Hazardous Waste Regulation

[18] The proposed Contaminated Sites Approved Professional (CSAP) system may result in instances where MOE consultation is obtained using an approved professional as a representative.
[19] Consultation with First Nations is subject to an evolving legal landscape as well as ongoing government process.

(formerly called the Special Waste Regulation; BC Reg. 63/88). The level of input from other interested parties is influenced by CSR provisions (*e.g.*, public consultation [S55.1], off-site notification [S57.1] and the Public Notification Regulation). The practitioner should consult Section 2.6 of this document to assess the potential involvement of other jurisdictions or agencies. Consideration of the federal perspective is recommended, even for risk assessments conducted under provincial guidance.

In practice, most sites require some consideration of federal policy, regulation or legislation through one or more triggers. A rationale for (or against) the inclusion of federal perspectives, emphasizing the site-specific information available to support the decision is useful. For those sites that appear to have a federal trigger, it may be sufficient to simply document how the federal perspective was accommodated through reference to existing federal policies and regulations. Formal dialogue with regulatory agencies is not mandatory for all sites, but is recommended for those sites where management goals require federal regulatory approvals, or for those sites where a significant federal regulatory interest is likely to exist (*e.g.*, the site contains sensitive and/or abundant migratory bird or salmonid fish habitat). To this end, the problem formulation is an opportunity to confirm that all regulatory interests have been accommodated prior to detailed study design.

Key Issues for the DERA Practitioner:

- Identify the lead regulatory agency for the risk assessment (*e.g.*, MOE).
- Determine whether the site is sufficiently complex to warrant formal dialogue prior to preparation of the problem formulation.
- Does the risk assessment connect with other jurisdiction's environmental regulatory issues at the site? If so, is formal liaison required to address these issues?
- What level of documentation will be necessary to solicit input from other interested parties? Examples include the problem formulation document for technical review, informal site visit, or a "briefing note" summary.

Project Timelines

Where possible, the project timeline should allow formal input from other interested parties (regulatory or otherwise). Formal input on a site often requires sufficient documentation and provision of a review/comment period (*i.e.*, initial assessment checkpoint). Site visits and kick-off meetings may provide a means to obtain informal input regarding the scope of the risk assessment. Ongoing informal dialogue is also advantageous.

Key Issues for the DERA Practitioner:

- Do project schedule constraints limit the level of interaction with other interested parties, including regulatory authorities?
- Is formal review of the problem formulation by outside parties required? If not, how will involvement of those parties be organized?
- Is the client aware of the uncertainties associated with postponing regulatory interactions until later in the risk assessment process?

3.2.4 Assembling a Study Team

The complexity of the study team and degree of specialization required are project specific, although a multidisciplinary study team[20] is typically required. Not all of the scientists involved need be experienced in risk assessment, provided that an experienced risk assessor is involved in coordination and report preparation. The appropriate level of professional designations (*e.g.*, R.P.Bio., P.Eng.), academic credentials (B.Sc.; M.Sc.; Ph.D. and specialties within), and documented expertise in a given discipline (or subdiscipline) for the study team must be considered.

Key Issues for the DERA Practitioner:

- Identify which specialties will likely be required to successfully complete the DERA, and where possible, involve those people in the preparation of the problem formulation.
- Confirm that proposed team members have the professional qualifications to conduct the proposed tasks, particularly where such tasks entail field data collections.

3.3 Step PF-2: Review Historical Documentation

The problem formulation provides an opportunity to consolidate and consider all relevant site characterization information, including:

- Stage I and II preliminary site investigations (PSI);
- Detailed site investigations (DSI);
- Environmental impact assessments;
- Physical, chemical, and/or biological monitoring reports; and,
- Previous ecological or human health risk assessments (screening-level or other).

[20] Examples include toxicology, ecology, fisheries/wildlife biology, botany, forestry, limnology, geology/hydrogeology, chemistry, environmental modelling, statistics and geographic information specialists.

PSIs and DSIs are often available for contaminated sites prior to initiation of the DERA. Other documents should be reviewed where available because biological data are often not incorporated in PSIs or DSIs. Biological data may be available in documents not directly connected to the contaminated site investigation. For example, a baseline environmental assessment for a development project[21] dealing with regional or watershed-level information may contain relevant ecological and biological information applicable to a contaminated site within the watershed. Other biological data sources include the Burrard Inlet Environment Review Committee project archives; MOE reports; and other multi-agency watershed level programs. Institutional libraries (*e.g.*, regulatory agencies; universities) are also potential sources of information.

3.3.1 Summarize Site Information

The location and details of the site should be clearly designated, and should include the following information.

- *Client Information* – name of site owner(s), company affiliation, and contact information.

- *Site Description* – name of property and primary activity conducted on site (past and present).

- *Site Location* – street address, municipality, geographical coordinates (*e.g.*, latitude/longitude or northing/easting [specify datum]).

- *Legal Description of Property* – lot, block, district lot, plan number, and/or property identification number (PID)

3.3.2 Review Previous Ecological Risk Assessments

All ecological risk assessments previously conducted for the site must be reviewed during the problem formulation. Several scenarios exist in this regard:

- An SLRA was completed following provincial risk assessment guidance and led to the initiation of the DERA – The risk assessor should review the SLRA in terms of its methodologies and conclusions and confirm its decisions regarding exclusions of receptors, pathways or contaminants from the DERA.

[21] BC Environmental Assessment Office (http://www.eao.gov.bc.ca/).

- An ERA was conducted for the site based on provincial Tier 1 or other ERA guidance – The risk assessor should determine which receptors, pathways or contaminants may be screened with confidence from further consideration.

- An SLRA was not completed (*i.e.*, the screening ERA stages were skipped for efficiency) – In these instances, the risk assessor is limited to the historical documentation described above.

Key Issues for the DERA Practitioner:

- Does the available documentation provide sufficient information about the ecology of the site to support the selection of the exposure pathways and receptors of concern?

- Is a site visit and/or habitat characterization by a professional biologist necessary to confirm or supplement the available ecological information?

- Is the biological characterization of the site limited to the legal site boundaries, or does it include descriptions of habitats in adjacent land parcels?

Content for the DERA:

- A narrative or tabular summary of each previous risk assessment should be provided in terms of receptors, pathways, contaminants (and/or physical stressors), risk assessment tools used, major conclusions, areas of uncertainty and recommendations for future work.

- A summary statement for each previous risk assessment should be provided, indicating agreement with the conclusions (or, if disagreement, a rationale for that determination).

3.3.3 Determine Applicable Ecosystem Type(s)

Site ecology is the primary factor to consider when developing, implementing and interpreting a detailed ERA. USEPA (1992) states that "knowledge of the ecosystem[22] potentially at risk can help identify ecological components that may be affected and stress-ecosystem interactions relevant to developing exposure scenarios." The following generic ecosystem types were developed based on commonly observed and broad differences in the biotic communities and exposure pathways (Figure 4):

[22] Ecosystem is defined as the biotic community and abiotic environment within a specified location in space and time (USEPA, 1998).

- **Deep Aquatic**: Deep aquatic ecosystems include subtidal marine areas and lake bottoms. These ecosystem types tend to have relatively stable sediments with depositional environments. Deep Aquatic ecosystems can be found in both freshwater and marine environments.

- **Shoreline**: Shoreline ecosystems include intertidal areas, shallow estuarine environments, wetlands, marshes, and rocky shorelines. These ecosystems typically reflect a dynamic and transitional environment (*e.g.*, freshwater to marine; tidal changes). Groundwater flux from upland areas to the aquatic receiving environment is often an important exposure pathway for this ecosystem type.

- **Rivers and Streams**: Freshwater environments with flowing water, often associated with more dynamic substrates.

- **Upland Terrestrial (Wildlands)**: Relatively natural terrestrial ecosystems with minimal direct anthropogenic influence. This ecosystem type can vary greatly in British Columbia (*e.g.*, coastal rainforests; high alpine meadows; semi-arid; montane).

- **Upland Terrestrial (Human Use)**: Terrestrial ecosystems that are significantly influenced by human activities. The degree of anthropogenic influence is reflected by land use considerations. For this ecosystem type, the magnitude and type of human use influences both the ecological setting and the protection goals of the ERA. Land use types are organized based on the prevailing land use classifications specified in both the CSR and the previous Tier 1 guidance for ERA (repealed). The land use types of industrial, commercial, residential, urban park, and agricultural may be viewed as subtypes of the upland terrestrial ecosystem type.

Policy Decision: Land Use and Linkage to Ecosystem Type – Selection of an ecosystem type (or types) is appropriate and is used to guide the selection of techniques for risk assessment and specification of the conceptual model. However, the practitioner should be aware that land uses (considered as subtypes above) are considered to be important to MOE in terms of compliance with standards, selection of receptors, and protection goals. Consult MOE policy for details.

These generic ecosystem types are provided as a starting point—combinations of multiple ecosystem types and transitional subtypes within a single site also exist. In some cases, these transitional ecosystem subtypes may be of significant interest in the DERA (*e.g.*, a riparian setback surrounding a stream may require consideration of study components from each of the "shoreline", "rivers and streams" and "wildlands" ecosystem types). A site-specific conceptual model should incorporate relevant components of one or more of the generic ecosystems above as needed.

Content for the DERA:

- The risk assessor should determine which among the five generic ecosystem types (or transitional ecosystem types) are applicable to the site in terms of quantity and configuration of existing habitat. The proportion of the total site area in each category and proximity of site habitats to habitats on adjacent land parcels is important.

- A brief description of relevant meteorological data (*e.g.*, seasonal trends; temperature ranges; rainfall) and the biogeoclimatic classification should be included because it provides context to the selection of ROPCs.

Key Issues for the DERA Practitioner:

- There may be insufficient information available to properly evaluate ecosystem types. Other sources of information (site visits; professional judgment based on relevant experience) may be necessary.

- If available, habitat mapping data should be used to supplement the characterization of ecosystem types. For example, habitat inventory and classification maps have been produced for FREMP[1] that show classes of intertidal and riparian habitat types and rate their biological productivity and suitability for development. Provincial wildlife habitat classification guidance is also available.

3.3.4 Summarize Site History

Site history, with emphasis on historical site uses linked to use or distribution of contaminants, should be summarized in the problem formulation. Site history is generally considered in detail in a site investigation; in these cases a brief review of the site history in the problem formulation will suffice. The review should consider:

- Historical subdivision or amalgamation of land parcels (*i.e.*, is the study area made up of many smaller properties, or was the site subdivided from other historical lots?);

- Approximate locations of former buildings and site operations in relation to soil, sediment, water, and biota; and,

- Historical activities on adjacent or nearby properties that may result in potential off-site contamination sources.

Information on historical site uses is primarily intended to allow the risk assessor to conduct a reality check on the adequacy of the available site information to support an ERA exposure assessment.

Content for the DERA:

- A narrative or tabular summary of site history, along with implications for the design of the DERA;
- Identification of site activities that may have altered the distribution or concentration of contaminants of potential concern (COPCs);
- Identification of COPCs that were not considered in previous site investigations; and,
- Identification of regional contamination issues if applicable.

Key Issues for the DERA Practitioner:

- What is the potential for on-site and off-site migration of contaminants at concentrations of potential environmental concern?
- Is there site-specific information relevant to bioavailability and/or mobility of contaminants that is not reflected in bulk chemistry measurements? (For example, PAHs associated with black soot particles and metals associated with grit particles tend to be less bioavailable).
- Does existing information provide sufficient detail to develop a comprehensive list of COPCs?
- Is the pattern of site contamination linked to historical site uses?

3.3.5 Evaluate Applicable Land Use(s)

Land use (historic, present and future) of the site is an important factor to consider when developing, implementing and interpreting a DERA. Land use governs the process used in SLRAs that follow guidance from Protocol 13. Land use classifications are particularly important for the uplands [human use] ecosystem type, because land use dictates specific ERA attributes, including level of protection for various receptor types.

Policy Decision: Adjacent Land Uses and Protection Goals – The determination of applicable land use in urban settings is relatively straightforward, but is more challenging in rural and semi-natural settings, particularly where the land use within property boundaries is distinctly different from the surroundings. In these cases, the land zoning classification for the site in question may be poorly aligned with the surrounding ecosystem, creating a potential disconnect between the ecosystem-based approach and the land-use classifications. The determination of applicable standards, selection of relevant receptors, and protection goals is land-use dependent, and these decisions are governed by MOE policy. The practitioner should evaluate both ecosystem types and land uses on and surrounding the site, and consult with MOE policy (or MOE representatives) to reach a final determination. Note that wildlands assessments are specific area of MOE policy development.

Land use is a less significant factor in the design of DERA for the aquatic ecosystem types (deep aquatic, shoreline, or river and stream). The protection goals for aquatic environments are based primarily upon habitat type and sensitivity, and only secondarily upon human use of the habitat (MOE, 2003a).

Policy Decision: Designation of Water Lots as Typical Contaminated Sites – MOE has identified factors for consideration in the designation of typical contaminated sites (TCS). Schedule 2 of MOE (2003a) identifies that marinas, docks, wharves and associated infrastructure located within these areas may be assessed making use of the typical criteria limits, which convey a lower degree of protection relative to sensitive sites. MOE requires that proponents present information to support their proposal to the appropriate agencies, including identification of existing resources in the area, the identification of offsite contaminant sources, and measures taken to eliminate onsite sources of contamination. Note that the default designation for aquatic habitats that are important to fish spawning or serve as important rearing habitat for fish is a sensitive contaminated site (SCS) designation (MOE, 2003a).

Incorporation of land use considerations in the uplands (terrestrial) ecosystem types may be complicated when a particular site does not "fit" well into the local mosaic of land use types[23]. Consequently, site ecology should be the primary consideration in the design of a DERA.

[23] Land use considerations are relatively straightforward in cases where the site is fully developed and is situated within a landscape of other, fully developed properties (*e.g.*, an industrial site within an industrial park; other properties in urbanized areas). Land use implications for the design of a DERA are less clear when sites are either partially undeveloped or decomissioned (*e.g.*., an undeveloped area zoned for residential use, but unlikely to be developed in the near future; a disused industrial property along a river). Land use implications are also problematic when the context of the surrounding landscape is considered (*e.g.*, a commercial property in a rural area surrounded by natural areas).

Content for the DERA:

- A summary of the current (and likely future) land uses within each ecosystem type.
- Discussion of land uses beyond the legal boundaries of the site but relevant to mobile receptors that cross site boundaries (*i.e.*, regional ecological setting, sites adjacent to or influencing surface waters).

Key Issues for the DERA Practitioner:

- Land use classifications based only on land-use zoning may be inadequate for evaluating the ecological attributes of a particular site (or subareas within a large site). Property boundaries are not the same as ecological boundaries.
- The context of the surrounding land uses should also be considered in terms of its implications. An industrial site bordering on sensitive and valued aquatic habitat (*e.g.*, wetland) does not have the same ecological attributes as an industrial site bordered by other industrial sites.
- Changes in land use (whether due to human use or natural succession) should receive special attention, to ensure that appropriate risks are assessed as the site transitions from one use to another.

3.3.6 Summarize Site Chemistry

A summary of the available site chemistry should be included in the problem formulation; it provides a basis for understanding the type and magnitude of contamination, and logically leads to the identification of COPCs (Section 3.4). The following summaries of site chemistry are generally required:

- A narrative or tabular summary of concentrations measured in the different environmental media sampled to date, including description of minimum and maximum concentrations, summary statistics (*e.g.*, 95% upper confidence limit of the mean, 90th percentile, mean, median), percentage of non-detects and treatment of non-detects for purposes of statistics, and sample size. This site chemistry summary is typically included in the historical document review to demonstrate familiarity with previous site investigations, and to document the underlying trends in the available chemistry data.

- An Excel-based or database system containing the results of individual analyses. This data summary is used to identify COPCs and will typically include coordinates to facilitate map or GIS-based presentation. Depending on the site and complexity of the site, compilation of the data in this format is recommended given their importance elsewhere in the problem formulation.

- A brief narrative describing the spatial and temporal variations in chemistry distributions should be provided, particularly as they relate to representativeness and sampling design for additional investigations.

QA/QC should be reviewed to determine if available site chemistry data are appropriate for the risk assessment. Issues include sample collection and storage methods, selection of analytical methods, performance of analytical QA/QC measures such as laboratory duplicates, matrix spikes and use of certified reference material, and the use of appropriate analytical detection limits. The chemistry data must conform to the official BC Lab manual methods or other methods acceptable to the Director. Laboratories providing data for use in DERA must also comply with the requirements of the Environmental Data Quality Assurance Regulation (BC Reg. 301/90; MOE, 2004).

Data without detailed QA/QC documentation may be rejected or utilized (with appropriate discussion of its uncertainty) at the discretion of the risk assessor; however, a data set that consists primarily of unverified data indicates that confirmatory sampling as part of the DERA is likely warranted.

Policy Decision: Summaries of Site Data – MOE provides policy and guidance on methods related to summary of site chemistry data. Policy determinations have been made regarding minimum sample size for site characterization here, minimum analytical detection limits, and statistical measures used (*e.g.*, 95% upper confidence limit of the mean (or alternative central tendency measures). The practitioners should evaluate the historical data from the viewpoint of provincial policy and determine whether data are adequate to satisfy the needs of the risk assessment. Lack of concordance with policy determinations should be clearly identified and included as information gaps in the problem formulation. Some deficiencies from source reports can be easily addressed (*e.g.*, central tendency measure recalculated), whereas others (high detection limits) may have implications for contaminant screening or study design.

Content for the DERA:

- A narrative, tabular or graphical summary of the available chemistry data for each medium. This overview should be linked to site history and describe potential or suspected contaminant sources.

- A spreadsheet or database containing the individual analytical results for use in screening of COPCs and graphical presentation.

- A brief summary of the spatial distribution of chemistry parameters.

Key Issues for the DERA Practitioner:

- Analytical detection limits for site characterization samples should be reviewed for environmental relevance.

- Data should be evaluated against MOE policy for the characterization of site chemistry data in risk assessments, and information gaps highlighted.

- Citation of software used and assumptions about distributions, *etc.* should be provided (this information is required during reviews to ensure that the appropriate statistical tests and assumptions were used).

- Ancillary data needed to interpret bulk chemistry data (*e.g.*, pH or hardness for metal concentrations) or facilitate other decision making within the DERA (*e.g.*, grain size and total organic carbon data in sediment to facilitate toxicity test species selection) may not be available. These data gaps will need to be addressed as part of the DERA.

3.3.7 Site Overview Map

A site overview map (often present in the DSI) should be modified as necessary to include the following information:

- Legal site boundaries and identification of adjacent properties. Placement of the specific study area within a regional context (in a smaller map window) is recommended.

- Locations of historical site buildings, areas of potential concern (APECs), and zones of known contamination.

- Locations of individual historical sample locations.

- Locations of other relevant site features, such as transportation corridors, water bodies, changes in topography and significant habitat features.

Content for the DERA:

- A geographical representation of the data presented in the problem formulation is strongly recommended.

Key Issues for the DERA Practitioner:

- Geographical representations facilitate examination of the adequacy of the existing spatial coverage of chemistry data relative to known or suspected contaminant sources as well as significant ecological features. Assessment of spatial coverage is supported by these geographic representations.

- GIS-based approaches facilitate the integration of data management and mapping, and are advantageous in terms of spatial analyses of chemistry [and other] data as well as risk communication.

3.4 Step PF-3: Identify Contaminants of Potential Concern (COPCs)

COPCs are selected primarily based on comparison of the available site data to the applicable numerical guidelines, standards or criteria values[24]. The presence of one or more samples with a concentration that exceeds these numerical values results in the selection of that analyte as a COPC. However, analytical chemistry data may not always be adequate for COPC selection. Professional judgment may be required to ensure that potentially relevant COPCs are not excluded due to lack of data. In general, a COPC should be retained for further evaluation unless sufficient information is available to warrant its exclusion. Examples of how professional judgment and provincial policy should be applied in COPC screening are provided below.

Inadequate Chemistry Characterization – COPC selection requires that the site has been adequately characterized. Completion of a DSI is assumed to represent an adequate characterization in terms of spatial coverage; however, a DSI may still have data gaps in terms of the adequacy of data relative to the specific exposure pathways. For example, if terrestrial exposure pathways are being evaluated for a site that will not be disturbed under its future land use, then COPC selection should be based primarily on surface soil conditions.

[24] This section uses the term "guidelines" in lieu of "guidelines, standards and criteria".

DSIs often contain chemistry data that may not be representative of ecologically relevant exposures, in part, due to one or more of the following factors:

- Soil data from DSIs may be from depths greater than a biologically relevant depth or composite soil samples from a range of depths. Non-composite samples from the upper 15 cm (subject to MOE policy discussed below) may be required to supplement the DSI.

Policy Decision: Plant Root Zone – The specification of 15 cm of soil to represent the plant root zone (excluding sites with deep tap roots) is a policy decision that is under active discussion. The practitioner should review current policy prior to specification of discrete sampling intervals. Related Ministry policy issues include the process for determining whether a deep tap root pathway is of concern, and the depth of engineered cover required to eliminate this pathway in a risk management plan (or process for determining such a depth).

- Construction, landscaping and/or remediation activities may result in a future surface soil horizon that is different than the surface soil characterized in the DSI.

- Sampling density in the DSI may not be appropriate relative to the foraging ranges and preferred habitats of site receptors.

Policy Decision: Minimum Sample Size – MOE provides recommended minimum sample sizes for environmental characterization of each medium of concern (soil, water, plants, *etc.*). MOE recognizes the large variability among potentially contaminated sites in regards to size, heterogeneity of the environment, and spatial patterns of contaminants and receptors, which precludes a single sampling design or inflexible rule regarding sample size. However, MOE recommends minimum numbers of samples as a default position. The practitioner should consult MOE policy on this subject, and provide clear rationales for deviation from the default policy.

Analytical Detection Limits – Chemistry data may have analytical detection limits that exceed the applicable numerical guidelines. Compounds that have analytical detection limits greater than guideline values should be retained as COPCs until confirmatory analyses with appropriate analytical detection limits can be conducted. Ideally, analytical detection limits should be less than the numerical guidelines by a factor of 10, subject to technical considerations. Where analytical or matrix limitations make this impractical, reference should be made to MOE policy.

Policy Decision: Analytical Detection Limits – MOE provides policy on the selection of detection limits in relation to guidelines and toxicity reference values. This should be taken into account in the development of and recommendation of analytical protocols.

Numerical Guideline Value Unavailable – COPCs should not be prematurely excluded based on a lack of CSR standards[25]. If CSR standards are not available, provincial ambient guidelines, numerical guidelines from other jurisdictions (*e.g.*, Canadian Council of Ministers of the Environment [CCME], United States Environmental Protection Agency [USEPA], Washington Department of Ecology), or toxicity values from the literature can be adopted. If used, values from other jurisdictions need to be fully documented, justified and ideally pre-approved by MOE. The degree to which the derivation procedures reflect the protection goals of the provincial CSR standards should be considered. Anthropogenic compounds present at quantifiable concentrations but without environmental quality guidelines should be retained as COPCs unless a sufficient technical argument can be made for their exclusion. Consultation with MOE may be appropriate to confirm that COPCs have not been deliberately eliminated from consideration for policy reasons.

Technical considerations in the screening of COPCs include:

- A review of the transport, fate and effects of COPCs conducted, to provide information for identification of exposure pathways (Step PF-4) and receptors of potential concern (Step PF-5).

- Some COPCs can be eliminated from consideration for certain pathways based on environment fate properties. For example, volatile organic compounds may be screened out of a food-web bioaccumulation pathway, because these chemicals rarely accumulate in organism tissues at levels of environmental concern. Organic compounds with high Henry's Law Constant values (H) means they readily partition to air, while compounds with low K_{OW} values means they tend to be highly water soluble (and therefore readily excreted).

Policy Decision: Defining Bioaccumulative Substances – MOE is developing policy on the identification of contaminants that should be considered strongly bioaccumulative and/or potential biomagnifiers. This will include consideration of K_{OW} thresholds for organic substances. There are also various efforts to develop policy on defining bioaccumulation in other jurisdictions and which lines of evidence for risk evaluation are appropriate. The practitioner should consult MOE policy for updates to this issue.

[25] CSR Schedule 10 notes that standards are protective of human health and emphasizes that ecological protection using a schedule 10 standard remains the responsibility of the Responsible Person.

- Some COPCs can be eliminated from quantitative evaluation provided that a related contaminant with higher toxicity and environmental concentration is available for comparison to environmental quality guidelines. For example, the toxic equivalency (TEQ) model is a technically defensible process for evaluating the combined effects of dioxin-like chemicals (*e.g.*, dioxins, furans, coplanar PCBs). Conservative mixture models may also be applied to address aromatic and aliphatic constituents of petroleum-related organic compounds. Full documentation of the rationale would be required and would receive scrutiny during review.

- An ecological relevance check can be conducted to assess whether the list of COPCs can be reduced. In some cases, the relevance check amounts to the application of common sense. For example, chloride may be eliminated from the list of COPCs for marine environments because it is a naturally occurring substance in high concentrations in seawater. In other cases, the relevance check is less intuitive and requires supporting evidence from peer-reviewed literature.

- In general, contaminants should be retained as COPCs if site history or other data indicate concentrations at elevated concentrations relative to background conditions are likely. For example, elevated concentrations of resin acids and fatty acids in the vicinity of pulp mill operations would warrant their inclusion as COPCs even though environmental quality guidelines for these substances are lacking. Metals should be retained if the pattern of their distribution suggests that anthropogenic influences have resulted in increased concentration or mobilization.

"Conventional" parameters (*e.g.*, sediment ammonia and sulphide concentration; water pH or hardness; soil or sediment organic carbon content; soil pH) that may mediate biological responses should be assessed even though these parameters may not have applicable guidelines.

Role of Background Concentrations – Provincial guidance (*e.g.*, CSR Protocols 4 and 9)[26] provides methods for the determination of background soil and groundwater conditions. An analyte should not be selected as a COPC if concentrations at the site are less than background (as determined by CSR protocol) and the background determination conducted under CSR Protocols 4 or 9 has been approved by the MOE. COPC selection in the DERA should describe that analyte concentrations exceeded the applicable numerical guideline, but not the background concentration. The background determination should be included as an appendix to the DERA or, at a minimum, cited.

[26] http://www.env.gov.bc.ca/epd/remediation/policy_procedure_protocol/index.htm

Content for the DERA:

- The practitioner should provide a narrative or tabular summary of each COPC considered during the screening phase, along with a rationale for its inclusion or exclusion.

- Arguments for the exclusion of COPCs based on environmental transport and fate, ecological relevance, or background considerations must be fully documented in the DERA.

Key Issues for the DERA Practitioner:

- It is a technical error to exclude COPCs simply because CSR numerical standards are not available.

- It is preferable to conservatively include a COPC even if professional judgment suggests that potential risks associated with the COPC are low.

- It is also preferable to retain a COPC for which there are scant environmental effects data and discuss the data limitations in the uncertainty assessment, as opposed to eliminating the contaminant based on lack of detailed information.

- CSR Schedule 10 lists generic soil and water standards specific to human health, but notes it "is the responsibility of the responsible person for the site to ensure that the use of the soil or water standards…do not constitute a significant risk or hazard to ecological health." Compounds listed on Schedule 10 should be included as COPCs if present at the site.

- DERAs are frequently tailored to reflect COPC-specific issues. Additional information regarding DERAs for metals, hydrocarbons and other contaminant groups is available in the literature.

3.5 Step PF-4: Identify Exposure Pathways of Concern

The following exposure pathways of concern should be considered:

- Soil invertebrates and terrestrial plants are in direct contact with elevated COPC concentrations in soil;

- Mammals, birds, amphibians and reptiles ingest elevated COPC concentrations via consumption of prey items. (Note: relevant prey items vary according to receptor);

- Mammals, birds, and amphibians and reptiles ingest elevated COPC concentrations via water ingestion;

- Mammals, birds, and amphibians and reptiles ingest elevated COPC concentrations via incidental soil/sediment ingestion;

- Aquatic species (macrophytes, plankton, invertebrates, and fish) are in direct contact with elevated COPC concentrations in surface water and/or sediment [Note: the proportion of surface water and sediment contact varies according to receptor]; and,

- Some aquatic species (*e.g.*, planktivores, piscivores) ingest elevated COPC concentrations via consumption of prey items.

Policy Decision: Inhalation and Dermal Contact Pathways – MOE has provided policy and rationale for elimination of inhalation and dermal exposure routes for wildlife. The practitioner should provide a rationale to confirm that this default assumption is appropriate, to ensure that the site is not one of the rare "special cases" for which these pathways are significant. As other jurisdictions are considering inhalation for wildlife in certain situations, the practitioner should consult provincial policy to ensure that the default policy remains in effect.

Based on previous provincial guidance with respect to inhalation and dermal exposure pathways, ERAGT (1998) noted that:

- Inhalation toxicity data are generally lacking for the majority of contaminants;

- Exposure via ingestion is assumed to be substantially larger than inhalation; and,

- Dermal exposure is limited by the presence of fur and feathers that reduce the actual dermal contact of the receptor to soil contaminants.

Although these factors suggest that inhalation and dermal exposure routes are unlikely to be applicable at the majority of sites, unique circumstances may warrant the inclusion of either pathway in the detailed ERA. Examples of unique circumstances include:

- The receptor is completely soaked in water or another carrier liquid that reduces the mitigating effect of fur or feathers (*e.g.*, waterfowl in an oil spill);

- The receptor inhabits subsurface burrows within soil contaminated by high concentrations of volatile compounds. Explicit consideration of the inhalation pathway may be warranted if the receptor involved is of special concern in the risk assessment (*e.g.*, it is a rare or endangered species); and,

- Dermal exposure (direct contact with soil and sediment) is a relevant exposure pathway for amphibians and reptiles; however, detailed guidance on how to assess dermal exposure is not available for all compounds or biota.

Policy Decision: Human Drinking Water Standards for Protection of Wildlife – MOE provides policy and rationale for use of human drinking water standards for protection of wildlife consumption; this assumption may be used to screen exposure pathways via drinking water. The practitioner should provide a rationale to confirm that this default assumption is appropriate, to ensure that the site is not one of the rare "special cases" for which ecological sensitivity exceeds human sensitivity (*e.g.*, exposure of freshwater fish to chlorine).

Content for the DERA:

- A narrative or tabular summary of each exposure pathway considered in the DERA, along with a rationale for its inclusion.

- Arguments for the exclusion of other exposure pathways must be fully documented.

Key Issues for the DERA Practitioner:

- The risk assessment is incomplete if exposure pathways were inappropriately excluded from consideration. It is preferable to conservatively include all possible exposure pathways at the problem formulation stage, even if professional judgment suggests that the exposure is likely minimal.

- Specific COPCs can increase the priority of different exposure pathways. For example, risks to carnivores via food consumption are a higher priority if the COPCs include biomagnifying compounds. Risks to aquatic life via groundwater flow are a higher priority if the COPCs are highly mobile.

3.6 Step PF-5: Identify Receptors of Potential Concern

The selection of receptors of potential concern (ROPCs) for DERA is based on site ecology and, where applicable, land use. The majority of ROPCs reflect populations of species; however, ecosystem- and community-level ROPCs can also be selected where appropriate (Suter, 1996a)[27].

Policy Decision: Population Definition – MOE is developing policy for the definition of a local population for the purposes of ecological risk assessment. This definition is linked to the specification of protection goals for wildlife.

One or more ROPCs should be selected for each receptor group present (or likely to be present) at the site. These receptor groups (Table 1) correspond to trophic levels or feeding guilds, depending on the desired level of assessment in the DERA. The underlying objective of the ROPC selection is that it must match the conceptual model for the site (Section 3.8).

3.6.1 Level of Ecological Detail

Table 1 provides generic examples of potential receptor groups. In general, a greater degree of ecological resolution in ROPC selection is appropriate when:

- **Habitat of high ecological importance is present:** For example, a bog or wetland habitat may require further subdivision of the "terrestrial plant" and "aquatic macrophyte" receptor groups listed on Table 1 into multiple subgroups (*e.g.*, floating macrophytes, emergent aquatic vegetation, carnivorous plants, rushes and grasses, shrubs). Conversely, subdivision of the terrestrial plant receptor group may be unnecessary if the site consists primarily of grasses and shrubs.

- **Rare, endangered or threatened species are present (or likely to be present):** If rare, endangered or threatened species are present (or likely to be present, based on the best-available information regarding species geographic distribution and habitat preferences), then an increased level of ecological resolution is appropriate. For example, if a rare small mammal was present, the detailed ERA should explicitly assess risks to that species' feeding guild as well as other small mammal feeding guilds (instead of simply evaluating risks to the larger small mammal receptor group). Practice in BC requires assessment of all species that are rare, endangered or threatened and known or suspected to occur at the site.

[27] An example of an ecosystem-level receptor would be "the wetland ecosystem", for instances where the measure of effect reflects an ecosystem-level process such as nutrient cycling or productivity. An example of a community-level receptor would be "the benthic invertebrate community", for instances where the measure of effect is community-level attributes such as diversity or abundance.

Key Issues for the DERA Practitioner:

- Consider all rare or endangered species known to be or likely to be present.
- Consider conducting a refined species inventory based on the approaches described in Appendix I-18 to assist in the selection of representative ROPCs.

Policy Decision: Rare, endangered or threatened species – MOE is developing policy for the assessment of these species in ERA. This definition is linked to the specification of protection goals.

3.6.2 Relationships to COPCs and Exposure Pathways

Known species sensitivities to COPCs should be considered in ROPC selection. (*e.g.*, birds are known to be sensitive to certain pesticides due to effects on egg shell thinning; some fish are known to be sensitive to selenium based on reproductive toxicity endpoints). Arguments that a single ROPC was selected as a surrogate for other ROPCs based on relative sensitivity are questionable unless supporting rationale is provided. For example, it is inappropriate to argue that earthworms should be the only soil invertebrate ROPC unless appropriate and relevant toxicity data are available, or the biology of the earthworm makes it inherently more sensitive to site-specific COPCs.

Information about the COPCs (Step PF-3) and exposure pathways (Step PF-4) under consideration should also influence selection of ROPCs. For example, if groundwater flow to aquatic life is an important fate pathway, this may indicate that hard-bottom intertidal receptors (*e.g.*, mussels; kelp) would be more appropriate than migratory fish. If there are strongly bioaccumulative/biomagnifying COPCs, they will special consideration. The duration of the potential exposure is also a relevant factor. For example, migratory birds can be included if present during the breeding season; consideration of the federal regulatory perspective on this issue is recommended if migratory waterfowl are present that trigger the *Migratory Birds Act*.

3.6.3 Land Use Considerations

For DERAs, land use should be considered in terms of its influence on habitat quality and availability; ROPC selection is therefore based on site-specific ecology (which may result in exclusion of several feeding guilds due to a lack of suitable habitat as a result of land use or development). Significant ROPCs should not be excluded from consideration based on simply on land zoning classifications.

Policy Decision: Identification of Receptor Types by Land Use – MOE provides policy on receptor groups considered relevant to each major land use classification. The current default selections are summarized in Table 2; however, the practitioner should check the policy decision summary for the most recent guidance. The defaults were developed in accordance with Ministry policy to provide greater protection of ecological resources on urban parks and agricultural lands than on industrial and commercial sites, with residential areas somewhere in between the two extremes. To reconcile this policy with the ecosystem-based approach, it is recommended that the practitioner conduct an ecological relevance check in which the land use designation is gauged against the surrounding site-specific ecology. Where no obvious conflicts are evident, the default Ministry lists in Table 2 should apply.

Policy Decision: Wildlands Receptors – MOE is developing procedures and decision rules for wildlands assessments. When complete, policy may govern the selection of representative species for this ERA type.

3.6.4 Species Inventory Methods

Assessing the ecological risks of contaminated sites to all potential receptors would be an unworkable task. Therefore, strategic selection of key receptors provides an efficient and effective way to meet the overall management goals of the site. Appendix I-18 documents a procedure for determining plant and wildlife species that may be present on a site, considering both the regional species inventories and information of habitat limitations. Depending on the scale and complexity of a DERA, this procedure may be required to document the range of potentially affected species. These procedures are helpful in identifying suitable representative species that are used as surrogates for other species in the same feeding guild.

Site visits by trained biologists are useful for making informed decisions regarding receptor selection; these visits can be used to further refine lists of species that are potentially present. Local and regional sources of information should be consulted (*e.g.*, local MOE wildlife officers, Canadian Wildlife Service, nature organizations, *etc.*) prior to conducting a detailed biological site investigation.

Key Issues for the DERA Practitioner:

- The value of conducting a detailed biological assessment should be carefully considered. The investigator must evaluate the benefits (*i.e.*, uncertainty reduction) achieved through application of detail species inventories relative to the investigation costs. For example, mink are known to be sensitive indicators of several contaminants (*e.g.*, PCBs, mercury) and would therefore serve as useful worst-case indicators of potential effects to piscivorous mammals. An investigator would need to consider the benefits and costs of conducting a local habitat survey to confirm that mink are present (or should be present) at the site, compared to relying on regional biological information.

Content for the DERA:

- A clear rationale should be provided for the selection of specific ROPCs considered to be applicable to the site and representative of (and protective of) other species in the same feeding guild.

- A clear rationale should be provided for the inclusion or exclusion of feeding guilds or trophic levels that might be present. This approach may begin with the use of land-use based defaults (from MOE policy and CSR land use designations) but should be customized to the site by considering local habitat conditions.

3.7 Step PF-6: Define Study Endpoints and Risk Hypotheses

3.7.1 Definitions

Assessment and measurement endpoints facilitate translation of management goals into a specific scope of work for the detailed ERA. The specific definitions of assessment and measurement endpoints vary among guidance documents. Commonly used definitions include:

- **Assessment Endpoint:** "The characteristic of the risk assessment that is the focus of the risk assessment" (CCME, 1996); also "an explicit expression of the actual environmental value that is protected, operationally defined by an ecological entity and its attributes" (USEPA, 1998).

- **Measurement Endpoint:** "An effect on an ecological component that can be measured and described in some quantitative fashion" (CCME, 1996); also "a measurable change in an attribute of an assessment endpoint or its surrogate in response to a stressor to which it is exposed"[28] (USEPA, 1998).

[28] USEPA (1998) uses the term "measures of effect" rather than "measurement endpoint".

For each management goal, multiple assessment endpoints may be necessary. For each assessment endpoint, multiple measurement endpoints may be necessary. Testable risk hypotheses for each measurement endpoint should be developed. Risk hypotheses "clarify and articulate the relationships that are posited through the consideration of available data, information from the scientific literature and the best professional judgment of risk assessors developing the conceptual model. This explicit process opens the risk assessment to peer review and evaluation to ensure the scientific validity of the work" (USEPA, 1998). Aquatic and terrestrial examples are provided below for illustrative purposes:

Aquatic Example:

- Management goal: Develop risk-based groundwater standards for use at a contaminated site.

- Assessment endpoint: Abundance and density of the aquatic macrophyte community along the shoreline of the site.

- Measurement endpoint: Measure the survival and growth of giant kelp (*Macrocystis pyrifera*) gametophytes exposed to groundwater concentrations representative of conditions at the point-of-discharge.

- Risk hypothesis: The survival and growth of giant kelp gametophytes exposed to groundwater concentrations are not reduced by more than 20% relative to the performance of reference samples.

- Alternate risk hypothesis: The survival of giant kelp gametophytes is not reduced below a value which previous scientific investigations determined to be the minimum survival necessary to support a viable population.

Terrestrial Example:

- Management goal: Determine if soil COPC concentrations represent an unacceptable risk to small mammals occupying the grassland portion of the site.

- Assessment endpoint: Population viability of the local deer mouse population occupying the site.

- Measurement endpoint: Compare the daily ingested COPC dose for deer mice at the site to a toxicity reference value that represents an acceptable level of effects (*e.g.*, a TRV based on an EC_X).

- Risk hypothesis: The estimated daily ingested COPC dose does not exceed the TRV.

- Alternate measurement endpoint: Compare the density, physical condition, and average weight of deer mice caught at the site relative to deer mice caught at a similar nearby grassland without elevated soil COPC concentrations (using the same level of sampling effort).

- Alternate risk hypothesis: The density, condition, and average weight of deer mice are consistent between the two sites. Note that this comparison can be made using statistical significance measures, effect size measures, or both.

Policy Decision: Populations and Protection Goals – In addition to provision of guidance concerning the definition of local populations, MOE is developing policy related to appropriate protection goals for organism populations and communities. This policy will be harmonized with other guidance. For example, the treatment of soil invertebrates will be discussed in the context of either: (1) ensuring a functioning soil ecosystem; or (2) identification as target receptors to be protected.

Risk hypotheses are not necessarily equivalent to the statistical testing of a null hypothesis; however, the risk assessor may opt to use statistical considerations depending on the particular assessment and measurement endpoints. In these instances, statistical power should be explicitly considered (*e.g.*, sample size, sample locations and study design, normal variability, appropriate alpha levels).

3.7.2 Importance in the DERA Framework

Assessment and measurement endpoints "provide direction and boundaries for the risk assessment" and "minimize miscommunication and reduce uncertainty" (USEPA, 1998). ***There must be a measurement endpoint that addresses each combination of COPC, exposure pathway and ROPC.***[29] Failure to properly define assessment and measurement endpoints was identified as a common limitation by USEPA (1993a). Other common pitfalls in selecting assessment and measurement endpoints include:

- A poorly framed assessment endpoint provides an ambiguous statement best suited to a management goal that cannot be translated into specific measurement endpoints. [Example: assessment endpoint is phrased as "protect the ecological integrity of the aquatic macrophyte community." The term "ecological integrity" is subject to interpretation.]

[29] A different measurement endpoint is not necessarily required for each combination. A food chain model for evaluating risks to small mammals would simulataneously address risks associated with soil ingestion, food ingestion and water consumption exposure pathways

- A poorly framed measurement endpoint provides an ambiguous statement that cannot be translated to a quantifiable property that can be accurately measured. [Example: measurement endpoint is phrased as "measure the productivity of the aquatic macrophytes at the site." The term "productivity" is not specified in sufficient detail, and the parameter of measurement interest is not specified.]

- A poorly selected measurement endpoint is subject to confounding factors or indirect effects that limit its utility for measuring the specific COPC and exposure pathway under investigation. [Example: measurement endpoint involves comparison of *in situ* percent coverage of aquatic macrophytes at the site relative to reference locations, but fails to consider major differences in substrate types between the locations]

Content for the DERA:

- A tabular summary of management goals, assessment endpoints, measurement endpoints and risk hypotheses.

- Clear articulation of protection goals, effect sizes of interest, and decision rules.

Key Issues for the DERA Practitioner:

- Measurement endpoints must be specified in detail so that they demonstrate that a quantifiable property exists, that the endpoint is relevant to the COPC/ROPC/exposure pathway being evaluated, and that the endpoint can be measured with adequate certainty.

- Identification of appropriate measurement endpoints crystallizes the selection of "tools" for inclusion in the DERA (Appendices I through III). The rationales should guide the selection of tools; not *vice versa*.

3.8 Step PF-7: Develop a Conceptual Model

Although this step is described near the end of the problem formulation process (which is consistent with other guidance manuals), creation of the conceptual model is an iterative and ongoing activity throughout all stages of the problem formulation. Therefore, the conceptual model is not an afterthought but rather an essential means of framing the issues relevant to a DERA.

3.8.1 Requirements of a Conceptual Model

A well-constructed conceptual model provides a summary of the site ecology. The development of the conceptual model is useful for communicating the risk assessment to others (especially laypersons unfamiliar with risk assessment terminology and assumptions). Visual depiction of the underlying relationships also facilitates a reality check on the scope of the risk assessment and the degree to which simplifying assumptions have been made in framing the risk issues. Conceptual models should include (Suter, 1996a):

- **Contamination sources:** Risk assessments may involve multiple point or non-point sources of contamination (*e.g.*, free-product zone; contaminated groundwater, soil, sediment, water, or air; effluent point sources) that should be included in the conceptual model. All on-site sources must be included; significant off-site sources should also be included. The purpose of including contamination sources in the conceptual model documents is to ensure that all relevant sources (which lead to exposure pathway and COPC selection considerations) are addressed.

- **Dominant exposure and fate pathways:** All exposure pathways considered in the DERA should be depicted in the conceptual model. Significant environmental fate pathways (*e.g.*, sediment deposition, microbial degradation, groundwater flux, sorption to organic carbon in soil) should also be indicated. Including exposure and fate pathways in the conceptual model documents that all relevant exposure pathways were addressed.

- **Relevant trophic levels or feeding guilds:** All relevant trophic levels and feeding guilds must be depicted in the conceptual model, along with significant interactions between the different trophic levels and feeding guilds (*i.e.*, the conceptual model should include a food web diagram). The inclusion of a food web diagram documents the outcome of the ROPC selection process, and also illustrates potential indirect effects that may complicate the assessment. [Example: conceptual model correctly indicates that elevated COPC concentrations in soil may impact both soil invertebrates as well as a small mammal ground insectivore].

3.8.2 Presentation Format

All conceptual models should be linked to a narrative that provides detailed rationale for the decisions made (*e.g.*, source identification, selection of COPCs, ROPCs and exposure pathways. Two different types of conceptual models are commonly applied, each with certain advantages and disadvantages:

Box Diagrams: A "flowchart" style of conceptual model. An advantage of this approach is that it facilitates a more rigorous examination of the pathways and connections among and between contaminant sources, exposure pathways, major fate processes, and biological units. Although a common symbology can be used to simplify these relationships (*e.g.*, a dotted line to indicate exposure pathways; a solid line to indicate fate processes), a highly complex box diagram conceptual model may be visually cumbersome or difficult to interpret by laypersons. An example of a box-style conceptual model is provided in Figure 5.

Pictorial: A cartoon-based conceptual model that incorporates visual representations of the pathways and receptors. This style of conceptual model is well suited to communicating contaminant source, exposure pathways, major fate processes, and feeding guilds/trophic levels to a non-technical audience. A disadvantage is that some fate processes and indirect effects cannot be represented easily in a pictorial fashion. An example of a pictorial-style conceptual model is provided in Figure 6.

Content for the DERA:

- A pictorial or box diagram conceptual model (or both) must be included.
- The conceptual model should, if applicable, discriminate among pathways that are considered to dominate exposure, versus those that are considered to be operable but with a limited influence on organism exposure.

3.9 Step PF-8: Finalize Risk Assessment Strategy

The final step of problem formulation provides an opportunity to lay out the overall strategy of the risk assessment. The strategy involves selection of specific risk assessment tools and organization of those tools into appropriate tiers. This strategy evolves throughout the problem formulation stage based on study design considerations (*e.g.*, sample size, sample locations, desired statistical power, and potential risk characterization methods). This strategy is documented in a Sampling and Analysis Plan (SAP) or Analysis Plan (in cases where sampling is not being conducted), which is submitted for review and input from interested parties. The risk assessment strategy is linked to the process for DERA shown in Figure 3 and discussed in detail in Section 2.4.

3.9.1 Choosing from the DERA "Toolbox"

Technical or financial constraints are invariably an issue. Although these constraints are a part of the reality of establishing measurement endpoints, bias or other errors described in Section 3.7.2 should be avoided. This section outlines operational guidance to translate

measurement endpoints and conceptual models developed during the problem formulation into a practical risk assessment strategy. Guidance on the application of DERA tools is provided in the exposure and effects assessment sections.

Four different categories of DERA "tools" are presented, which range from the collection of raw data to high-level interpretative methods. These tool categories are: (1) direct measurement (Appendix I); (2) modelling (Appendix II), (3) interpretative (Appendix III); and, (d) synthesis (Section 6.3). The following factors should be considered when selecting specific tools from the DERA toolbox, and in many respects, these considerations reflect the need to consider the potential uncertainty in selected approaches as part of the problem formulation (*i.e.*, proactively).

- **Specificity:** Specificity refers to the degree to which a tool is tailored to the COPC/exposure pathway/ROPC combination being investigated. Tools should be specific to the relevant exposure scenario to the extent possible.

- **Ecological Realism:** Ecological realism refers to the degree to which a tool incorporates the processes and interactions observed in the field, as opposed to requiring highly simplifying assumptions. Conservative (protective) assumptions are appropriate in simple risk assessments (with full rationale provided with uncertainty described), but a DERA should maximize ecological realism at more sophisticated levels of evaluation, subject to practical and scientific constraints.

- **Reliability:** Reliability is the ability of the tool to generate meaningful data for the purposes of the risk assessment. Reliability is improved when the tool has written protocols available, the influence of confounding factors are well-documented, and established decision criteria exist for interpretation of results. *Avant garde* and non-standard tools may be useful, but typically require an increased effort to generate scientifically defensible data.

Evaluating tools for potential use in DERA requires consideration of the above factors, which in large part determine their frequency of use. For example, aquatic toxicity tests are commonly applied in DERAs because of high reliability attributes: (1) they are based on established regulatory protocols; (2) the influence of common confounding factors is relatively well-understood for most tests; and (3) decision criteria are available (*i.e.*, provincial policy establishes a 20% reduction relative to the negative control as the permissible level of effects)[30]. Conversely, a fish or wildlife population survey has lower reliability because protocols are less specific, confounding factors are difficult to control for, and interpretation of results is not straightforward. Despite these limitations, data from a properly constructed field survey may have equal or greater value than toxicity tests once potential confounding factors and uncertainties are properly addressed. In the latter case, ecological realism of field studies compensates for reduced reliability.

[30] Similar decision criteria are also available for soil toxicity testing in the Tier I guidance manual; the ECx varies by land use.

3.9.2 Tiering/Iteration

Ideal tool(s) for a DERA are highly specific, ecologically relevant, reliable, and cost-effective; however, the reality is that the costs and level of effort tend to increase in proportion to specificity and ecological realism. DERA tools are therefore frequently implemented in a tiered or iterative manner with tools of increasing ecological realism (and cost) used only if required to achieve the desired level of uncertainty relative to site management goals. Risk assessments have been described as using a "tiered" or "iterative" approach; regardless of the term, the operational concept of starting the risk assessment with a subset of potential tools and then progressing to more complex tools (or refining existing tools) only as needed remains the same. This concept is explicitly represented in the DERA framework (Figure 3); iterative elements include the use of supplemental site investigations, and feedback loops during the implementation and evaluation stages of DERA.

To the extent possible, the problem formulation should consider the relationships of various tools to one another, along with the decision points to move through the various tiers or iterations. Flowcharts are valuable for scoping (and communicating) the potential tiers or iterations of the DERA with the client and other interested parties. Flowcharts also provide a rationale for why increasingly complex DERA tools may (or may not) be required relative to consideration of uncertainty and site management goals. Several examples of potential tiering and/or iterative arrangements are provided below for consideration. Note that decisions regarding how to organize different DERA tools are highly study- and site-specific, and therefore, these examples are provided for illustrative purposes only.

- A potential arrangement of DERA tools used to assess risks to avian and mammalian wildlife in the uplands (wildland) ecosystem is provided in Figure 7.

- A potential arrangement of DERA tools used to assess risks to aquatic receptors in the streams and rivers ecosystem is provided in Figure 8.

Key Issues for the DERA Practitioner:

- Has an appropriate DERA tool (or tools) been selected for each measurement endpoint documented in the problem formulation? Are the selected DERA tools specific to the relevant exposure pathway, appropriately ecologically relevant for the desired level of uncertainty, and adequately reliable for the objectives of the risk assessment?

- Have the relationships among different DERA tools been established to the degree needed for the problem formulation? Does the practitioner document a tiering or iterative strategy for how additional DERA tools could fit in the overall plan for this risk assessment should refinement of the risk estimates becomes necessary?

3.9.3 Preparation of a Sampling and Analysis Plan

A Sampling and Analysis Plan (SAP) (or Analysis Plan, if no sampling is required) feeds into the data gathering and analysis (exposure and effects assessments) stages of the DERA framework (Figure 3). SAPs can be combined with the problem formulation (PF/SAP) or prepared as a stand alone document. SAPs should provide information about:

- Proposed study design (*i.e.*, a rationale for number and location of samples) for each risk assessment tool, including consideration for how the data will be interpreted and used in the risk characterization.

- Data collection activities needed to implement each risk assessment tool, including sampling, analytical or test methodologies to be followed. Shipping, transport and storage requirements are usually included.

- Quality assurance/quality control measures for each data collection activity are described and data quality objectives are specified.

- Field safety and health and safety plans.

- Outline of proposed data analysis procedures and interpretation techniques, including statistical analyses.

The level of detail in the Sampling and Analysis Plan will vary depending on the complexity and nature of the risk assessment as well as the requirements of the client. For most sites, the above information will be adequate for the purposes of documenting sampling and analysis procedures.

For large and complex sites, elements of the USEPA Superfund Ecological Risk Assessment 8-Step Process may be appropriate. Specifically, Steps 4 and 5 of that guidance emphasize study design, data quality objectives, and field sampling verification. Deliverables contemplated under this process include (in addition to the PF components described above[31]):

- Field Sampling Plan (FSP) – this document provides a detailed description of the samples required to satisfy the objectives and scope of the investigation outlined in the PF. Details may include: (1) sampling type and objectives; (2) sampling locations, timing, and frequency; (3) sample designation; (4) sampling equipment and procedures; and (5) sampling handling and analysis.

- Quality Assurance Project Plan (QAPP) – this document provides a description of the organization, practical activities, and quality control procedures necessary for maintaining data quality consistent with study objectives. Quality control procedures are plans, objectives, and guidelines used to facilitate a high level of quality in the collection, analysis, and the handling of samples and associated data.

- Field Sampling Plan Verification – The primary purpose of this document is to ensure that the samples specified by (SAP) can be collected. The verification document is responsive to changes in site conditions or understanding over time. The verification may specifically address issues raised during limited supplemental site investigations (if conducted), as such often provide important information on sampling substrate, availability of target species, and suitability of reference locations.

These three supplemental deliverables are only necessary on very large and complex sites; for most sites, these elements can be addressed at an appropriate level of detail within the default SAP components listed above. Alternatively, the QAPP and FSP components can be appended to the main PF/SAP deliverable in the form of appendices, as applicable.

3.9.4 Review by Interested Parties

Review of the PF and SAP by the client, the reviewer[32] and/or other interested parties is appropriate, and corresponds to the first two checkpoints in the DERA process (Figure 3). As a practical consideration, useful input on a PF is facilitated when a SAP is included, because the SAP provides details on proposed sample locations and the specific risk assessment tools.

[31] The DERA PF document, as described in this report, includes the relevant components of what is called a "Workplan" in USEPA Superfund terminology; therefore a Workplan would not be required as a separate deliverable even for complex sites.

[32] As described in Section 2.5 there are three possible routes of review (Ministry Review, CSAP Review [Protocol 6] and External Review). Risk assessors should work with those involved to secure reviews at the appropriate times in the ERA process.

4.0 EXPOSURE ASSESSMENT

Exposure assessment is one of the main components of the analysis phase of a risk assessment (Figure 3). Exposure assessment is the process of estimating or measuring the magnitude, frequency and duration of exposure of an organism of interest (receptor) to a stressor. An exposure assessment must include the following components:

- For each COC, a description of the extent of the exposure and the patterns of exposure over space and/or time;
- Analysis of the variability of measured values, including a summary of formal quality assurance results;
- Discussion of how observed patterns relate to the risk endpoints developed in the Problem Formulation[33]; and,
- Discussion of the assumptions and uncertainties associated with measurements and/or model simulations;

This section focuses on central themes when selecting, applying and interpreting DERA tools within the exposure assessment phase of the risk assessment. Readers should also refer to specific DERA tools (described in Appendices I - III for direct measurement, modelling and interpretative tools, respectively) for additional information.

This section has the following central themes:

- Section 4.1: Selecting an Appropriate Exposure Measure;
- Section 4.2: Direct Measurement Versus Modelling; and,
- Section 4.3: Ecosystem-Specific Issues.

4.1 Selecting an Appropriate Exposure Measure

Most screening-level ERAs focus on an external exposure metric, as quantified by the total contaminant concentrations in soil, water or sediment. However, DERAs should consider how abiotic factors influence the true external and internals exposures to which organisms are potentially exposed.

External exposures consist of two separate fractions[34] depending on the temporal scale involved (Semple *et al.,* 2004); the bioavailability of contaminants in soil and sediment

[33] For example, concentration data should be discussed in the context of intercorrelations among COCs and whether the gradient of exposures is amenable to a quantitative analysis of concentration-response relationships.

[34] **Bioavailable:** The fraction of the total contaminant concentration that is immediately available for uptake by organisms. **Bioaccessible:** The fraction of the total contaminant that may be available to an organism; this fraction includes the portion of the total that is currently bioavailable, plus the portions that may become bioavailable over time.

typically decreases with aging as molecules of a COPC slowly move into locations within the environmental matrix that cannot be accessed by organisms (Alexander, 2000). Differentiation of the external dose fractions has numerous implications. In general, exposure assessment tools that measure the bioavailable fraction are preferred to those that only measure the total COPC concentration. Consideration of the degree to which the bioaccessible fraction can become bioavailable as a result of temporal or other changes is also important. For example, increased knowledge regarding sorption of organic compounds to soot carbon in sediment has implications for risk assessment methodologies such as the use of equilibrium partitioning and biota-sediment accumulation factors (Cornelissen *et al.,* 2005).

DERAs should also consider how biotic factors influence the true internal exposure[35], or dose, to which an organism is exposed. Differentiation of the bioabsorbed and bioreactive fractions also has implications for DERA. Tools that consider the bioabsorbed fraction (*e.g.*, relative bioavailability factors for soil) or bioreactive fractions (*e.g.*, physiologically based pharmacokinetic [PBPK] models; organ-specific tissue residue guidelines) provide increased ecological realism, and are an area of ongoing research.

Key Issues for the DERA Practitioner:

- Operational definition of exposure (or dose) in terms of internal/ingested versus external is adequate for most DERA applications; however, the ecological realism is enhanced when exposures are considered in terms of the bioaccessible, bioavailable, bioabsorbed and bioreactive fractions. This latter approach (with selection of appropriate risk assessment tools) is recommended if justified by the desired level of information needed to support site management.

- Selection of the appropriate exposure metric is strongly influenced by the availability of applicable and appropriate effects data. Units and types of measurements need to be consistent between the exposure and effects assessment phases.

4.2 Direct Measurement versus Modelling

Environmental fate and transport models are often utilized in the exposure assessment; there is a broad range of model types of varying complexity available. Models include strictly abiotic models of contaminant transport (*e.g.*, groundwater plume modelling) to biotic models (*e.g.*, uptake models ranging from simple bioaccumulation factors to complex food web models).

[35] **Bioabsorbed:** The fraction of the total contaminant concentration that is taken up by an organism (*i.e.*, passes across the gill, integument or gut). The bioabsorbed fraction is not necessarily the same as the ingested fraction because a significant mass of some contaminants may be excreted from the organism. **Bioreactive:** The fraction of the total contaminant concentration that is actually able to cause toxicity (*i.e.*, the bioabsorbed fraction minus the fraction that is depurated, internally sequestered, or used by the organism for its own needs).

4.2.1 Strengths and Limitations

Direct measurements and modelling have different advantages and limitations in a DERA, as follows.

Advantages for direct measurement: Direct measurement of exposure to COPCs through chemical analyses is generally considered to be more reliable and credible than simulation of COPC concentrations through modelling.

Limitations of direct measurement: Collecting sufficient exposure chemistry data may require considerable project resources, depending on the size of the area under investigation, number of COPCs and number of exposure pathways requiring sampling. Direct measurement may be precluded for safety or reasons of restricted access. Destructive sampling may be inappropriate, especially when the exposure assessment requires sampling of biological tissues (particularly for higher trophic organisms). Direct measurement generally only provides a snapshot of the potential exposure at the time of sampling (exceptions include lead in bone or arsenic in hair)—seasonal or other trends are not captured unless sampling is repeated. Finally, in situations where a future scenario is being risk assessed, direct measurement may not be possible and therefore other tools (*e.g.,* modelling) are required.

Advantages of models: Models can be used for interpolation (*i.e.*, to fill in spatial, temporal or taxonomic gaps if the measured data are insufficient) or for extrapolation (*i.e.*, once validated, models can be used to explore hypothetical scenarios regarding site management or to assess the effects of future changes in environmental conditions with time). Models can be used to gain a better understanding of the relative importance of different exposure pathways and the influence of factors that limit bioavailability (thus reducing the overall exposure). Models also facilitate a quantitative evaluation of the uncertainty in the exposure assessment that is more sophisticated than simply measuring the standard deviation or other summary statistics based on measured data.

Limitations of models: Models are limited in that the accuracy of a model prediction is unknown until the model is validated against site-specific data. A large number of data are required to parameterize some models (*e.g.*, physical properties such as water volume and flow, sediment or soil organic carbon contents, and biological properties such as lipid contents and feeding relationships for major species). Although some generic fate and exposure models are available, expertise is required to determine if the generic model is appropriate for use, or to construct a site-specific model. *Generic models should not be used unless they are relevant to the site* because structural errors in a model may result in unrealistic estimates of exposure concentration. Relatively simple models (*e.g.*, ORNL uptake models) are less sensitive to structural issues; however, assessments regarding their accuracy for a given site should still be performed.

4.2.2 Deterministic versus Probabilistic Models

Models can be either deterministic or probabilistic.

Deterministic models are advantageous because they: (a) are relatively simple to implement and interpret, and (b) require fewer data relative to probabilistic models. However, deterministic models ignore variability in parameterization by focusing on single values (*e.g.*, mean, 95% upper confidence limit). Selecting conservative estimates for these point estimates, by definition, introduces a bias such that the model is automatically overprotective for a large fraction of the model domain[36], and automatically under-protective for a smaller fraction of the model domain. Deterministic models also ignore uncertainty in the parameterization by emphasizing single values. The uncertainty analysis is therefore limited to qualitative statements about each individual parameter rather than a quantitative estimate of the total uncertainty in the model itself.

Policy Decision: Use of Point Estimates in Deriving Exposure Point Concentrations – MOE provides policy on the choice of central tendency measure where deterministic estimates of exposure are made. For example, use of 95% upper confidence limit (UCL) is specified for deriving risk quotients. MOE also acknowledges that if sample sizes are small or if spatial heterogeneity is extreme, the UCL estimation may be unrealistic and will accept the use of the highest measured concentration if it is lower than the estimated 95% UCL.

Probabilistic models are advantageous because they explicitly consider the variability and uncertainty in the distribution of each parameter; as a result, risk estimates are also provided as a distribution. As a result, risk estimates can be expressed in terms of a range or as mean with confidence intervals rather than a single value. The main disadvantages of probabilistic methods is that they are less amenable to a simple, readily interpretable, conclusion, they have parameterization requirements that are often difficult to satisfy, and they require expertise to avoid error or misinterpretation of results. Information on correlations between different parameters is also needed to avoid unrealistic amplification of the risk estimate bounds.

Distinguishing between variability and uncertainty is important in probabilistic assessment. Both variability and uncertainty produce statistical distributions of values, but those distributions are interpreted differently. For example, consider multiple water samples that are collected, analyzed for a given COPC, and the results expressed as a statistical distribution. If the differences among individual measurements reflect spatial or temporal variability in the concentration, then the distribution reflects variability. If the differences among individual measurements reflect measurement error (imprecision in

[36] Domain refers to what is being modeled: receptors, changes over time or space, *etc.*

the analytical technique), the distribution reflects uncertainty in the true value. In many cases, elements of both variability and uncertainty are present in the data, and discriminating among them can be challenging.

USEPA (1997a) states that "probabilistic analysis techniques such as Monte Carlo analysis, given adequate supporting data and credible assumptions, can be viable statistical tools for analyzing variability and uncertainty in risk assessments"; however, not every assessment requires or warrants a quantitative characterization of variability and uncertainty. Deterministic models should be implemented first to determine whether a probabilistic model would contribute to the site management objectives. USEPA (1997b) indicates that probabilistic approaches are unnecessary when deterministic risk estimates, generated using conservative methods, are clearly below levels of concern or the costs for site remediation are low.

Probabilistic approaches should be considered when:

- It is necessary to quantify the uncertainty associated with point estimates of exposure, or it is necessary to prioritize different risk estimates for site management purposes

- The costs/impacts for site remediation are high (*i.e.*, the ecological damage from site remediation is severe/irreversible and/or the financial implications of remediation merit further examination).

Key Issues for the DERA Practitioner:

- Will having a bounded confidence interval for risk have any influence on risk estimates (and therefore management decisions)?

- Are the necessary data available (*i.e.*, estimates of variability or uncertainty for all important parameters; information on correlations among parameters)?

- Probabilistic approaches should be used when it is necessary to rank risk estimates or quantify the uncertainty associated with the risk estimates.

- If a probabilistic model is used, it may be necessary to obtain input from regulatory agencies regarding an acceptable probability for a defined level of predicted adverse effects.

4.2.3 Use of Modelling in DERA

Direct measurements (Appendix I) should be the basis of the exposure assessments in the majority of DERAs, supplemented by models (Appendix II) under some circumstances (described below). Direct measurement is particularly important for measuring COPC concentrations in different exposure media when those COPC concentrations are

subsequently used as the basis for other models[37]. Models should be used to supplement direct measurement only in appropriate situations, such as: (1) the site is relatively large and models can be used to infer COPC distributions with an adequate certainty; (2) many media or species need to be sampled but only some are available (*e.g.,* avoidance of sampling higher-trophic levels or listed species); (3) temporal variability needs to be considered; or (4) there is spatial variability in habitat types on site, which may affect exposure for particular receptors..

Validation

All models should be validated where physically possible (even simple bioaccumulation factors[38]) and should be subjected to an uncertainty assessment. The concept of validation in the environmental modelling community has been the subject of substantial debate. Because models are necessarily simplifications or natural systems, modellers have acknowledged that absolute validation of models is unattainable (Schwartz, 2000). A more pragmatic approach considers model validation as a means of model evaluation as part of a broader quality assurance task, and some authors have abandoned the term "validation" entirely, in favour of "evaluation" or other terminology (Beck, 2002; CREM, 2003).

Because the degree to which any model can be "validated" is subject to philosophical debate, it is important to define model validation clearly. Several authors (Schwartz, 2000; Beck *et al.*, 1994) have suggested that judgement about a model's validity must be made based on the pre-defined purpose of the model. In this context, the goal of validation is to understand the realism of the model relative to its intended purpose (Schwartz, 2000). The following definitions of "validation" reflect this objective:

- Determination of "accuracy for providing the details required in a given application and to provide confidence in the results" (NRC, 1994);

- The process of determining "whether the simulation model is an acceptable representation of the real system, given the purpose of the simulation model" (Kleijnen, 1999);

[37] For example, it is highly uncertain to model groundwater concentrations based on measured soil COPC concentrations if those groundwater data are subsequently used to predict COPC concentrations in sea urchins. Risk assessors should avoid linking models wherever possible due to the compounding uncertainties involved.

[38] "Universal" bioaccumulation models (*i.e.*, based on analyses of data from multiple sites) such as those presented by Efroymson *et al.* (2001) typically quantify the uncertainty in the model (*e.g.*, BAF ± standard deviation) that should be considered in the sensitivity analyses. A reality check of the models against site-specific data is recommended wherever possible.

- "Substantiation that a computerized model within its domain of applicability possesses a satisfactory range of accuracy consistent with the intended application of the model" (Sargent, 1999).

- "The process used to generate information to determine whether a model and its analytical results are of a quality sufficient to serve as the basis for a decision" (CREM, 2003).

In its most narrow form, model validation has been operationally defined as a "comparison of model results with numerical data independently derived from experience or observations of the environment" (ASTM, 1984). By partitioning field data into "calibration" and "validation" data sets, the objective is often to apply a model calibrated to one data set against field observations (*e.g.*, environmental monitoring data) from an alternative data set. This cross-validation technique, although often applied, is not always possible or practical, and has been described as insufficient for environmental exposure models (Schwartz, 2000; Beck *et al.*, 1994). Beck *et al.* (1994) assert that model validation can be based on two main factors:

- The performance of the model in terms of being a valid predictive tool. This is regarded as an essentially *external* measure of validity, in the sense that it compares data derived from the model with data (or conditions) deduced from sources of knowledge independent of the specific model whose validity is to be established.

- The composition of the model, or the manner in which its constituent hypotheses are assembled, with some measure of the consensus (or disagreement) regarding the model mechanisms. This can be regarded as an essentially *internal* measure of validity; judgment about the model is being made by reference to its intrinsic mechanisms.

In summary, validation may consist of an assessment of the conceptual underpinnings of the model (*i.e.*, documentation of acceptance by scientific community, favourable evaluation in peer-reviewed literature). Alternatively, a performance-based validation can be conducted to provide a quantitative assessment of model validity. For most models applied in DERAs, a qualitative assessment of model validity is adequate. The degree of rigour required in the validation varies depending on the type of model. For example, the process of equilibrium partitioning of hydrophobic organics among lipid phases of organisms is widely acknowledged; therefore, the process of normalizing hydrophobic organics to lipid content in exposure assessment is widespread and requires minimal documentation. However, more complex models or processes, such as biphasic elimination from tissues, three-phase partitioning of organics in the water column, non-steady-state models, or other complex exposure models, require additional supporting rationales (including appropriate literature citations).

For complex or high-profile sites, where detailed mechanistic fate and bioaccumulation models are applied, several guidance documents are available to assist in the implementation of model validation, including :

- Model Validation for Predictive Exposure Assessments (Beck *et al.*, 1994);
- General Principles of Model Validation and Verification (USEPA, 1994);
- Information Quality Guidelines (USEPA, 2002d); and,
- Council for Regulatory Environmental Modelling (CREM) – Draft Guidance on the Development, Evaluation, and Application of Regulatory Environmental Models (CREM, 2003).

Such models should include consideration of how minor variations in model parameterization impact the results (*i.e.*, a sensitivity analysis), particularly for those pathways that contribute significantly to risk estimates.

Probabilistic Models

Probabilistic approaches may provide an improved consideration of uncertainty, and increasing use of probabilistic approaches for DERA is anticipated by USEPA as the science for this issue advances (Dearfield *et al.,* 2005), particularly for models with many variables.

An iterative approach (*i.e.*, initial use of deterministic models followed by an increasing degree of probabilistic parameterization, only as needed relative to risk management goals) is recommended. Probabilistic approaches should quantify the uncertainty and variability in as many parameters as possible (or at least, the parameters with the greatest impact on risk estimates). Partially probabilistic models are acceptable, provided that the interpretation acknowledges the fact that the bounds on the risk estimates do not represent the total uncertainty/variability in the model.

Key Issues for the DERA Practitioner:

- Has direct measurement been used to the extent possible?
- If models (probabilistic or deterministic) were used, were they validated against site-specific data?
- If numerical validation of a model was not feasible, was the validity of internal model processes (composition) and underlying assumptions critically assessed?
- Has a sensitivity analysis of the model been included in the discussion of uncertainty?
- If probabilistic models are used, how do risk estimates reflect site-specific uncertainty and variability?
- Have the sources of uncertainty (variability and incertitude) been properly characterized in the discussion and interpretation of model output?

4.3 Ecosystem-Specific Issues for Consideration

4.3.1 Deep Aquatic Ecosystem

Provincial guidance regarding the design and implementation of sediment quality assessments is provided in MOE (2005) and should be reviewed for applicability within the objectives of the site-specific DERA. In addition, the following issues are presented for consideration for the exposure assessment for deep aquatic DERAs:

- Selecting analytes for sediment DERAs;
- Addressing subsurface sediment;
- Sampling design for sediment quality assessments; and,
- Incorporating porewater chemistry data.

4.3.1.1 Selecting Analytes for Sediment DERAs

Chemistry samples should be subjected to a broad range of analyses including not only site-specific COPCs but also ancillary parameters. Data for multiple potential confounding factors will be required to properly interpret any subsequent effects data (*e.g.*, toxicity testing, benthic community structure), including percent organic carbon, particle size distribution, and porewater ammonia and sulphide concentrations. Acid volatile sulphide and simultaneously extractable metals (AVS:SEM) measurements

provide information about the potential bioavailability of selected divalent metals (Appendix II-7). The risk assessor should also consider the contribution of COPCs other than those attributed to the specific site (*i.e.*, regional or ubiquitous contaminants). Sediment assessments for urbanized harbours should consider the significant role that harbour-wide non-point sources play in influencing sediment quality. Nearby point-sources (*e.g.*, stormwater or combined sewer outfalls) should also be considered in the context of sediment transport patterns (*e.g.*, tributyltin should be measured if a former shipyard is nearby; pesticides should be considered if stormwater outfalls are in the vicinity).

4.3.1.2 Subsurface versus Surface Exposure Pathways

Sediment deposition and burial lead to a gradual reduction in the exposure of most COPCs (and thus reduce bioavailability) over time. The DERA must explicitly consider if exposure to subsurface conditions will occur. Examples where exposure to subsurface conditions may occur include dredging, construction (*e.g.*, installation of new pilings), general slope stability, propeller scour, major storm events or floods. Exposure pathways involving undisturbed, buried subsurface sediments may be excluded from consideration in the DERA provided that they cannot be resuspended under a reasonable likely scenario (Chapman and Anderson, 2005). Inclusion of subsurface sediment is appropriate if the risk assessor cannot reasonably exclude future resuspension scenarios.

4.3.1.3 Sampling Design for Sediment Quality Assessment

Gradient-based sampling designs are useful to assess the potential influence of other contaminant point-sources. A “near-field/far-field” approach is useful when assessing the potential influence of harbour-wide conditions. It may be necessary to tier the chemical analyses to minimize potential costs and address holding times; the investigator may consider rush analyses for the broader suite of potential COPCs on a subset of samples in order to determine analyte selection for the remaining majority of samples. Sample holding times are often an issue in this tiering approach. For some parameters, the investigator can also direct the analytical laboratory to extract samples immediately upon delivery; extracts for some organic analytes can be held longer than the original sediment sample. Limited supplemental investigations (*e.g.*, limited surface and core sampling in advance of the actual DERA sampling to scope methods and core depths that need to be sampled) may also be appropriate depending on the number and quality of data available in the problem formulation.

4.3.1.4 Sediment Porewater Chemistry

Information on COPCs concentrations in porewater may be relevant exposure data. However, *ex situ* porewater collection methods (*e.g.*, centrifugation; vacuum extraction) results in inevitable alteration of the speciation and bioavailability of the sample; *in situ* collection methods (*e.g.*, peepers; solid phase extraction) result in limited sample volumes or require specialized analytical techniques (Chapman *et al.,* 2002a). Consideration of the relative importance of the porewater exposure route within the context of the combination of ROPCs/COPCs selected in the problem formulation is necessary. Many benthic taxa are primarily exposed to surface water rather than porewater (*e.g.*, epibenthic amphipods inhabit sediment surfaces; clams extend siphons; some tube-dwelling organisms irrigate their tubes with surface water). Measurement of porewater COPCs as a surrogate for whole-sediment exposures is not recommended. However, such measurements are valuable in those instances where the porewater exposure route is of explicit interest (*e.g.*, consideration of equilibrium partitioning of compounds from sediment particles; flux of porewater out of sediment).

4.3.2 Shoreline Ecosystem

Potential issues for consideration for the exposure assessment for shoreline DERAs include:

- Implications of variable geochemical conditions; and,
- Implications of variable hydrological conditions (*e.g.*, groundwater plumes).

4.3.2.1 Geochemical Considerations

Exposure pathways in the shoreline ecosystem involve considerable environmental gradients and associated alterations in contaminant biogeochemistry. For example, geochemical changes as COPCs discharge to aquatic receiving environments from groundwater have implications in terms of using groundwater chemistry data as a measure of exposure. Changes in redox potential, for example, influence the mobility and toxicity of some metals as they transition from groundwater to seepage zones to the receiving water body. Risk assessors should consider these changes in geochemistry, and consider sampling techniques that more closely approximate conditions at the point of discharge (*e.g.*, mini piezometers in the shoreline; use of subsurface seepage samplers). Risk assessment tools that consider COPC geochemistry may not be applicable under all circumstances (*e.g.*, AVS-SEM does not apply to oxygenated sediment; estuaries have unique and variable geochemistry that impact speciation and biotic ligand models).

4.3.2.2 Hydrogeology (Groundwater Plume) Considerations

Expert advice from hydrogeologists regarding contaminant flow pathways (*i.e.*, groundwater plumes) is recommended to select appropriate sampling locations for groundwater exposure assessments. For most sites, the hydrogeological investigations conducted for site characterization purposes are sufficient; however groundwater plume models can provide useful information regarding the likely exposure concentrations at various locations (thus indicating potential sample locations) within a groundwater plume. It may be necessary to implement additional hydrogeological studies if the site has considerable temporal or spatial variability. For example, groundwater discharges from shallow aquifers in an estuarine environment tend to be relatively complex, and thus require detailed examination to justify sample placement (*e.g.*, Westbrook *et al.,* 2005). Alternatively, if detailed hydrogeological investigations and/or groundwater plumes are not available, a "picket fence" (*i.e.*, a row of samples along the shoreline) sampling approach is recommended to maximize the chances of intercepting the actual exposure pathway. Repeated sampling over time will likely be necessary to capture natural variations in groundwater flow patterns.

Policy Decision: Groundwater Plume Dilution – MOE is developing guidance on risk assessment of groundwater plumes reaching surface waters. The CSR groundwater standards assume an effective 10:1 dilution for groundwater discharged to waters supporting aquatic life. In practice, this assumption is conservative (protective) for most groundwater samples, but may not be appropriate for all situations (*e.g.*, foreshore monitoring wells). The practitioner should consult MOE policy as it is promulgated.

4.3.3 Upland Wildlands Ecosystem

Policy Decision: Wildlands Evaluation Procedure – MOE is currently developing a systematic procedure to characterize wildlands. This procedure will incorporate exposure, receptor selection, protection goals, and other factors that drive risks. The practitioner should consult Ministry guidance on this topic as it becomes available.

Policy Decision: Default Soil Ingestion Rate – MOE provides a policy statement on the default assumption of 2% soil content in wildlife diet (*i.e.*, default value in the absence of relevant species-specific information). As published soil consumption rates for wildlife species range from 0 to 30%, the practitioner should evaluate the appropriateness of the default procedure. The default ingestion assumption should be applied unless there is a compelling argument for a markedly different value.

A potential issue for consideration for the exposure assessment for upland wildlands is the appropriate level of detail in food chain (trophic-transfer) models. Food chain models are frequently used to estimate the total exposure received by wildlife ROPCs through a combination of food, water and incidental soil ingestion. Food, water and soil ingestion rates for specific ROPCs are usually based on allometric scaling equations and assumptions regarding ROPC body weight. Other model parameters needed for calculating COPC exposure includes ROPC-specific dietary preferences as well as percent moisture data for each dietary item. Food chain models need to balance the use of modeled (*e.g.*, allometric scaling formulae) and site-specific measured data for each parameter.

In general, DERA food chain models should:

- Include more dietary items than would be normally assessed in a model constructed for screening-level purposes. For example, it is appropriate to divide the soil invertebrate dietary item into foliar, soil-dwelling and litter-dwelling invertebrates because differential COPC accumulation within the food chain of the soil invertebrate community is likely (*e.g.*, Roth, 1993). Differential accumulation of COPCs by different plant species (*e.g.*, Torres and Johnson, 2001) warrants that the plant community should be subdivided into different functional groups such as grasses, forbs, shrubs and trees (and potentially varying tissue types such as leaves, shoots and berries).

- Include more direct measurements of COPC concentrations in dietary items instead of using literature-based bioaccumulation factors or uptake models;

- Include more site-specific ROPCs that more closely mirror the selected measurement endpoints rather than generic receptor types;

- Use site-specific dietary preferences that reflect the relative abundance of dietary items actually available in the site of interest. For example, including earthworms as a dietary item in the food chain model is only meaningful if the site contains both earthworms and ROPCs that consume earthworms; and,

- Use a metabolic-based model to estimate COPC dose where appropriate. Daily ingestion rates (kg food per day) are expressed in terms of daily required energy (calories per day), and the energy content of various dietary items is estimated (or measured). These models are more complex and require additional data, but provide a more realistic representation of a receptor's feeding behaviour at a given site.[39]

[39] A metabolic-based ingestion model is described in USEPA (1993b) and elsewhere in the literature. The complexity of the model can be increased to reflect temporal changes (*e.g.*, an organism's energy requirements vary depending on growth and reproductive status as well as season) and site-specific ecology (*i.e.*, feeding behaviours tend to maximize the energetic return per unit effort by focusing on abundant food items with high energy contents).

4.3.4 Rivers and Streams Ecosystem

A generic consideration for this ecosystem is the highly dynamic nature of streams and rivers, such that it is appropriate to consider the potential influence of water flow and temperature on the exposure assessment. Additionally, see guidance in Section 4.3.2.2 if groundwater discharges to rivers and streams are being assessed.

ERATG (1998) identified a number of principles for exposure assessment of aquatic organisms that remain applicable:

- Rooted aquatic plants (also called macrophytes) such as cattails, rushes, or salt grass take up contaminants from the water column (through their leaves) as well as by their roots from sediment. Non-rooted aquatic plants (*e.g.*, duckweed, water lilies) also take up contaminants from the water column through their leaves. Exposure assessment for these species may require assessment of both sediment and overlying water concentrations.

- Planktonic invertebrates are exposed to toxicants primarily by absorption from the water column, although ingestion is an additional route.

- Aquatic insects can be exposed through the water column, sediment, or ingestion of plant material or other insects.

- Clams and other shellfish are exposed through the water column and this will be the primary route for water-soluble materials. Ingestion is the main exposure pathway for materials bound to particulates or that bioconcentrate in plankton.

- Fish have 100% exposure to the water column. However, lifestyle determines exposure to the sediment. Flatfish or other bottom dwellers and burrowers are exposed to surface sediments and the interstitial water concentration of the sediment so those media should be used as exposure pathways in addition to water concentration. Higher trophic level fish (such as some of the salmonids) also are exposed by eating smaller fish with contaminants in their tissues.

4.3.5 Upland Human-Use Ecosystem

Food Chain Models

A potential issue for consideration for the exposure assessment for upland (human-use) is the appropriate level of detail in food chain (trophic transfer) models. Food chain models are frequently used for estimating COPC exposure for wildlife ROPCs. Items identified for consideration in Section 4.3.3 are equally applicable for food chain models for this ecosystem type. In addition, the following issues are applicable to food chains that model COPC exposure for the uplands (human use) ecosystem:

- Effect of human modifications to the environment that alter bioavailability must be considered (*e.g.*, type, depth, and permanence of cover materials that isolate receptors from exposure). A permanent and impermeable barrier means that COPC uptake by dietary items from those particular areas of soil is negligible. Engineered covers of clean material have varying abilities to obstruct COPC transmittal depending on depth, quality of soil relative to the underlying material, and the future species assemblage[40].

Policy Decision: Engineered Cover Depth – MOE is evaluating policy decisions concerning the depth and characteristics of cover materials required to effectively isolate receptors from underlying soil contamination. This determination is linked to policy for assumed soil rooting depths (and hence exposure) for natural, introduced, and ornamental vegetation types.

- ROPCs for food chain models need to reflect the overall habitat quality and quantity. ROPCs should be tolerant of the level of human presence at the site, and included only if they use the area for feeding.

- Habitat range factors assume that the ROPC moves equally through all parts of a contiguous habitat range. Habitat range factors are not appropriate if habitat is highly fragmented, or adjacent areas contain habitat of relatively low quality that would limit the ability of the ROPC to move and feed equally in all areas. In some instances, the site in question may contain higher (or lower) habitat quality than the surroundings. Policy decisions related to soil exposure patterns and ingestion rates are also applicable to this ecosystem.

- Exposure data (*e.g.*, soil chemistry) need to be specifically targeted to the areas included in the food chain model.

Exposure Assumptions

Exposure assumptions relevant to this ecosystem type that were identified by ERAGT (1998) and that remain applicable include:

- All plants on the site are assumed to be exposed to contaminated soil, as their roots have the potential to take up materials out of the soil. Deep-rooted plants also may contact contaminated groundwater.

[40] For example, a 0.5 meter soil cap is likely sufficient to block the accumulation of COPCs from the underlying material by ornamental grass, but may not be sufficient to block the accumulation by large rooted trees and shrubs.

- All soil invertebrates (such as earthworms, centipedes, and beetles) are considered exposed through ingestion of soil or movement of contaminant across their skin. Therefore, exposure to soil invertebrates should be assumed, unless the contamination is present only during the dormant period of the year (*e.g.*, when the ground is frozen).

- Birds are exposed only if they eat soil invertebrates or plants (leaves or seeds) on the terrestrial portion of the site, or if they eat aquatic invertebrates or fish from a contaminated water source.

- Birds may not feed on the site for their entire life. Migratory birds leave the area during the winter and so have the potential to be exposed only during the late spring, summer, and early fall; this should be considered in the exposure profile, particularly with respect to dietary items available during the resident season. The following habits of the birds using the site must be considered: (1) proportion of the year the bird resides in the area (*i.e.*, year-round resident, summer resident, or winter resident); (2) proportion of total foraging area provided by the site; (3) composition of diet (seeds, leaves, invertebrates and/or soil).

Spatial Weighting and Receptor Foraging

For mammalian and avian ROPCs, spatial considerations are important with respect to the exposure data considered in a food chain model, including:

- How to define a single "reasonable worst-case" soil concentration, if a single point estimate is used – integration of multiple individual soil values is appropriate because the ROPCs are mobile (and assumed to move equally around all portions of the site). This assumption is used to simplify the food chain model but can be replaced by a habitat-weighted food chain model if needed (*i.e.*, the ROPC preferentially spends more time in certain areas based on habitat quality considerations).

- Food chain models incorporate highly conservative soil concentrations (*i.e.*, 95% upper confidence limits of the mean; 90th percentile)[41]. This conservative policy-based assumption is the primary factor mitigating against the use of other additional uncertainty factors in a food chain model.

In the event that the spatial coverage of the available soil data is adequate and refinements to the risk predictions are required, alternatives to the use of a single reasonable worst-case soil concentration are recommended. Examples include the curve model or the construction of a spatially explicit food chain model incorporating GIS software.

[41] Existing risk assessment guidance (BCMELP, 2000) requires use of the lower of the 95% UCLM or the maximum COPC concentration. See policy decision box.

Policy Decision: Soil Exposure Assumptions – MOE is evaluating issues related to characterization of soil exposures at sites with heterogeneous contamination. Policy may be forthcoming on (1) acceptability of various central tendency measures (maximum, 90th percentile, arithmetic mean, geometric mean, *etc*. under various exposure/receptor scenarios); (2) when to apply different assumptions regarding integrated exposures (*e.g.*, random walk, preferred foraging in hotspot areas, avoidance of hotspots, area-weighted exposure estimates).

The curve model (Freshman and Menzie, 1996) is used to describe the risk to wildlife that forage over the contaminated site. The model is based on grids or areas of sampling in the site map. If the organisms are sessile, then the model reduces to the spatially distinct exposure model. However, incorporation of successively less contaminated habitat units reduces the weighted average exposure concentration. An advantage of this approach is that it moves away from single point estimates of exposure point concentrations, but retains conservatism because it effectively assumes that receptors will preferentially exploit the most contaminated habitats for feeding.

Unlike birds and mammals, soil invertebrates and plants are immobile or have low dispersability; therefore, the use of site-wide "reasonable worst-case" concentrations for the exposure or effects assessment is overly conservative. If toxicity testing is used, risks to soil invertebrates and plants should be determined for individual soil samples collected on an appropriate scale (*e.g.*, an individual soil sample is only representative of a very small area[42]). Compositing of multiple soil samples for the effects assessment is therefore problematic. Selection of specific locations to appropriately represent the range and mixture of contaminants present at the site (a decision often erroneously based on limited site characterization data) is critical. Consequently, the spatial scale for effects data for soil invertebrate and plant ROPCs tends to be greater (*i.e.*, require more sampling per unit area) than for effects data for mammalian and avian ROPCs.

[42] No specific guidance on what constitutes the appropriate area is available; however, it likely ranges from 1 to 25 square meters, depending on the heterogeneity of the contaminant and the potential range of exposure for the specific ROPCs (*e.g.*, considering root networks; movement of soil invertebrates, *etc*). Justification would need to be provided site-specifically.

5.0 EFFECTS ASSESSMENT

Effects assessment is one of the main components of the analysis phase of a risk assessment (Figure 3). Effects assessment is the process of evaluating the relationship between a dose (or degree of exposure to a substance) and the incidence, probability, and/or severity of organism response. Findings of an effects assessment often entail:

- *Exposure-response analysis* – The results of this analysis describe the relationship between magnitude or duration of a contaminant exposure and the magnitude of the response;

- *Threshold derivation* – If exposure-response analyses cannot be performed, exposure thresholds deemed protective of effects endpoints may be determined; and,

- *Causality assessment* – A determination of likelihood that the contaminants found at the site actually cause the observed effects on the measurement and assessment endpoints.

This section focuses on central themes to select, apply and interpret DERA tools within the effects assessment phase of the risk assessment. Readers should also refer as needed to details for specific DERA tools (provided in Appendices I - III for direct measurement, modelling and interpretative tools, respectively).

This section has the following central themes:

- Section 5.1: Ecologically Relevant versus Statistically Significant Effects;
- Section 5.2: Using Literature-Based versus Site-Specific Data;
- Section 5.3: Using Toxicity Testing in a DERA;
- Section 5.4: Deriving Toxicity Reference Values for Food Chain Models;
- Section 5.5: Site Observations and Field Surveys; and,
- Section 5.6: Ecosystem Specific Issues for Consideration.

5.1 Ecologically Relevant versus Statistically Significant Effects

Effects data can be interpreted within the DERA based on ecological relevance and/or statistical significance.

Policy Decision: Permissible Levels of Effects – MOE has established policy determinations related to permissible level of effect (EC_X guidance). The permissible level of effects (*i.e.*, what is considered an ecologically relevant effect from a policy point of view) for measurement endpoints involving toxicity tests is presently an EC_{20} for aquatic ROPCs at all land uses and a variable EC_X for avian, mammalian, plant and soil invertebrate ROPCs as a function of land use. Although a reality check step is essential in evaluating the reasonableness of the policy on a site-specific basis, the MOE defaults are the starting point for evaluation.

This default MOE guidance should be applied where appropriate, provided that the permissible level of effects makes sense in light of the selected measurement endpoint. For example, it is not permissible to have a 20% reduction in the survival of anadromous salmon (due to federal policy), or have a 50% reduction in the reproduction of a rare mammal species in the vicinity of a commercial operation (due to federal and provincial policy).

Policy Decision: Effect Sizes for Sediment Quality Guidelines – MOE has established policy on the development of numerical standards for sediment quality for typical and sensitive contaminated sites (Schedule 9 of CSR). The $SedQC_{SCS}$ are intended to define concentrations of COPCs below which there is a relatively low probability (*i.e.*, roughly 20%) of observing statistically significant adverse effects in standardized toxicity tests with sensitive benthic species and life stages. The $SedQC_{TCS}$ are intended to define the concentrations of COPCs below which there is a moderate probability (*i.e.*, about 50%) of observing statistically significant adverse effects in standardized toxicity tests with sensitive benthic species and life stages.

Reliance on statistical significance alone is equally problematic within the DERA framework because different lines of evidence have varying tendencies towards Type I and Type II errors. Risk assessors should consider statistical power without ignoring the actual magnitude of the observed effects. Test protocols should be consulted with respect to statistical considerations for toxicity testing; however, there is also general agreement that EC_X approaches are preferred to NOECs and LOECs (both approaches are frequently reported as per test protocols) (van der Hoeven, 1997; Chapman *et al.* 1996). A formal PF/SAP (see Section 3.9) that addresses study design and statistical considerations is necessary for most measurement endpoints that do not involve toxicity testing because regulatory protocols for site surveys (*e.g.*, study design, replication, and desired power) are not available.[43]

[43] Formal consideration of statistical power may lead to a decision that statistical significance is not a desired outcome of the study design; however, a clear statement to this effect is necessary so that the transparency of the risk assessment process is maintained.

Key issues for DERA practitioners:

- Interpretation of effects data in the DERA framework requires simultaneous consideration of statistical significance and ecological relevance. Reliance on one approach to the exclusion of the other should be avoided.

- Ecological relevance is not synonymous with policy-based decisions concerning an acceptable level of effect. A reality check on the implications of the observed effects in light of their implications for each measurement endpoint is required.

- The statistical power of each measurement endpoint is an important consideration. Although formal power analyses are not always required, the tendency for an assessment endpoint towards false positive and false negative results should be considered in the uncertainty analysis.

5.2 Using Literature-Based versus Site-Specific Data

Literature-based toxicity data are frequently used to set threshold concentrations for use in a site-specific DERA. Examples of threshold concentrations include:

- Deriving an effects-based water, sediment, or soil quality guideline;
- Deriving toxicity reference values for ROPCs;
- Deriving an effects-based tissue residue guideline; and,
- Deriving bioaccumulation factors or uptake models.

The first application (deriving an effects-based water, sediment or soil quality guideline) should only be used in conjunction with other risk assessment tools in a DERA because they involve "double-counting" of environmental concentrations (*e.g.*, surface water data are considered a measure of exposure, as well as a measure of effect when compared to the threshold concentrations). Effects-based guidelines are primarily useful for identifying areas with the highest hazard potential to target other risk assessment tools appropriately.

Several guiding principles are proposed below for appropriate use of literature-based toxicity data in the DERA process for the remaining three applications.

5.2.1 Level of Effort in Literature Search

The quality of the literature search in large part dictates the reliability of the resulting threshold concentrations. Literature searches must be comprehensive if literature-based toxicity data are used in a DERA. A description of the nature of the literature search should be provided (*e.g.*, list the search engines used or compendiums consulted; provide date ranges; provide the number of studies identified and retrieved; list the key words used). The following considerations for the design of literature searches are provided:

- Older toxicity data (*i.e.*, pre-1990) are frequently relevant, but are less represented in electronic search engines because older articles tend to be listed only by the keywords selected by the author (newer articles tend to include full abstracts in the keywords). An electronic search engine provided by a single journal publisher is inadequate.

- Keywords should be kept as broad as possible because their use is highly inconsistent. For example, a search of "zinc" and "aquatic" and "toxicity" will miss many relevant papers because "aquatic" is not consistently utilized as a keyword.

- Original papers must be retrieved wherever possible44. Risk assessors should not rely on toxicity data reported by others (especially in online compendia such as ECOTOX) because these compendia do not necessarily provide adequate context for evaluating the quality of the study design or considering confounding factors. Transcriptional errors are also potentially present. Using compilations prepared by other jurisdictions or published in the peer-reviewed literature is acceptable, provided that the risk assessor reviews the methods involved to determine their adequacy relative to the considerations listed above.

- Citation lists in relevant journal articles should be reviewed to identify other relevant papers which may not be captured through other aspects of the literature search

Key Issues for the DERA Practitioner:

- Original literature should be retrieved and reviewed wherever possible. Uncertainty associated with not reviewing the original literature must be documented.

[44] Most post-secondary institutions contain hard copy or electronic versions of the majority of relevant journals and have multi-institutional sharing agreements in place to access less-common journals. Alternatively, the NaturalResearch Council offers a fee-based documental retrieval system (http://cisti-icist.nrc-cnrc.gc.ca/docdel/) that can deliver journal articles electronically.

5.2.2 Literature Data Review

All literature data retrieved should be assessed in terms of its quality and relevance. Guidelines for reviewing toxicological data are provided in documents such as CCME (1999) and USEPA (2005a), but in general, guidelines can be divided into three categories:

Literature Exclusion Criteria: The category includes factors that would immediately result in the paper being rejected for use. Exclusion of the paper is usually due to a toxicological investigation being conducted for reasons that are inconsistent with the DERA. USEPA (2005a) list exclusion criteria for evaluating toxicological data for deriving soil standards that include: study conducted to test biological toxins, drugs, or sewage; study used *in vitro* (*e.g.*, cell lines, tissue cultures) methods rather than whole organisms; testing involved a mixture of chemicals[45]; data developed using QSAR or modeled results rather than measured data; data are not from a primary source; test duration not reported.

Study Acceptance Criteria: If a study is not immediately rejected, the study design should be evaluated further for appropriateness. USEPA (2005a) suggest the following criteria for data review in the derivation of soil standards: chemical form and concentration are reported; test medium was a natural or artificial soil; pH reported and within range of 4 – 8.5; organic content reported and less than 10%; study includes at least one control treatment with at least two additional test treatments; study reports ecologically relevant endpoints such as reproduction, population, growth or plant physiology. The objective is to match the available literature to site conditions to the extent possible using a transparent study evaluation method.

Study Quality Criteria: Studies that pass the literature exclusion and study acceptance checks may need to be reviewed in greater detail to ascertain their quality. USEPA (2005a) assigns a score of 0, 1 or 2 to each of the following nine quality criteria, and rejects any study that does not score 10 or greater. Potential factors for consideration include:

- Testing was done under conditions of high (or, for the purposes of DERA, appropriate) bioavailability;
- Experimental designs were documented and appropriate;
- Concentrations of test substances in soil were reported;
- Control responses were acceptable;
- Chronic or life-cycle tests were used;

[45] The argument regarding exclusion of data for toxicity of contaminant mixture is based on the assumption that the mixtures tested in the study are not necessarily applicable to the site in question. However, if the mixture toxicity data are in fact applicable, these data may be considered.

- Contaminant dosing procedure was reported and was appropriate;
- Dose-response relationship reported or can be established from available data;
- Statistical tests used and level of significance were described; and,
- Origin of test organisms was described.

Key Issues for the DERA Practitioner:

- Literature data should be evaluated for relevance and quality using a consistent and transparent system.

- It is not necessary to conduct a quantitative assessment of study quality; however, the DERA should indicate the rationale used to screen studies and data.

5.2.3 Derivation Methods

Derivation methods for establishing effects threshold values using laboratory-based toxicity data tend to utilize one of the following general approaches:

- **Single Toxicity Data Value -** Threshold values are based on the selection of a single data value (or the geometric mean of multiple data values; usually the lowest value[s] available), followed by application of a safety factor. Different toxicological measurements are used, depending on the application (*e.g.*, NOEC, LOEC, EC_{20}, and LC_{50}).

- **Dose-Response Curve:** The entire dose-response curve (compiled from all the screened literature values reviewed) can be compared to the COPC exposure to improve the estimate of potential risks.

- **Species Sensitivity Distributions (SSDs) -** SSDs emphasize protection at the community level rather than traditional methods that emphasize protection of individual species (Posthuma *et al.,* 2002). The basic premise of a SSD is that a "safe" concentration for the community at large can be extrapolated based on the distribution of toxicity data for the individual species that make up the community. In this respect, SSDs are fundamentally different from the common practice of dividing the lowest toxicity data point by a safety factor, and are superior because the SSD relies on the entire data distribution, not just the lowest data value. Additional discussion of SSDs is provided in Appendix III-3.

5.2.4 Dealing with Uncertainty in Literature-Based Toxicity Data

Incorporating literature-based toxicity data into the DERA process introduces considerable uncertainty if not done appropriately:

- Do a reality check of the methods, data quantity and data quality used to generate the literature-based threshold value, dose-response curve, or SSD. Applying additional uncertainty factors to compensate for poor quality or less relevant data is incorrect (*e.g.*, deriving an avian toxicity reference value based on mammalian toxicity data and an extra uncertainty factor is not recommended).

- Reduce uncertainty to the extent possible by considering how factors that influence COPC bioavailability vary between the laboratory exposures and the actual field exposures that are the subject of the DERA. The ability to address these factors is influenced by the level of effort expended on the literature review.

- Apply uncertainty factors sparingly. Default values of 10 are commonly applied for each area of uncertainty (intra-to-interspecies, acute-to-chronic, NOEC-to-LOEC, laboratory-to-field, and so on) resulting in an overall safety factor ranging from 10 to 10,000. Uncertainty factors are frequently misapplied—their original purpose was to compensate for sparse data sets, not to facilitate an extreme application of the Precautionary Principle that requires the use of an infinitely large (and thus overprotective) safety factor (Chapman *et al.,* 1998). If uncertainty factors are necessary, they should be based on the available data instead of simply assuming a default value of 10^{46}. Note that situations where multiple default uncertainty factors are necessary suggest that the available data were not entirely relevant to the objectives of the DERA.

Policy Decision: Uncertainty Factors – MOE has established policy related to derivation of Toxicity Reference Values (TRV) for wildlife and plants suggests that reported literature values be divided by appropriate uncertainty factors when extrapolating to other species. Extrapolation from one species to another should incorporate an uncertainty factor (UF) of 10-fold (policy decision) if it is not known whether or not they are likely to have similar physiological responses. Detailed analysis can be used to support the use of an alternative factor (or no factor) if specific and contaminant-specific information is available.

[46] Default safety factors are often applied initially, and replaced only if risks are found to be unacceptable. This iterative refinement can be part of the tiering strategy. Furthermore, MOE guidance uses a default value for interspecies extrapolation where information on physiological compatability is lacking.

- Use EC_X-based data instead of NOEC and LOECs wherever possible. NOECs and LOECs are driven by the selection of test concentrations, and do not necessarily reflect an acceptable level of effects (Chapman *et al.,* 1996). Concentration- response relationships or SSDs are preferred (provided that adequate data are available) over the single data point approaches. For SSDs, input from regulators regarding an acceptable percentage of species to be impacted will likely be required if this approach is adopted.

ROPCs, measurement endpoints and risk hypotheses may need to be re-examined in light of whether or not sufficient toxicity data of appropriate quality are available. For example, a DERA conducted using brown trout (*Salmo trutta*) as an ROPC may wish to generalize the ROPC as "cold water salmonid" in order to incorporate a more robust rainbow trout toxicity data set. In other cases, the risk assessor may wish to abandon the use of literature-based threshold values and instead focus project resources on direct measurement of site-specific measurement endpoints.

Key Issues for the DERA Practitioner:

- Uncertainty factors are primarily intended to compensate for sparse data sets. Comprehensive literature searches should be used to determine if the data set are truly sparse, or simply difficult to assemble.

- Default and multiple uncertainty factors should be avoided where possible.

5.3 Application of Toxicity Testing in DERA

Several "big picture" issues regarding the appropriate use of toxicity data in the DERA framework are discussed below in greater detail; however, the reader should also refer to the modules outlining the advantages and disadvantages of different types of toxicity testing provided in Appendix I.

5.3.1 Which Toxicity Test(s) Should be Selected?

The number and types of toxicity tests selected are entirely dependent on the different routes of exposure and ROPCs being evaluated, and therefore, specific guidance for or against particular toxicity tests would be inappropriate[47]. The DERA must provide a detailed rationale for the selected toxicity tests, including consideration of toxicity-modifying factors such as grain size, pH, organic carbon content, and soil moisture, as well as confounding factors such as ammonia and sulphides. Rationale and linkage of the selected toxicity tests to the ROPCs and exposure pathways is necessary, particularly when "non-standardized" toxicity tests are chosen.

[47] Mammalian and avian toxicity testing is exceptionally rare in DERA, and therefore, all further discussion regarding toxicity tests is focused on soil, sediment or water toxicity testing.

Policy Decision: Toxicity Test Preferences – MOE has investigated the feasibility and utility of identifying a subset of the available toxicity tests that should be recommended for use in DERA. This included a workshop in September 2007 to evaluate test types, data interpretation, and weight-of-evidence evaluations. The sediment workshop did not recommend selection of specific toxicity tests that should be required for aquatic DERAs (*e.g.*, use of pore water tests, recommended species, durations, or endpoints). However, the workshop identified several test types that are useful, commonly applied, and that may serve as a preferred starting point for establishing a study design. The value of multiple species testing (battery approach) was acknowledged, as was the importance of confirming that selected species are appropriate to the environmental conditions (grain size, salinity, organic carbon, *etc.*) of interest.

5.3.2 What Type(s) of Toxicity Data are Needed?

For most DERAs involving toxicity testing, a battery of toxicity tests (usually ranging from three to five tests) to reflect different trophic levels or major taxonomic groups is recommended. Several scenarios involving commonly-available toxicity tests are provided below for illustrative purposes:

- Candidate toxicity tests to evaluate groundwater quality discharging to a marine rocky shoreline: 7-d giant kelp germination and growth; 48-h bivalve larval development; 10-min echinoderm fertilization; 7-d larval fish survival and growth.

- Candidate toxicity tests to evaluate marine sediment quality: 10-d amphipod survival; 48-h bivalve larval development (on sediment elutriate); 20-d polychaete survival and growth; 28-d amphipod survival, growth and reproduction. Porewater toxicity testing may also be appropriate, depending on the goals of the investigation.

- Candidate toxicity tests to evaluate freshwater surface water quality: 7-d cladoceran survival and reproduction; 7-d larval fish survival and growth; 72-h algal growth; 7-d aquatic macrophyte growth; 7-d fish embryo development.

- Candidate toxicity tests to evaluate soil quality: 7-d seed germination, growth and root elongation; earthworm survival and growth (various durations); 28-d collembolan reproduction; 42-d enchytraeid reproduction test.

Arguments that "the most sensitive species was tested" are sometimes made to support testing a single species, However, this argument is rarely valid unless a battery of various toxicity tests was previously conducted for the site. Without such a battery of site-specific data, the argument for single species testing would require that: (1) a single COPC per exposure pathway is being evaluated (*i.e.*, no mixtures of COPCs); (2) literature-based toxicity data were available for multiple test organisms; (3) the data from the literature were derived under similar test conditions as the site in question (*e.g.*, consistent water hardness, grain size, organic carbon concentrations and so on); and (4) the most sensitive species to that particular COPC was also used for the site-specific toxicity testing. This scenario is extremely unlikely to occur.

5.3.3 What Constitutes a Chronic Toxicity Test?

The DERA should emphasize chronic toxicity data over acute toxicity data; however, the terms "acute" and "chronic" are not consistently defined or applied, in part, due the use of the terms to describe effect as well as duration. A review of definitions used by selected jurisdictions is provided below.

Environment Canada toxicity test methods: Environment Canada (1999) defines *acute* as within a short period (seconds, minutes, hours, or a few days) in relation to the life span of the test organisms, for any discernable adverse effects (lethal or sublethal). Conversely, *chronic* is defined as occurring during a relatively long period of exposure, usually a substantial proportion of the life span of the organism (such as 10% or more) and involving long term effects related to changes in metabolism, growth, reproduction, or ability to survive.

USEPA toxicity test methods: USEPA (2002a, 2002b, 2002c) has published test methods for measuring toxicity of effluents and receiving waters. Acute test methods were those designed to provide information on lethality (*e.g.*, LC_{50}) associated with 24-h to 96-h exposures. Short-term chronic test methods were developed for freshwater and marine/estuarine species with test durations ranging from <2 h to 9 days, or using embryo or larval life stages that are generally considered to be the most sensitive life stages.

Provincial water quality guideline derivation: MOE (1995) classified toxicity data as either acute or chronic toxicity. Acute referred to the results of short-term tests with toxicity endpoints that occur within 96 h of exposure (*e.g.*, less than or equal to a 96-h LC_{50}). Chronic referred to tests with lethal or sublethal endpoints that exceed 96 h of exposure. However, MOE (1995) notes that the normal longevity of the animal tested must be considered in this decision. For example, 96 hours is a relatively short time in the life cycle of most fish, whereas it may constitute most or all of the life cycle of some invertebrates or lower life forms.

USEPA water quality criteria derivation: USEPA (1985) reviews toxicity data for use in water quality criteria derivations and classifies the data as acute or chronic. Tests for daphnids, other cladocerans, or midges were deemed acute if the duration was near 48 h and the endpoint reported was either an EC_{50} for immobility or an LC_{50} for lethality. For embryos and larvae of crustaceans, molluscs and echinoderms (*i.e.*, barnacles, clams, mussels, oysters, scallops, sea urchins, lobsters, crabs, shrimp, abalone), the test was considered acute if the duration was 48 to 96 h and the endpoint was an EC_{50} for incomplete shell development plus mortality. For all other freshwater or marine animal species, and older life stages, the test was considered acute if its duration was 96 h and the endpoint was an EC_{50} based on a combination of loss of equilibrium, immobility and lethality. Tests with single-celled organisms were not considered to be acute tests, even if the test duration is ≤96 h. Chronic toxicity data are defined as coming from life-cycle tests, except that partial life-cycle tests or early life-stage tests may be used for some fish species.

The above discussion indicates that definitions of acute and chronic vary widely among jurisdictions and among organism types. Although simplifying rules for categorizing tests can be prepared, longer tests do not necessarily equate with better, more sensitive, or more reliable tests. A summary of commonly-available toxicity tests, along with rationale for its designation as acute or chronic for the purposes of DERA is provided in Table 3.

Key Issues for the DERA Practitioner:

- Tests are defined as chronic only if the test duration represents a significant fraction (*i.e.*, greater than 10%) of an organism's life cycle. These data are preferred for DERA purposes.

- Tests with duration of less than 10% of the organism's life cycle, but measuring a sensitive stage of the life cycle should be properly described as a surrogate for chronic toxicity. These data are acceptable for DERA purposes, but the uncertainty associated with their use as a surrogate for chronic exposures should be noted.

- Tests with duration of less than 10% of the organism's life cycle and not measuring a sensitive stage of the life cycle should be described as acute. Acute tests are valuable for screening purposes, but on their own, should not form the basis for concluding that effects are negligible in the DERA framework.

- The terms "lethal" and "sublethal" should also be used to describe the type of effect or endpoint being measured. Sublethal endpoints must be included, because lethality, on its own, should not form the basis for concluding that effects are negligible in the DERA framework.

5.3.4 Improving Extrapolation from the Lab to the Field

Laboratory-based toxicity testing is advantageous for DERA purposes because it facilitates a standardized, quantifiable measure of adverse effects of field-collected samples to individual ROPCs. However, toxicity testing is also limited by the fact that its application requires an inherent extrapolation from the laboratory to the field. This extrapolation represents a source of uncertainty that cannot be avoided. However, the uncertainty can be reduced or clarified through the application of additional risk assessment "tools". The underlying intent of these refinements is to conduct laboratory toxicity testing that more closely approximates site-specific environmental factors that influence COPC bioavailability and biological factors that influence potential acclimation and adaptation to COPCs. Potential techniques (not a comprehensive list) are listed below:

Use of Site Water: Toxicity testing using site water for dilution instead of laboratory water will more closely approximate site-specific factors that influence bioavailability (*e.g.*, pH, hardness; dissolved organic carbon concentration; major ion concentrations). This approach is similar to water-effect ratio testing (Jop *et al.,* 1995)

Test Organism Acclimation: Several common metal COPCs are also essential elements, and therefore organisms used in toxicity testing could have increased sensitivity to these metals if they were cultured and/or acclimated in media with low metals concentrations. For example, Muyssen and Janssen (2002a, 2002b) and Muyssen *et al.* (2002) found that culturing test animals (specifically, *Ceriodaphnia dubia* and *Daphnia magna*) in media deficient in zinc resulted in laboratory populations that were unnaturally sensitive to those same metals during toxicity tests.

Test Organism Adaptation: The ubiquitous nature of metals in the environment often leads to naturally-elevated background levels (*i.e.*, in proximity to ore bodies); organisms have also evolved adaptive mechanisms to thrive in those areas. The ability of organisms to adapt to high concentrations of metals is not currently integrated or even considered in existing regulatory frameworks (Janssen *et al.,* 2000). Adaptation of test organisms to these natural background concentrations of non-anthropogenic substances such as metals and PAHs should be considered where appropriate (*e.g.*, use field collected organisms for toxicity testing, especially for DERAs for metalliferous areas).

Key Issues for the DERA Practitioner:

- Toxicity test methods can be extensively modified, if and as appropriate, to reflect site-specific issues regarding COPC bioavailability, acclimation and adaptation.
- Practitioners should consider the benefits and limitations of applying standardized toxicity tests versus refined tests with greater potential site relevance but poorer comparability to routine protocols.

5.4 Deriving Toxicity Reference Values for Food Chain Models

The selection of toxicity reference values represents the effects assessment phase for wildlife food chain (trophic-transfer) models. TRVs can be narratively defined in several ways to reflect the desired assessment endpoint (*i.e.*, the TRV can reflect an acceptable level of risk or reflect a threshold below which adverse effects are not believed to occur). TRVs are calculated in a number of ways, including: (1) selection of the lowest reported value; (2) consideration of a range or mid-point from multiple reported values; or (3) derivation of a percentile from a statistical distribution of reported values (analogous to a species sensitivity distribution; see Interpretative Tool #3). Selecting a TRV that appropriately balances conservatism and ecological realism is an essential step for the appropriate application of food chain models in DERA (Tannenbaum *et al.,* 2003; Tannenbaum, 2005). Understanding the inherent uncertainty in TRV derivation is also necessary. Important factors for selecting and/or TRVs are described in the following sections.

Policy Decision: TRV Derivation – MOE may develop policy related to derivation procedures for wildlife TRVs based on general outcomes from ongoing expert panel debate. An expert TRV panel is currently finalizing recommendations, scheduled for release in autumn of 2008. This guidance will include discussion of allometric scaling and uncertainty factors of different types (effect to no-effect, sub-chronic to chronic, interspecies, *etc.*).

5.4.1 Level of Effort

TRVs proposed by Sample *et al.* (1996) were intended for screening-level ERA only, and therefore the risk assessor should consider if refinement of the existing TRV derivation or development of a site-specific TRV is appropriate. Limitations of the Sample *et al.* (1996) TRV derivation approach (with recommendations for refinement) are discussed further in McDonald and Wilcockson (2003). A substantial literature search effort is required to derive site-specific TRVs, except for those few instances where existing compendia of toxicity data are adequate and appropriate for deriving TRVs (*e.g.*, the ECO-SSL documents produced by USEPA).

5.4.2 Appropriate Toxicological Endpoints

Provincial risk assessment policy (Policy Decision Summary; BCMELP, 2000) currently states that acceptable toxicological endpoints include reproduction, growth, lethality, and tumour formation or other gross deformities in embryos and young, whereas subcellular responses (*e.g.*, enzyme activity, DNA breakage, haematological parameters) are not suitable for risk assessment purposes. Subcellular responses may in fact be appropriate for DERA applications at certain sites, provided that the science is adequately well-developed to demonstrate how the subcellular response has resulted in an unacceptable adverse effect. TRVs for DERA purposes should be based on chronic data where possible because the use of acute toxicity data requires multiple uncertainty factors (UF) that may result in unrealistic risk estimates.

Policy Decision: Endpoint Suitability – MOE has specified policy regarding acceptable toxicological endpoints in risk assessments (*e.g.*, exclude biochemical, behavioral, and subcellular endpoints, except where there is a site-specific rationale for their use in an ERA). Practitioners should use this policy (and any updates) as a default starting position in study designs, and apply alternative endpoints only where strongly warranted on a site-specific basis.

5.4.3 Permissible Level of Effects

For most environmental receptors such as plants or animals (*i.e.*, not humans), the goal is not to protect each individual from any toxic effect, but rather to protect enough individuals so that a viable and healthy population and community of organisms can be maintained. Therefore, a TRV is chosen (single point estimate, concentration-response curve or SSD that provides reasonable protection for a specified percentage of the organisms).

Policy Decision: Permissible Effect Size – MOE has specified policy regarding acceptable effect sizes that should be adopted by practioners. These are the default levels of response considered to be appropriate for satisfying the narrative intent of most assessment endpoints. Where an investigator believes that an alternate approach is warranted, a clear rationale must be presented.

ERAGT (1998) specified additional broad rules[48] to select the appropriate EC_X:

- Give preference to generally accepted toxicity reference values generated for that particular medium (accepted with caveats, peer reviewed, governmental, or NGO groups). For example, water quality criteria.

- Give preference to reproductive endpoints, but use lethality studies if they are the only ones available. Preferred endpoints of a toxicity test include any reproductive endpoint (*e.g.*, number of offspring, number of eggs laid, eggshell quality, fruit size and yield, presence of deformities in embryos or young), growth rates, and tumor formation or other gross deformities in embryos or young.

- If an EC_X is not reported, generate the concentration-response curve from the data provided and calculate the EC_X. As a last resort, use the lowest observed adverse effects level rather than the EC_X and do not divide by any uncertainty factors.

- If data are available from more than one study for an organism of concern, use the lowest EC_X, taking into consideration the quality of the studies and their similarity to site conditions.

- Use information for the contaminant of concern from any test (*e.g.*, bioassay, laboratory, field study) conducted with the organisms under consideration, if available.

- If the organism of concern has not been tested, use the most closely related (phylogenetically) organism. Carefully consider the phylogenetic histories of the test species compared to the organisms of concern and consider any drawbacks to extrapolating between species.

For birds and mammals, development of TRVs should ideally assign preference to studies conducted within the same feeding group (*e.g.*, passerines, raptors, galliforms). However, a competing consideration is the quantity and quality of available studies for the COPC; it is appropriate to extend the phylogenetic range of test organisms when data sets are limited. Wildlife TRVs should give preference to feeding studies (not single dose studies, or injection studies), particularly of weeks to months in duration, and during critical life stages such as reproduction and development.

For aquatic organisms (algae, invertebrates, and fish), ERAGT (1998) advocated the following considerations (practitioners should confirm this is consistent with current MOE policy):

[48] As this is an area of emerging policy, practioners should confirm that original MOE policy still applies.

- Use species from same class (*e.g.*, teleost fish). Agnatha (jawless fish) and Chondricithyes (sharks and rays) have very different biochemistries, especially with respect to PCBs and other estrogenic compounds.

- Use test species with similar routes of exposure as the organism of concern. Sediment tests conducted to estimate the toxicity of a burrowing worm should use burrowing organisms as the test organism. Filter-feeding mollusks should be the organism of choice when estimating muscle or oyster sensitivity.

- Aquatic phytoplankton are represented by single species algal toxicity tests and many kinds of test organisms are available.

- Give preference to tests conducted during a significant portion, or the most sensitive portion, of the test organism's lifespan.

Pending policy updates, and as a last resort, ERAGT (1998) recommended that LOAEL values should be used without additional uncertainty factors for all land uses. TRVs for DERA should incorporate an EC_X-based approach wherever possible, which often requires that risk assessors retrieve and reanalyze the original mammalian and avian toxicological literature used as the basis for the TRV. Graphical interpolation of the EC_{50}, EC_{20} and EC_{10} values will be necessary for the majority of studies because historical investigations are unlikely to report data for individual replicates; additionally, the available statistical power in the original study may not support calculation of the EC_{10} or EC_{20} thresholds.

If data to support an EC_X-based TRV are not available, LOAEL-based TRVs for common species are the last resort, but the effect-size that they represent should be documented (which recognizes the policy, inherent in the use of an EC_X value, that it is permissible to impact a wildlife population to a limited degree). For rare, threatened or endangered species, when EC_X values are not available, NOAEL-based TRVs (with documentation) can be used (because these species are protected at the individual organism level rather than at the population-level).

5.4.4 Uncertainty Factors

Policy Decision: Uncertainty Factors – MOE has established policy determinations related to derivation of Toxicity Reference Values (TRV) for wildlife and plants suggests that reported literature values be divided by appropriate uncertainty factors when extrapolating to other species.

McDonald and Wilcockson (2003) noted that TRV derivation often involved the use of multiple default uncertainty factors (UFs). For example, Sample *et al.* (1996) used an UF of 10 to extrapolate from subchronic (test duration less than one year) to chronic (test duration greater than one year), and a second UF of 10 to extrapolate from LOAEL-based TRVs to NOAEL-based TRVs. Tier 1 risk assessment guidance (ERAGT, 1998) suggested that a LOAEL-based TRV that was derived from a feeding study measuring reproductive, growth, lethality or deformity endpoints would not require a LOAEL to NOAEL extrapolation[49]. Risk assessment practitioners must document and provide a rationale for all uncertainty factors that are used in a risk assessment.

5.4.5 Allometric Scaling

Mammals

Allometric scaling for mammals relies on an assumption that toxicity is dependent on body weight; this assumption is based on the underlying relationship between body weight and metabolic rate. In general, smaller organisms have higher metabolic rates as a function of body weight, which influences other toxicokinetic variables linked to metabolic rate (*e.g.*, blood flow, renal clearance, respiration rate; metabolic half-life) (Bachmann *et al.,* 1996; Kirman *et al.,* 2003; Savage *et al.,* 2004). The relationship between field metabolic rate and body weight has been well-documented, and is found to approximate a value of ¾ (Nagy *et al.,* 1999; Savage *et al.,* 2004). Other authors have argued that the scaling factor is closer to ⅔ (*e.g.*, Dodds *et al.,* 2001); however, these relationships were based on basal metabolic rates and earlier assumptions that metabolism was a function of surface area instead of body weight (Savage *et al.,* 2004).

Allometric scaling (specifically, a scaling factor of ¾) was able to explain a substantial fraction of the variation in acute mammalian toxicity data sets (Goddard and Krewski, 1992; Travis and White, 1988), which is not surprising because toxicity is also dependent on toxicokinetic variables. Sample *et al.* (1996) converted TRVs expressed in terms of mg COPC/kg body weight/day) using the scaling factor of ¾. Sample and Arenal (1999) subsequently re-examined the ability of default scaling factors (*e.g.*, 1, 0.75 and 0.66) to explain variations in acute toxicity data. Sample and Arenal (1999) calculated a mean scaling factor of 0.94 ± 0.03 (range: -0.15 to 1.69) for mammalian species based on a broader variety of compounds than previously examined; however, the majority of compound-specific scaling factors were not statistically different than any of the existing default scaling factors (0.66, 0.75 or 1). Sample and Arenal (1999) concluded that default scaling factors were appropriate for drug compounds (*e.g.*, the data originally used to evaluate rodent-to-human scaling factors), but might not be applicable for all classes of

[49] ERAGT (1998) notes that preference should be given to studies with a duration of "weeks to months"; however, this is not an explicit requirement.

compounds. However, Kirman *et al.* (2003) used physiologically-based pharmacokinetic (PBPK) modelling to demonstrate that the ¾ scaling factor was applicable over a broad range of compounds other than drugs[50].

Birds

There are limited and contradictory data regarding the selection of an appropriate scaling factor for avian species. Nagy *et al.* (1999) calculated a scaling factor of 0.681 based on an analysis of field metabolic rates for 95 bird species. However, a single scaling factor for all birds may not be appropriate, given the likely differences in energy requirements for various avian species (*e.g.*, passerine versus non-passerine). An examination of acute avian toxicity data (pesticides) failed to support the use of the ¾ or ⅔ scaling factors used for mammalian toxicity data; in fact, a scaling factor greater than 1 was proposed (scaling factor of 1.2; Mineau *et al.* 1996). Sample and Arenal (1999) found a mean scaling factor of 1.19 ± 0.05 (range: 1.16 to 3.09) was determined for avian species, which was consistent with the 1.2 scaling factor proposed by Mineau *et al.* (1996). No scaling factors were used for avian TRVs derived by Sample *et al.* (1996).

Policy Decision: Allometric Scaling in TRV Derivation – An expert TRV derivation panel is currently finalizing recommendations, including recommendation on the issue of allometric scaling, scheduled for release and peer review (as a journal publication) in autumn of 2008. Existing MOE policy discourages the application of allometric scaling; this remains the default procedure pending the incorporation of the expert panel recommendations.

Tier 1 provincial risk assessment guidance (ERAGT, 1998) recommended against allometric scaling of TRVs, and instead, suggested that an uncertainty factor of 10 should be used to derive TRVs for "not so closely related" species. ERAGT (1998) suggested that uncertainty factors should not be used for closely-related species (for example, all rodents are considered to be closely-related, as are all waterfowl). Specific guidance for what constitutes closely-related species is not available. Practitioners should consult the most recent MOE policy guidance.

The implications of allometric scaling versus uncertainty factors for deriving wildlife TRVs is a topic of ongoing scientific debate[51] for which a clear consensus has not yet been developed. Both approaches address the same underlying issue: toxicity data are

[50] Compounds tested by PBPK modelling by Kirnan *et al.* (2003) included benzene, carbon tetrachloride, chloroform, ethanol, ethylene oxide, methylene chloride, methylmercury, tetrachloroethene and vinyl chloride.

[51] Risk assessment guidance for the use of allometric scaling includes documents from: Oak Ridge National Laboratory (Sample *et al.* 1996); Great Lakes Water Quality Initiative (USEPA, 1995), as well as the Total Risk Integrated Methodology model (USEPA, 2005b). Guidance documents recommending against the use of allometric scaling include the ECO-SSL approach (USEPA, 2005a).

rarely available for relevant wildlife species, and therefore, risk assessors are forced to rely on data for common laboratory species (*e.g.*, mouse, rat, quail, chicken) that are often smaller than the wildlife receptors of potential concern. Further debate to develop a policy on this issue is recommended; however, the following information is provided for consideration:

- Compound specific allometric scaling factors are available for some compounds (Sample and Arenal, 1999), and are superior to using a generic scaling factor.

- Physiological differences between different taxonomic groups are often cited as a major argument against the use of allometric scaling factors. Toxicity data should be from species with similar gastrointestinal physiology wherever possible; allometric scaling should not be used to extrapolate between distant taxonomic groups. Extrapolation between distant phylogenetic groups (mammals, birds and amphibians) should be avoided.

- Default uncertainty factors of 10 have minimal scientific basis; the problem is greatly compounded when multiple default uncertainty factors are applied. This should be considered when applying UFs in place of allometric extrapolations.

Key Issues for the DERA Practitioner:

- Applying multiple, default UFs of 10 to the TRV derivation is inappropriate for DERA purposes because the purpose of the DERA is to emphasize ecological realism. Multiple UFs may be appropriate during initial risk estimates, but often need to be augmented with a literature search.

- Risk assessors need to consider the overall uncertainty in the TRV derivation process, and either: (1) apply a single UF (preferably not a default value); (2) use a dose--response curve from literature values to characterize uncertainty; or (3) use allometric scaling (preferably, a compound-specific factor). A discussion of the uncertainty in the TRV derivation process should be provided, irrespective of the option selected.

- Where allometric scaling is contemplated, a clear rationale for its use must be presented. Otherwise, the default MOE policy should be applied.

- Scientific consensus on uncertainty factors versus allometric scaling is not available at this time. An uncertainty factor approach provides a more conservative TRV, but does not necessarily improve the certainty in the risk estimate. An allometric scaling approach may provide a less conservative TRV (for wildlife species larger than the laboratory test species), but again, does not necessarily improve the certainty of the risk estimate.

5.5 Site Observations and Field Surveys

A site observation method was part of the Tier 1 risk assessment guidance (now repealed) to "determine if plants and animals actually occur on site and whether or not these plants and animals show any obvious signs of toxicity". However, site observations regarding the presence/absence of specific plants or animals are more appropriate to the problem formulation phase of the DERA. Qualitative assessments of whether signs of toxicity are present (based on a question-based checklist) are not appropriate for DERA purposes. It would be inappropriate, for example, to attribute bare patches of ground to phytotoxicity without consideration of soil type and level of disturbances (*e.g.*, trampling, soil compaction, seasonal biology).

Properly designed field surveys for measuring the potential magnitude of effects associated with COPCs are more relevant to the objectives of the DERA. For example, measures of plant community characteristics can add substantially to the understanding of impacts to the plant community. Potential measures of effects include: biomass, dominant species, presence of sensitive species, structural stage, percent cover, and other biophysical characteristics such as soil type or moisture holding capacity. In general, field studies provide a level of ecological realism not readily attainable in laboratory studies, although multiple stressors frequently make it difficult to identify a particular stressor as the cause of observed ecological effects (USEPA, 1993a). Consequently, field surveys for DERA purposes need to consider the following:

- Study designs need to be appropriate to achieve the desired statistical power, both in terms of sample locations (*e.g.*, stratified or random sampling) as well as sample number. Note that for small sites, statistical power considerations may be less of an issue because the sampling program effectively samples all portions of the site.

- Field surveys for DERA purposes often involve comparison between impacted and reference locations. Selection of appropriate reference locations requires considerable project resources (*i.e.*, it is necessary to document that the sites are consistent in all respects with the exception of the contamination). A reference envelope approach (*i.e.*, the use of multiple reference locations to define the range of acceptable conditions) rather than basing the comparison on a single reference location is encouraged. Gradient designs are also beneficial, especially in those instances where obvious reference locations are not evident.

- The data to be collected from the site survey must reflect the assessment and measurement endpoints, risk hypotheses and decision criteria established in the problem formulation.

Policy Decision: Omission of Plant Evaluation in Winter – MOE has provided a policy statement on the utility of plant evaluation during winter months. Because many plants are senescent and because evaluations during the non-growing season are prone to misinterpretation, it is recommended that plant evaluations be conducted only in the spring or summer.

Key Issues for the DERA Practitioner:

- Field surveys need to be designed to address statistical power, the use of reference sites, and should be conducted by experienced biologists/ecologists (preferably with regional expertise).

- Highly qualitative site surveys (site observation method) are generally inappropriate for a DERA. The ability to make a credible professional judgment regarding site effects (or lack thereof) based on a qualitative survey is limited unless the site investigator is a highly specialized expert for a given community and contamination type.

The DERA toolbox (Appendix I) includes expanded discussions of field survey techniques for selected ecological units (*e.g.*, rocky intertidal communities, vascular plant communities). Additional tools may be provided over time as supplemental modules to DERA guidance.

5.6 Ecosystem-Specific Issues for Consideration

5.6.1 Deep Aquatic Ecosystem

A potential issue for consideration for the effects assessment for deep aquatic DERAs is the use of porewater toxicity testing. Porewater toxicity tests have been described as advantageous due to their increased sensitivity to chemical contaminants, overall ecological realism and their ability to avoid confounding factors (*e.g.*, grain size) common to whole-sediment toxicity tests (Carr *et al.*, 2001; Carr and Nipper, 2003). The increased sensitivity has described as follows:

- Porewater toxicity testing provides "an indication of potential sublethal effects which could otherwise not be analyzed" (Nipper *et al.,* 2002); and

- "Porewater toxicity testing may be an order of magnitude more sensitive than whole-sediment toxicity testing, which allows for further investigation for those sediments that may be causing more complex changes to the benthic community." (Carr *et al.,* 2001)

Other authors have cautioned that porewater toxicity testing has many inherent liabilities that may limit its utility for routine sediment quality investigations (*e.g.*, Chapman *et al.*, 2002a). Side-by-side comparisons of porewater and whole-sediment toxicity, although limited, indicate that toxicity is greater in porewater samples but linked primarily to ammonia rather than site-specific COPCs (Burgess *et al.*, 1993; Anderson *et al.*, 2001; McDonald, 2005). Ho *et al.* (2002) suggested that the increased influence of ammonia (relative to whole-sediment toxicity testing) may be an artifact of the test system (*i.e.*, ammonia is water soluble, and therefore more likely to result in over-exposure in a porewater sample).

Key Issues for the DERA Practitioner:

- Porewater toxicity testing for DERA should: (1) evaluate the potential role of ammonia; and (2) collect data for porewater COPC concentrations as a measure of exposure that is relevant to the measure of effect.

- Practitioners considering the application of porewater toxicity tests should carefully consider the intended application of the data. For example, studies with the objective of developing a site-specific sediment standard may not benefit from porewater tests because the exposure metric would not be in concentration units compatible with the intended standard. Furthermore, harmonizing effects data across multiple species is more difficult when tests are conducted using different media. Conversely, a study that contemplates a toxicity identification evaluation may be more amenable to porewater testing.

- The September 2007 workshop on toxicity testing may influence MOE policy on bioassay selection, including for porewater; related policy should be consulted when it becomes available.

5.6.2 Shoreline Ecosystem

Potential issues for consideration for the effects assessment for shoreline DERAs are:

- Phototoxicity; and,
- Groundwater plumes.

5.6.2.1 Phototoxicity

Phototoxicity should be considered when designing DERAs for shoreline ecosystems impacted by known phototoxic compounds such as PAHs. PAHs (and other compounds), once accumulated into biota, have the ability to absorb ultraviolet light (UV) energy. These photoactivated compounds can damage cellular membranes, resulting in biological

impairment and death. Severe PAH phototoxicity has been demonstrated to multiple taxa, primarily using laboratory-based exposure systems. However, the most relevant question regarding phototoxicity of contaminated sediment [and water] is "whether phototoxicity is of ecological relevance or merely an interesting laboratory artifact" (Boese *et al.,* 1999). Diamond and Mount (1998) noted that the risk from PAH phototoxicity depends on the "likelihood [for organisms accumulating PAH that can be photoactivated] of receiving activating solar radiation", and therefore, quantifying the UV exposure is equally as important as quantifying the PAH exposure. The traditional practice of evaluating phototoxicity using laboratory-based toxicity tests has minimal ecological realism for the following reasons (McDonald and Chapman, 2002):

- UV doses in laboratory experiments are generally maximized by the use of environmentally unrealistic light sources (*e.g.*, inappropriate photoperiods, wavelength distribution, and intensity);

- Attenuation of light in the water-column is minimized due to lower amounts of humic acid, dissolved organic carbon, and total suspended solids which absorb or block UV transmittal; and,

- Laboratory exposure systems also prevent test organisms from utilizing behavioural adaptations, such as the utilization of refugia, which minimize the internal UV dose; laboratory-cultured organisms also lack resistance and/or tolerance mechanisms that may be present in natural populations.

For these reasons, practitioners should be cautious in their application of phototoxicity studies and, if conducted, should take care to simulate natural conditions as closely as possible.

Key Issues for the DERA Practitioner:

- Laboratory-based toxicity tests are not recommended for investigating the potential effects associated with phototoxicity unless steps are taken to improve the ecological realism of issues such as UV doses, light attenuation and refugia in the toxicity testing. Laboratory-based toxicity testing without these modifications grossly overestimates effects.

- Incorporation of *in situ* toxicity testing, or additional field based risk assessment tools (*e.g.*, benthic community measurements; recolonization experiments) in the DERA is recommended.

5.6.2.2 Groundwater Plumes

Effects assessments for groundwater plumes frequently involve aquatic toxicity testing. The following modifications for toxicity tests designed for effluents and surface water should be considered if applied to groundwater samples:

- The objective of the DERA is to characterize effects at the point of discharge to the receiving environment; however, groundwater samples are normally collected from upland sites located at a distance from the receiving environment. The dilution series for the toxicity test should reflect the range of likely groundwater concentrations at the point of discharge, as determined by site-specific groundwater modelling (rather than simply assuming that a 10-fold attenuation exists; see policy statement below).

- Regardless of the dilution series selected, the test should always include the maximum possible concentration. For freshwater sites, the maximum test concentration will be 100% groundwater. For marine sites, the maximum test concentration will vary from approximately 70% – 100%, depending on the amount of hypersaline brine needed to adjust the groundwater salinity to the surface water salinity.

- Toxicity testing requires that samples be well-oxygenated and have pH values that are capable of supporting aquatic life (typically pH 6.5 – 8.5). Sample manipulations to achieve the necessary test conditions may also alter contaminant bioavailability, and thus, represent a source of uncertainty in the toxicity data.

Site surveys of the groundwater discharge areas may also provide useful information regarding potential effects (*e.g.*, measure the diversity and abundance of organisms in the discharge pathway). Note that soft-bottom benthic community surveys are not appropriate as a measure of effect for groundwater discharges because the benthic community reflects exposure to sediment-associated contaminants, not groundwater.

Policy Decision: Groundwater Plume Dilution – MOE is in the process of developing guidance on risk assessment of groundwater plumes reaching receiving environments. The CSR groundwater standards assume an effective 10:1 dilution for groundwater discharged to aquatic life. In practice, this assumption is conservative (protective) for most groundwater samples, but may not be appropriate for all situations (*e.g.*, foreshore monitoring wells). The practitioner should consult MOE policy as it is promulgated.

5.6.3 Upland Wildlands Ecosystem

Potential issues for effects assessments in uplands wildlands DERAs are:

- Consideration of the potential for indirect effects.

- Levels of acceptable risk in soil invertebrate or vegetative community assessments – Studies on this topic are being conducted in association with upstream oil and gas sites, but are not yet ready for incorporation into provincial guidance or policy.

- Approach and protection goals for wildlands ecosystems – the MOE is currently conducting studies on this topic, but they are not yet ready for incorporation into provincial guidance or policy.

5.6.3.1 Indirect Effects

Indirect effects occur when a toxicant-related effect on one species causes a resulting effect on a second species due to altered ecological interactions such as predation, competition or resource availability. Examples of indirect effects include:

- Indirect effects on a passerine bird population may occur as a result of changes in habitat availability (*e.g.*, a soil COPC is phytotoxic, which in turn reduces forest cover and thereby changes the habitat).

- Indirect effects on an insectivorous small mammal population may occur because soil invertebrates may avoid areas with elevated soil COPC concentrations, thereby altering food availability.

- Indirect effects on fish populations occur as a result of a change in the zooplankton community (thus reducing food availability) associated with elevated water COPC concentrations.

Preston (2002) stated that single-species toxicity testing does not necessarily capture the complexity of the potential effect on an ecosystem-level effect. Numerous risk assessment tools are intended to compensate (at least partially) for this limitation, including: (1) the use of a battery of toxicity tests; (2) mesocosm toxicity testing; (3) species sensitivity distributions; and (4) integrating the results of field surveys with toxicity data using a weight-of-evidence approach. None of these tools can fully account for the myriad of interactions that will occur in a natural ecosystem. Nevertheless, increased consideration of the complexity of a site's ecology and the multitude of factors that drive an ecosystem's response to a chemical stress is recommended to address the implications of indirect effects. To date, minimal guidance on how to incorporate indirect effects into a risk assessment framework is available in the literature.

5.6.4 Rivers and Streams Ecosystem

No ecosystem-specific implications for the effects assessment are currently identified. Consult guidance in Section 5.6.2 if groundwater discharges to rivers and streams are being assessed.

5.6.5 Upland Human-Use Ecosystem

No ecosystem-specific implications for the effects assessment are currently identified. Section 5.6.3 provides discussion of topics of potential relevance to this ecosystem type.

6.0 RISK CHARACTERIZATION

Risk characterization (shown in Figure 3) is the process of estimating the magnitude (and where possible, the probability) of adverse ecological impacts based on the information obtained from the exposure and effects assessments.

Risk characterization provides the discussion of the "strengths, limitations and uncertainties arising from the data and models used to provide conclusions" (CCME, 1996) and accomplishes the following objectives:

- Risk characterization demonstrates how the results from multiple tools are integrated into a conclusion for each individual line of evidence, and how the conclusions from multiple lines of evidence are integrated into an overall conclusion regarding ecological risks. This integration is necessary to maintain transparency of the risk assessment process.

- Risk characterization requires that conclusions are presented in a clear and unambiguous manner (*i.e.*, conclusions are stated in plain-language). The tendency for technical reports to obscure conclusions using jargon should be replaced by clear statements of what was estimated (and how). Emphasis on clarity in the risk characterization is necessary so that the DERA can be used by site managers in their decision making process.

- Risk characterization also requires that the uncertainty in the conclusions be discussed. Again, the goal is to provide site managers with information needed for site planning purposes.

This section focuses on the following three interpretive tools that are typically used in the risk characterization phase:

- Hazard quotients;
- Multivariate statistical analyses; and,
- Weight-of-evidence approaches.

Detailed descriptions of these tools are presented in the modules contained in Appendix III. Additionally, guidance is provided regarding the application of best professional judgment, the appropriate terminology used to narratively describe risk estimates, and the role of uncertainty analysis in the DERA process.

6.1 Quotient Methods

Hazard quotients (HQs) are widely used in DERA due the prevalence of literature-based toxicity data and food chain models. The hazard quotient for each combination of contaminant and receptor (plant or animal) of concern is calculated by dividing the estimated environmental concentration (EEC) by a single-point toxicity reference value (TRV). However, HQs measure hazard (as the name implies) rather than the classical definition of "risk" (*i.e.*, they do not contain information about the probability that an adverse effect will occur). More information on HQs is provided Appendix III-1.

HQs are also subject to the following considerations:

- Quotient methods are only as reliable as the values in the numerator and denominator (with associated uncertainty).

- Quotient methods assume that both the numerator and denominator exist in all locations and all occasions when, in fact, environmental concentrations are variable. The use of point estimates for the numerator and denominator mask the underlying uncertainty and variability in the data.

- A single HQ can be calculated for the entire site by using the 95% upper confidence limit (UCL) of the mean for all of the measured values (Gilbert, 1987) for each medium or the maximum measured concentration, whichever is lowest. This will result in a conservative estimate of risk, particularly for a small site with relatively few environmental sampling points or a site with one or more small areas of high contamination.

- Spatially distinct risk quotients can also be calculated, and the probability of exceeding a hazard quotient of a given magnitude can be computed. This technique is generally applied when the single HQ method (screening assessment) yields a value above 1.0 and where the single HQ method is considered to be over-conservative.

- HQs are not proportional to the magnitude of "risk". Although a very large HQ demonstrates a greater "risk" than a HQ slightly greater than 1, it is not true that minor changes in the HQ provide a meaningful differentiation (Ritter *et al.,* 2002).

- The number of significant figures in the HQ should reflect the lowest number of significant figures in the numerator or the denominator. Inclusion of excessive decimals implies a level of certainty that is not actually present. Most HQs can be rounded to one or two significant figures.

Policy Decision: Prohibition Against Adding Risk Quotients – MOE policy for ERA (BCMELP, 2000) is that HQs should not be added across species for substances due to number of assumptions incorporated in both the exposure and toxicity assessment values. Furthermore, for single species exposed to multiple substances at a site, HQs should only be considered additive under special circumstances, in which a clear mechanism for additive toxicity is hypothesized. Currently such models exist for dioxin-like compounds (TEQ method) and for select organic contaminants using the polar narcosis model (specifically multiple individual PAHs).

Key Issues for the DERA Practitioner:

- Hazard quotients do not provide a measure of risk. Hazard quotient approaches, if used as a line of evidence in a DERA, should be supplemented by other methods that provide more information about the magnitude and/or probability of adverse effects.
- Hazard quotients are generally applied as a screening tool. HQs greater than 1.0 generally indicate that additional assessment is warranted, using tools that are site-specific and relate more directly to the assessment endpoints.
- Hazard quotient assessments are iterative and generally begin with conservative assumptions that are replaced with more site-relevant refinements to exposure and effects metrics in subsequent stages.
- Hazard quotients should not be interpreted to be linearly related to the potential for harm. At best, HQs provide only order of magnitude indications of potential for adverse effects.

6.2 Multivariate Statistical Analyses

Multivariate statistical analysis refers to any of various statistical methods for analyzing more than two variables simultaneously. Assessing effects at a community or ecosystem levels usually involves measuring a large number of abiotic and biotic variables. Assessing each variable individually or with many pairwise bivariate analyses can be cumbersome, difficult to interpret, and cannot detect patterns that emerge from the interactions of variables. Multivariate techniques can be used to draw overall patterns from a large set of variables. Multivariate techniques can also be invaluable in displaying these patterns and communicating them to a non-technical audience.

There are three broad types of applications for multivariate techniques: ordination (data reduction), classification (clustering and discrimination), and canonical ordination (investigating relationships between sets of variables).[52] Appendix III-6 provides an overview of the common multivariate statistical approaches and identifies potential pitfalls. See Sparks *et al.* (1999) for more information on specific techniques as they have been applied to risk assessment; additionally, a statistician with experience in biological or ecological investigations should be consulted as needed[53]. Note that the application of specific statistical techniques is subject to ongoing research, and therefore, the techniques listed below are meant only to illustrate the range of likely approaches. Selection of different statistical techniques will be study- and data- specific.

6.3 Weight-of-Evidence (WOE) Assessment

Key Issues for the DERA Practitioner:

- A formal WOE evaluation provides a framework for rigorous consideration of the strengths and weaknesses of various measurements, and of the nature of uncertainty associated with each of them.
- Practitioners should evaluate the overall weighting assigned to each measurement endpoint, using systematic procedures to gauge endpoint attributes (*e.g.*, strength of linkage to assessment endpoints, data quality, and study design and execution).
- The magnitude of response observed in each measurement endpoint should be evaluated using rules that are as consistent as possible.
- The concurrence or divergence among outcomes of multiple measurement endpoints should be carefully evaluated.
- WOE determinations may be quantitative or qualitative, but should always be transparent.
- Professional judgement may be exercised, but a transparent analysis should be applied to elucidate the influence of professional judgement on the results.

In most risk assessments, multiple lines of evidence for each assessment endpoint are evaluated, including where applicable or available:

[52] Bayesian approaches provide alternative methods for statistical analyses that explicitly incorporate uncertainty. Specific guidance for application of Bayesian approaches in a DERA is not available at this time. Risk assessors should consult a statistician for further information.

[53] For those risk practitioners without access to a statistician, Simon Fraser University offers statistical consulting services (www.stat.sfu.ca). Other college/university statistics departments, or other consulting firms may also offer similar services.

- Field surveys/studies – these tools range from qualitative site observations to highly detailed and specific wildlife studies.

- Toxicity tests – these tools range from routine laboratory-based tests to highly customized procedures such as *in situ* testing and toxicity identification evaluation.

- Comparisons of effects in the literature to a site-specific exposure model – these tools range from single hazard quotients based on UCLs and default TRVs, to probabilistic models of wildlife exposures compared against site-specific TRVs (*e.g.*, experimental feeding study).

Whatever the tools applied, the goal of the WOE is to fairly and objectively evaluate each tool in terms of its strengths, weaknesses, uncertainties, and contribution to an overall risk statement. In risk characterization, risk assessors must provide their opinion of the significance of results generated with regard to confidence, uncertainty and impact significance. The quality of a DERA is in large part determined by the clarity and consistency used by the practitioner in reaching these overall conclusions.

All pollutants are contaminants, but not all contaminants are pollutants[54] because substances introduced into the environment may be more or less bioavailable to organisms depending on their chemical form, modifying factors in the environment, the environmental compartment they occupy, and the reactions (behavioural and physiological) of exposed biota (Chapman *et al.,* 2003). Accordingly, determining when contamination has resulted in pollution requires not only chemical but also biological measurements (*i.e.*, both exposure and effects assessment).

Because there are no perfect tools for determining pollution (*e.g.*, we cannot measure all possible contaminants, run all possible tests, or determine the health of all organisms), risk assessments require that the results from multiple tools be integrated into a single conclusion regarding the likelihood and magnitude of ecological risks[55]. This integration is normally accomplished using a WOE assessment framework (Chapman *et al.,* 2002b) that evaluates possible ecological risks based on appropriate, multiple lines of evidence (LOE). Although concurrent measurement and simultaneous consideration of multiple LOE are common, WOE-type approaches using a more linear approach are also available (*e.g.*, sequential analysis of lines of evidence [SALES]; Hull and Swanson, 2006). The manner to which the WOE incorporates different LOE (*i.e.*, in sequence or simultaneously) is dependent on the study design.

[54] Contamination refers to substances present where they would not normally occur, or at concentrations above natural background. Pollution refers to contamination that causes adverse biological effects in the natural environment.

[55] See Section 6.5 for a discussion of narrative descriptors of risk.

WOE can be applied to any DERA for any environmental media, although numerous WOE assessments to date address sediment quality issues. This is largely due to the evolution of the Sediment Quality Triad (*e.g.*, Long and Chapman, 1985; Chapman *et al.,* 1997; Chapman 1990, 1996). Examples of WOE frameworks are provided for sediment (and other media) in Chapman *et al.* (2002b) and Chapman and McDonald (2005).

Examples of other WOE frameworks for non-sediment related assessments include:

- Johnston *et al.* (2002): WOE for an estuarine site which included LOE focused on pelagic fish, epibenthos, benthos, eelgrass, salt marshes and waterfowl.

- Sample and Suter (1999): WOE for piscivorous wildlife in a large river-reservoir system based on a literature-based food chain model, biomonitoring and field observations.

- Lowell *et al.* (2000): WOE for aquatic insects in large river systems based on a combination of field surveys, streamside artificial mesocosms, stable isotope analyses and bioindicators.

- Menzie *et al.* (1996): broad guidance for the construction of WOE assessments for DERAs.

6.3.1 Guiding Principles

Guiding principles for all WOE assessments (irrespective of the environmental media under investigation) include:

- Lines of evidence incorporated in the WOE should include both: (1) laboratory studies with individual organisms[56] and, (2) field measurements of resident populations (Chapman and Hollert, 2006). These different LOE provide complementary information that strengthens the ability of the WOE to make proper conclusions. Laboratory-based LOE provide the ability to measure contaminant-related effects under standardized conditions which reduce the influence of other non-contaminant related stressors, while field-based LOE capture information about adverse effects under realistic exposure conditions. The number and complexity of different risk assessment tools within each broad category of LOE can be (and should be) tiered.

[56] Inclusion of both laboratory and field studies is not possible in every instance (*e.g.*, a WOE assessment of mammalian and avian ROPCs would not likely involve laboratory-based toxicity studies).

- If the WOE indicates that adverse effects are present based on consideration of the laboratory- or field-based LOE, the risk assessor should consider implementing additional LOE to evaluate causation (*e.g.*, *in situ* measurements of toxicity to assess differences between the laboratory and the field potentially related to tolerant field populations; measurements of contaminant body residues in organisms related to effects thresholds; chemical manipulations combined with laboratory toxicity measurements [TIE]). These causality investigations are often useful for resolving potential disagreements between different LOE. Criteria for evaluating the causality in other LOE can also be established (for example, see Lowell *et al.* 2000)[57].

- It is necessary to establish an *a priori* framework (to the extent possible) for integrating different LOE. The *a priori* framework should be agreed to by appropriate interested parties and should include a description of how the magnitude of response observed in each LOE and the concurrence among multiple LOE will be evaluated in terms of arriving at a risk estimate.[58] The use of an *a priori* framework means the data are fit to an agreed-upon decision-making framework, rather than the framework being fit to the data. This approach also matches the basic scientific paradigm of developing testable hypotheses prior to experimentation.

- WOE is not a static methodology. Its greatest strength is its flexibility in terms of the inclusion of different LOE to reflect the latest scientific knowledge and practices. The best available science should be used in applying any WOE assessment. Design and implementation of a WOE assessment reflects the experience of the scientists involved. Thus, WOE assessments also require suitable state-of-the-art expertise in the various disciplines comprising a particular assessment.

6.3.2 How to Weigh Different Lines of Evidence

WOE assessments need to be applied within the context of common sense; they should not be applied inflexibly. Critical to the WOE process are three factors: the weight assigned to each LOE; the magnitude of response observed in each LOE; and concurrence among multiple LOE (Menzie *et al.*, 1996). The weight assigned to different LOE is determined as follows (Chapman and Anderson, 2005):

[57] Lowell *et al.* (2000) established *a priori* causal criteria for evaluating different LOE in a WOE for northern rivers. Criteria included: spatial and temporal correlation; plausible explanation linking stressor and effect; experimental verification of stressor cause-effect relationship under controlled conditions; strength of the correlation, specificity of the effect to the COPC, evidence of COPC exposure in the body of the ROPC; consistency of association across other studies within the region and in analogous studies in other regions. Other examples of causality criteria are summarized in Lowell *et al.* (2000) and elsewhere in the literature.

[58] The complexity of the *a priori* framework is project dependent. For example, a terrestrial ERA might include upwards of 20 or more measurement endpoints (*e.g.*, toxicity testing on multiple species in addition to different plant and soil invertebrate community metrics), which would make it difficult to establish the precise weighting of each different endpoint. However, it should still be possible to establish *a priori* what would constitute an unacceptable effect for each measurement endpoint, and to lay out general guidelines for how different types of data would be integrated.

- Chemistry data should not be used alone for decision-making except for "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; Wenning *et al.,* 2005).

- Greater weight must be applied to biological (effects) data than to exposure data.

- Within the effects data, LOE (*e.g.*, laboratory toxicity tests, models) that contradict the results of properly conducted field surveys with appropriate power to detect changes "are clearly incorrect" (Suter, 1996b) to the extent that those toxicity or model LOE are not indicative of adverse biological effects in the field. Conversely, data from field studies without appropriate statistical power should not be ignored, but rather, weighed appropriately in the WOE (along with toxicity, model or other LOE) depending on its strengths and limitations.

6.3.3 Numerical versus Non-Numerical Ratings in WOE

The symbology of the WOE can vary from assessment to assessment. Numerical ratings for each measurement endpoint (or LOE) were proposed by Menzie *et al.* (1996), based on a set of eleven attributes scored between 1 and 5 based on *a priori* narrative criteria[59] (similar to causal criteria established by Lowell *et al.* [2000]). The relative weight of each attribute was established on a scale between 0.0 and 1.0 based on a survey of 10 experienced risk assessors. The WOE was based on the sum of the (quality score x relative weight) scores. WOE frameworks proposed by Chapman and coauthors (*e.g.*, Chapman *et al.* [2002b]) used non-numerical rating systems (*e.g.*, "○", "⊙", "●"). The specific symbols used in the WOE are not relevant.

Numerical ratings should only be used if the risk assessor can make meaningful differentiations between varying magnitudes of effect within a LOE as well as the relative weight between different LOE. Relative weighting systems such as those used by Menzie *et al.* (1996) are suitable; however, Menzie *et al.* (1996) emphasized that weighting systems should reflect collective professional judgment to minimize the influence of bias. When these conditions cannot be satisfied, numerical ratings are not recommended because they: (1) likely reflect arbitrary and subjective differentiations; and (2) imply a level of precision that is not actually present (*e.g.*, a score of 5 is worse than a score of 6 when in fact the uncertainty in the LOE means that both scores are functionally equivalent). Non-numerical rating systems are recommended under these circumstances.

[59] Attributes considered were: strength of association between measurement endpoint and assessment endpoint; site-specificity; stressor specificity; quality of data; availability of an objective measure for judging harm; sensitivity of the measurement endpoint to detect change; spatial representativeness; temporal representativeness; ability for the endpoint to be expressed quantitatively; correlation of stressor to response; and use of a standard method.

6.3.4 Applying WOE in DERA

In summary, WOE assessments must be objective, transparent, scientifically rigorous, and appropriate to the level of certainty needed for site management purposes. No specific framework is proposed (because the framework should be study- and site-specific), provided that it meets these criteria. WOE assessments provide the best means for risk characterization of environmental stressors (not restricted to just chemical contamination). They can be designed to address site-specific considerations as well as both localized and regional risks. Because WOE findings can be made readily understandable to interested parties, they provide not only useful information for decision-making, but also useful risk communication tool.

6.4 Incorporating Professional Judgment

Key Issues for the DERA Practitioner:

- Professional judgement may be exercised, but a transparent analysis should be applied to elucidate the influence of professional judgement on the results.
- The ability to replace professional judgement with more objective approaches should be considered in the evaluation of study uncertainties.
- The practitioner should avoid conjecture and supposition, and instead focus on a discussion of required assumptions and their potential influence on study results.

Professional judgment plays a major role in the DERA framework. Selection of COPCs, ROPCs and exposure pathways requires a degree of professional judgement. The construction of measurement endpoints and risk hypotheses (as well as selecting the risk assessment tools to test the hypotheses) also requires professional judgment based on education and experience, as does the interpretation of effects data (and if applicable, integration within a WOE framework). Risk hypotheses are accepted when the evidence in favour of the hypotheses is considered sufficient, and rejected when the evidence is not in favour of the hypotheses (or deemed insufficient). Some data will have decision points established by regulatory policy (*i.e.*, a 20% reduction or greater in aquatic toxicity endpoints considered unacceptable), whereas others can utilize statistically significant differences[60]. However, some study endpoints are not amenable to fixed decision rules and therefore require the risk assessor to evaluate whether they are evidence of an adverse effect or not.

[60] Note that decisions regarding statistical significance that rely on $p > 0.05$ also involve professional judgment, albeit to a lesser degree (because the Type I error rate is somewhat arbitrary).

Professional judgment is essential to risk assessment because the goal of the risk assessment is not limited to identifying those substances that are *scientifically proven* to be harmful, but also those substances for which there is scientific evidence that they *may* be harmful (Wandall, 2004). Wandall (2004) argues that proper application of professional judgment in risk assessment required that (1) risk assessors are aware of what underlying values[61] they are relying on, (2) the values are justifiable, and (3) transparency is ensured. This requirement for transparency is the foundation of properly applied professional judgment, and translates into the following guiding principles for applying judgment in the DERA framework:

- Risk assessors should determine if alternate or additional tools would provide data less reliant on professional judgment. Arguments against implementing these additional tools based on their cost or time required are not, on their own, sufficient to justify using professional judgment alone when alternate methods are available.

- All assumptions and decisions must be supporting with a rationale, especially for those instances where education and training (*i.e.*, no citations are available) were used as the basis for the professional judgment.

- Declarative and unqualified conclusions such as "the risk assessment proved that there are no adverse effects" should be avoided. Instead conclusions should reflect where professional judgment was applied in the evaluation (*e.g.,* "The risk assessment, based on our professional judgment of ABC data, and subject to assumptions XYZ, found no evidence of adverse effects").

[61] Values refer to the attributes of "doing good science" rather than consideration of political or socioeconomic factors. Examples of values that scientists apply when creating and testing hypothesis include: ability of the hypothesis to explain the available data; simplicity of the hypothesis itself, fidelity of the hypothesis with other established facts; and whether a conservative burden of proof has been met (Wandall, 2004).

6.5 Narrative Descriptors of Risk

Ideally, risk estimates should be a quantitative statement which includes a probability (*e.g.*, "there is a 20% chance of 50% mortality"). However, a minority of tools support the estimation of probability, and therefore, conclusions are usually presented as a qualitative statement (*e.g.*, "there is a high chance of mortality occurring") (USEPA, 1992). Risk assessors must provide an opinion regarding their results generated with respect to confidence, uncertainty and significance of impacts; a statement about probability is ideal but not required. Hazard quotients are frequently used as a line of evidence in risk assessments despite the fact that they are not truly an estimate of risk (see Section 6.1; Appendix III-1). The emphasis on narrative descriptors of risk can lead to a situation in which risk characterization language varies widely in application between different risk assessors.

The following operational guidance for the use of different narrative descriptors is provided:

- **Negligible risks:** Implies that adverse effects, based on the totality of available data, are very unlikely to be present, and that the risk assessor has high confidence that adverse effects will not be present in the future. This term should only be used in situations where multiple lines of evidence demonstrate a lack of adverse effects, and where each line of evidence (or the overall risk estimate) has relatively low uncertainty. Risk management or remediation is not necessary. Hazard quotients, if used as a line of evidence, tend to be less than one, although screening hazard quotients greater than one can still be associated with negligible risk in a WOE.

- **Low risks:** Implies that adverse effects are likely not present based on the totality of data available. Low risk differs from the term negligible risk in that the former designation is more appropriate for situations where the conclusion is based on the balance of probabilities. Adverse effects are unlikely to be present, although some data may indicate limited adverse effects, or the uncertainty is such that one cannot definitively exclude potential adverse effects in the future. Risk management or remediation is not necessary.

- **Moderate (or intermediate) risks:** Implies that some degree of adverse effects are likely, based on the totality of data available. Risk estimates suggest that risk management or remediation is necessary, unless further refinement of the risk estimate is conducted.

- **High (or severe) risks:** Implies that adverse effects are likely (and of relatively high magnitude) based on the totality of data. Risk estimates suggest that risk management or remediation is necessary, and that this conclusion is unlikely to change even if further refinement of the risk estimate is conducted.

6.6 Uncertainty Assessment

Risk assessment involves estimation, extrapolation, and the use of models and assumptions that generate uncertainty in risk estimates. The following sources of uncertainty are identified within the context of providing operational guidance regarding uncertainty analyses for detailed ERAs.

- **Parameter uncertainty:** refers to missing or ambiguous data resulting from inadequate sampling, analytical errors, or lack of site-specific data.

- **Variability:** refers to the inherent variability or heterogeneity of a parameter or attribute. The variability in a data set can be characterized and evaluated, but it cannot be reduced. This type of uncertainty is also called stochasticity.

- **Structural (or model) uncertainty:** refers to gaps in understanding or scientific theory on which models are based[62]. This type of uncertainty is also called *incertitude*. Models can be improved as they incorporate more precise and site-specific physical, chemical, and ecological information, but can never reduce uncertainty to zero. For this reason, Box and Draper (1987) coined the common refrain that "essentially, all models are wrong, but some are useful." Inappropriate application of generic models results in increased structural uncertainty relative to models with a strong mechanistic basis.

6.6.1 Assessing Uncertainty in Risk Estimates

Risk assessment exists to support sound management decisions. The uncertainty analysis is intended to make the risk assessment process more transparent by acknowledging and, to the extent possible, quantifying the uncertainty in the risk estimate. An incomplete uncertainty assessment is problematic because it contributes to a false sense of confidence regarding both the accuracy and the precision of the risk estimate. Identifying sources magnitude of uncertainty accomplishes two objectives: (1) it helps decision-makers determine whether additional information should be obtained prior to making a decision; and (2) provides a qualitative context for each particular risk estimate. Uncertainty analyses for DERAs should incorporate the following considerations:

1. *Identify and characterize sources of uncertainty* – Describe what is known and what is not known. Are we dealing with something that is unknowable, or about which we are totally ignorant? What would it take to reduce the uncertainties? Some uncertainties can be reduced and some cannot.

[62] "All models are wrong; some models are useful." George E.P. Box, statistician

2. *Quantify uncertainty in the risk estimate* – Quantitative uncertainty analyses (*e.g.*, with probabilistic methods such as Monte Carlo simulations and probability bounds analysis[63]) allow the assessor to assess when further study is needed and when decisions can be made in the presence of uncertainty. In general, quantifying the uncertainty for models or other highly quantitative risk assessment tools involves the following steps:

   - List all uncertain parameters (include additional parameters if necessary to represent uncertainty in model structure), and determine the maximum range of potential values for each uncertain parameter.

   - Determine a probability distribution for values occurring within this range. Consider correlations among parameters (*e.g.*, if a maximum value is likely for one parameter, then what would be the likely values for other correlated parameters?). The objective of this step is to avoid having all parameters set to a maximum if such a scenario is ecologically irrelevant or otherwise impossible.

   - Propagate the uncertainty in the model parameters to produce a probability distribution of model predictions, and prepare quantitative statements of uncertainty in terms of a confidence interval for the risk estimate that reflects the range of parameters used to calculate the risk estimate.

   - Rank the parameters contributing most to uncertainty in the model prediction by performing a sensitivity analysis.

3. *Describe uncertainty in the risk estimate* – A quantitative approach to uncertainty analyses is preferred; however, it may not be possible (or appropriate) in all instances. A qualitative approach that follows the same logic as described above is recommended for those lines of evidence that do not lend themselves to the quantitative method described above. The qualitative approach involves a narrative description of: (1) which lines of evidence were used in the risk estimate; (2) how the results from individual areas or samples were integrated into an overall site-wide risk estimate; (3) how different lines of evidence relate to one another; (4) what the risk estimates could be if the worst-case values from individual areas or samples were used; and (5) which of the lines of evidence had the greatest influence on the risk estimate. Note that some lines of evidence may incorporate individual quantitative uncertainty analyses (*e.g.*, statistical power of a toxicity experiment can be calculated) in addition to the qualitative uncertainty analysis of the overall risk estimate.

[63] Monte Carlo methods are appropriate when input distributions are known precisely; however, they may not adequately represent the effects of uncertainty about how to parameterize variability in the input distributions. Probability bounds analysis is a tool for separating variability and uncertainty to obtain bounds on the result that explicitly account for uncertainty about the input distributions. As in Monte Carlo analysis, the overall slopes of the bounds indicate how much variability exists in the system. The distance between the bounds, on the other hand, is an indication of the uncertainty that exists due to lack of knowledge (*i.e.*, incertitude).

6.6.2 When to Refine Risk Estimates

If risks are measured or predicted (*e.g.,* HQ > than 1, limited toxicity), the practitioner (and client) must make a decision to: (a) further refine the exposure or effects assessments to reflect site-specific conditions, or (b) conclude that risk is unacceptable or unresolvable and that remediation and/or risk management options should be considered. The qualitative uncertainty assessment influences the appropriate degree of precaution with respect to the need to conduct additional investigations to reduce uncertainty. A matrix based on varying levels of estimated risk and uncertainty (based on Persons and Hopley, 1999) is proposed:

	Low Magnitude of Risk	**High Magnitude of Risk**
Low Uncertainty in Risk Estimate	Low Precaution	Medium Precaution
High Uncertainty in Risk Estimate	Medium Precaution	High Precaution

Refinement of risk estimates for the "high" category of precaution is recommended; the "medium" category of precaution may also indicate a need to reduce uncertainty as necessary to support management actions. This refinement may involve one or more of the following strategies:

- Reduce parameter uncertainty by gathering additional data. Supplemental data collection should be targeted to deal with the underlying cause of the parameter uncertainty (*e.g.*, address spatial coverage, improve analytical detection limits, collect bioavailability information, evaluate cause and effect mechanisms).

- Reduce structural (model) uncertainty by adopting a more appropriate model (or increasing the sophistication of the existing model). Risk assessment should be an iterative process where new data may require reassessment of previous approaches or conclusions. This iterative process allows risk assessment to be a dynamic process well suited to ecological study, and does not indicate a failure of the initial screening risk estimate.

- Provide risk managers with multiple risk scenarios for consideration as a series of risk estimates with different assumptions and descriptions of uncertainty.

Several other strategies are often employed; however, they do not directly reduce parameter or model uncertainty. For example:

- Professional judgment is often used to fill in gaps in model structure. This may reduce uncertainty, but it may not, and there is no objective way to know. Conservative assumptions are often used as part this strategy; although it does not reduce uncertainty, it ensures that the majority of the uncertainty errs on the side of caution. The challenge in using conservative assumptions lies in balancing conservatism and ecological realism relative to site management needs.

- Increase the number and types of lines of evidence considered in a weight of evidence approach. This strategy does not reduce the uncertainty in any single line of evidence, but does reduce overall uncertainty in the conclusions of the risk assessment because the limitations of one line of evidence are frequently balanced by the strengths of another.

6.7 Linking Risk Assessment with Risk Management

Risk managers "use the results of the risk assessment, along with information on technical feasibility, and social, economic and political concerns to reach a decision" (CCME, 1996). Dialogue between the risk assessor and risk manager throughout the risk assessment process is useful, particularly with respect to how study results will be applied to management decisions. Separation of risk management and risk assessment may be more difficult when clients request input from risk assessors regarding appropriate (or likely to be acceptable) management actions; in those cases, any advice or conclusions regarding risk management should be clearly distinguished from the risk estimates.

All parties involved should understand that ERA involves multiple, conflicting goals, namely (Barnthouse, 2008):

- Ensure that the assessment addresses management needs;
- Maintain the distinction between management and science;
- Use the best available science;
- Use all available and relevant science;
- Ensure that the assessment process is transparent; and,
- Ensure that the methods and results are comprehensible to decision makers and stakeholders.

Barnthouse (2008) notes that these goals clearly transcend science and ERA is not a conventional scientific discipline like chemistry, toxicology, or ecology. He observes that "ERA is best viewed as a bridge between science and management". The risk assessor should deliver a document that provides a systematic approach to organizing scientific information to support environmental decision-making.

7.0 REPORT LIMITATIONS

This report was prepared for the exclusive use of the Science Advisory Board (SAB) and is intended to provide a professional opinion related to guidance for detailed ecological risk assessment (DERA). Although references are made to certain policy statements made by the Ministry of Environment (MOE), the interpretations of those statements are those of the authors, and such interpretations should not be construed to represent official MOE positions. Furthermore, recommendations made with respect to specific methods, approaches, and interpretative tools have not received MOE endorsement, and should not be viewed as a code of practice for detailed ecological risk assessments.

Any use that a third party may make of this report, or any reliance on or decisions made based on it, are the responsibility of the third parties. We disclaim responsibility for consequential financial effects on site management, or requirements for follow-up actions and costs.

The services performed as described in this report were conducted in a manner consistent with the level of care and skill normally exercised by other members of the science professions currently practicing under similar conditions, subject to the time limits and financial and physical constraints applicable to the services. This report provides professional opinion and, therefore, no warranty is expressed, implied, or made as to the conclusions, advice and recommendations offered in this report. This report does not provide a legal opinion regarding compliance with applicable laws or regulations.

8.0 CLOSURE

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TABLES

TABLE 1: Levels of Biological Organization for Selecting Receptors of Potential Concern for Generic Ecosystem Types Considered in a Detailed ERA

Generic Ecosystems	Receptor Group	
	Lower Level of Resolution	**Higher Level of Resolution**
Aquatic **1. Deep Aquatic** **2. Shoreline** **3. Streams & Rivers**	Primary producers	Phytoplankton and periphyton Aquatic macrophytes
	Water-column invertebrates	Zooplankton Invertebrate planktivores
	Benthic community	Epibenthic invertebrates Invertebrate filter-feeders Benthic infauna
	Fish	Detrivorous fish Planktivorous fish Piscivorous fish
	Mammals	Piscivorous mammals
	Waterfowl	Piscivorous birds Benthivorous brids Detrivorous birds
	Amphibians	Amphibians
Terrestrial **4. Uplands (Wildlands)** **5. Uplands (Human Use)**	Microbes	Microbes
	Invertebrate	Litter-dwelling invertebrates Soil-dwelling invertebrates Arboreal invertebrates
	Plants	Mosses Grasses Shrubs/Trees
	Small mammals	Small mammal ground insectivores Small mammal arboreal insectivores Small mammal omnivores Small mammal herbivores Small mammal carnivores
	Small birds	Avian ground insectivores Avian arboreal insectivores Avian omnivores Avian herbivores
	Large mammals	Large mammal herbivores Large mammal omnivores
	Carnivores	Raptors Carnivorous mammals
	Reptiles	Reptiles
	Amphibians	Amphibians

Note: Multiple ecosystem types and transitional subtypes may exist within the boundaries of a single site, which may influence ROPC selection.

TABLE 2: Existing Guidance for Selecting Mammalian and Avian ROPCs based on Land Use Considerations

Receptor Group	Land Use Designation				
	Industrial	**Commercial**	**Residential**	**Urban Park**	**Agricultural**
Large mammals (*e.g.*, deer, elk, bear, coyotes, fox, skunk, raccoon)	Excluded	Excluded	Excluded	Included	Included
Large rodents (*e.g.,* rabbits, beaver)	Excluded from urban areas	Excluded from urban areas	Excluded	Included	Included
Mustelids (*e.g.,* rabbits, beaver)	Excluded	Aquatic mustelids may be included	Aquatic mustelids may be included	Aquatic mustelids may be included	Aquatic mustelids may be included
Small rodents (*e.g.,* mice, vole)	Included	Included	Included	Included	Included
Bats (*e.g.,* little brown myotis)	Excluded	Excluded	Included	Included	Included
Aquatic birds (*e.g.,* shorebirds, wading birds, waterfowl, seabirds)	Excluded	Excluded	Included	Included	Included
Raptors (*e.g.*, eagle, osprey)	Include if threatened/ endangered	Include if threatened/ endangered	Include if threatened/ endangered	Included	Included
Galliforms (*e.g.*, pheasant, quail)	Excluded from urban areas	Excluded from urban areas	Excluded	Included	Included
Cavity-dwelling birds (consumers of foliar invertebrates)	Excluded	Excluded	Included	Included if trees are present	Included if trees are present
Hummingbirds	Excluded	Excluded	Excluded	Excluded	Excluded

Source: ERAGT, 1998 (check the most recent MOE policy decision summary for latest guidance).

TABLE 3: Classification of Common Toxicity Tests for the Purpose of DERA

Test	Classification	Regulatory Agency	Rationale
Water Toxicity Tests			
48-h cladoceran (*Daphnia* sp.) survival	Acute	Environment Canada USEPA	Described as an "an acute test with the additional endpoint of immobility" by Environment Canada (EC 1RM11). Also, described as "acute" by USEPA (EPA-821-R-02-012).
48-h cladoceran (*Ceriodaphnia dubia*) survival	Acute	USEPA	Described as "acute" by USEPA (EPA-821-R-02-012).
48 or 96-h mysid survival (various species)	Acute	USEPA	Described as "acute" by USEPA (EPA-821-R-02-012).
96-h fish survival (various species)	Acute	Environment Canada USEPA	Described as "acute" by Environment Canada (EC 1RM9, EC 1RM10) and USEPA (EPA-821-R-02-012).
48-h bivalve larval development (various species)	Chronic Surrogate	USEPA	Described as a "estimate of chronic toxicity" by USEPA (EPA/600/R-95/136)
48-h echinoid larval development (various species)	Chronic Surrogate	USEPA	Described as a "estimate of chronic toxicity" by USEPA (EPA/600/R-95/136)
7-d cladoceran (*Ceriodaphnia dubia*) survival and reproduction	Chronic	Environment Canada USEPA	Described as "chronic" by Environment Canada. Also notes: "for tests with cladocerans, chronic is typically defined as continuing until three broods are produced." The document also refers to *Daphnia* sp. tests requiring 14 or 21 days duration as chronic (EC 1RM21). Described as "chronic" by USEPA (EPA/821/R-02/013).
20-min echinoid fertilization (various species)	Chronic Surrogate	Environment Canada	Described as "sublethal" by Environment Canada. Also notes an acute test for echinoids would have a duration of "a few days for echinoids, which generally have a life span of 4 – 8 years for sea urchins." However, the document also notes: "The fertilization assay is a sensitive sublethal test. The fertilization assay is not a chronic test, however, because of its very short duration relative to the life spans of the species (some years). The fertilization assay described in this report is not intended to replace chronic toxicity tests using echinoids, because it might not estimate the effects of longer exposures. However, this test can be expected to yield results closer to such chronic tests than would conventional lethality tests with marine or freshwater species" (EC 1RM27).The methodology used by USEPA (EPA-821-R-02-014) is comparable to Environment Canada. This test is described as an estimate of chronic toxicity

TABLE 3: Classification of Common Toxicity Tests for the Purpose of DERA (cont'd)

Test	Classification	Regulatory Agency	Rationale
7-d fish larval survival and growth (various species)	Chronic Surrogate	Environment Canada USEPA	Described as "sublethal" by Environment Canada. Also notes the test "is not of long enough duration relative to the life span of the fish, and is therefore not a chronic test". However, the document also notes: "The seven-day test is sensitive, however, because larval fish are usually among the most vulnerable stages of the entire life cycle. In general, the seven-day test could be expected to estimate the toxicity in a 30-day exposure of early life-stages of fathead minnows closely in some cases, and within a factor of 2 in other cases, but it might sometimes under-predict by an order of magnitude". The 7-d larval fish test" does not necessarily replace chronic toxicity tests, but comes much closer to results of such chronic tests than would a conventional lethality test with juvenile fish" (EC 1 RM22).Described as an estimate of chronic toxicity by the USEPA (EPA-821-R-02-014 and EPA-600-R-95/136)
7-d fish early life-stage survival (various species)	Chronic Surrogate	Environment Canada USEPA	7-d embryo (E) test described as an "acute" test, while the embryo-alevin (EA) test and embryo-alevin-fry (EAF) tests are referred to as "longer" tests. Also notes that "Because of the long life span of salmonids, early life-stage tests do not measure chronic toxicity, although the intent of this test is to estimate approximately, what such sublethal chronic toxicity might be." Also: "Results from full and partial life-cycle tests with several fish species and a variety of chemicals indicate that the early development stages (*i.e.*, embryo, larval, and early juvenile) can be equally or more sensitive to aquatic contaminants than the adults" (EC 1RM28). Described as an estimate of chronic toxicity by USEPA.
9-d fish embryo-larval survival and teratogenicity (various species)	Chronic Surrogate	USEPA	Described as a "chronic estimate" by USEPA (EPA-821-R-02-014). Refer to 7-d fish early life stage for explanation.
72- or 96-h phytoplankton (*Selenastrum capricornutum*) growth inhibition	Chronic	Environment Canada USEPA	Defined as "chronic" by Environment Canada. Also notes that algae are exposed "over several generations" (EC 1RM25). Described as "chronic" by USEPA (EPA-821-R-02-013).
7-d duckweed (*Lemna* sp) growth inhibition	Chronic Surrogate	Environment Canada	Not defined as "acute" or "chronic" by Environment Canada (EC 1RM37).
48-h giant kelp (*Macrocystis pyrifera*) germination and growth	Chronic Surrogate	USEPA	Described as "chronic estimate" by USEPA (EPA-600/R-95/136)

TABLE 3: Classification of Common Toxicity Tests for the Purpose of DERA (cont'd)

Test	Classification	Regulatory Agency	Rationale
Sediment Toxicity Tests			
48-h bivalve larval development (various species)	Acute	PSEP	Not defined as "acute" or "chronic" by PSEP (PSEP, 1995).
48-h echinoderm embryo growth and survival (various species)	Acute	PSEP	Not defined as "acute" or "chronic" by PSEP (PSEP, 1995).
10-d amphipod survival (various species)	Acute	Environment Canada USEPA PSEP	Described as "acute" by Environment Canada. Also notes "amphipod species used for this test are known or presumed to have annual life cycles, so a 10-d test would represent an acute exposure" (EPS1/RM/26). Described as "acute" by USEPA. Note that this test could be considered chronic for two species (*A. abdita* and *L. plumulosus*) because of their relatively short life cycles. Reburial of surviving amphipods is an additional measurement that can be used as an endpoint (EPA/600/R-94/025).
10-d chironomid (*Chironomus* sp) survival and growth	Chronic	Environment Canada USEPA	Described as "chronic" by Environment Canada. In the laboratory, the life span for *Chironomus tentans* is approximately five to six weeks, so a 10-d test exposure would represent at least 10% of the organism's life span (EC 1RM32). Described as a "short term" test by USEPA (EPA/600/R-99/064). However, based on the life cycle of *C. tentans* as mentioned above this test has been classified as a chronic test
10 or 14-d amphipod (*Hyalella azteca*) survival and growth	Chronic	Environment Canada USEPA	Not defined as "acute" or "chronic" by Environment Canada (EPS1/RM/41). Described as "short term" by USEPA (EPA/600/R-99/064). However, life cycle of *H. azteca* is complete in 5 weeks, which represents more than 10% of the organism's life span, leading to a classification as a chronic test
14-d polychaete (*Polydora cornuta*) growth and survival	Chronic	Environment Canada	Described as "chronic" by Environment Canada. Under laboratory conditions the life cycle of the test organism, *P. cornuta*, can be completed in approximately 28 days (EC 1RM41).
20-d (*Neanthes* sp) polychaete survival and growth	Chronic	PSEP	Not defined as "acute" or "chronic" by PSEP although PSEP does note the life cycle of *Neanthes* is completed in 3-4 months. Test is considered chronic because a 20 day test duration is greater than 10% of the life cycle.
23-d chironomid (*Chironomus* sp) emergence	Chronic	USEPA	Described as "chronic" by the USEPA (EPA 600/R-99/064).
28-d amphipod (*Leptocheirus plumulosus*) survival, growth, and reproduction	Chronic	USEPA	Described as "chronic" by USEPA. The life cycle of *L. plumulosus* is complete in 4 weeks. A 28 day test exposure covers more than 10% of the life cycle and is therefore considered a chronic test (EPA/600/R-01/020).

TABLE 3: Classification of Common Toxicity Tests for the Purpose of DERA (cont'd)

Test	Classification	Regulatory Agency	Rationale
42-d *Hyalella azteca* amphipod survival, growth, and reproduction	Chronic	USEPA	Described as a "long term" test by USEPA. Refer to 10-d *H. azteca* test for life cycle information (EPA/600/R-99/064).
Soil Toxicity Tests			
24- or 48-d nematode survival (various species)	Acute	Washington State Dept. of Ecology	Protocol available at: http://www.ecy.wa.gov/pubs/0409044.pdf.
14-d seed germination (various species)	Acute	Washington State Dept. of Ecology	Protocol available at: http://www.ecy.wa.gov/pubs/96324.pdf.
7-d earthworm survival	Acute	ASTM	Described as a lethal, short-term test by ASTM.
14-d earthworm survival	Acute	Environment Canada	Described as an acute test (EPS 1/RM/43).
48 or 72-h earthworm avoidance	Acute	Environment Canada	Described as an acute test (EPS 1/RM/43).
56-d earthworm survival, growth and reproduction	Chronic Surrogate	Environment Canada	Described as "prolonged exposure" by Environment Canada (EPS 1/RM/43); decision to not describe the test as chronic is based on fact that test duration does not meet the criterion of >10% of an organism's life cycle (because earthworms can live for 4 -5 years). However, Environment Canada also notes that the intent of the test is to approximate a chronic exposure.
14- or 21-d seedling emergence and plant growth (various species)	Chronic Surrogate	Environment Canada (Draft)	Draft test methodology (June 2004) does not discuss the test's classification; however, the duration of the test is less than 10% of the lifespan of any of the twelve plant species described in the method (EPS 1/RM/45).
21 to 35-d collembolan (springtail) survival and reproduction (various species)	Chronic	Environment Canada (Draft)	Draft test methodology (August 2005) does not discuss the test's classification; however, the duration of the test is greater than 10% of the lifespan for at least one of the species (*Folsomia candida*; 28-d test duration versus 190 day life maximum life span) described in the method (EPS 1/RM/47).

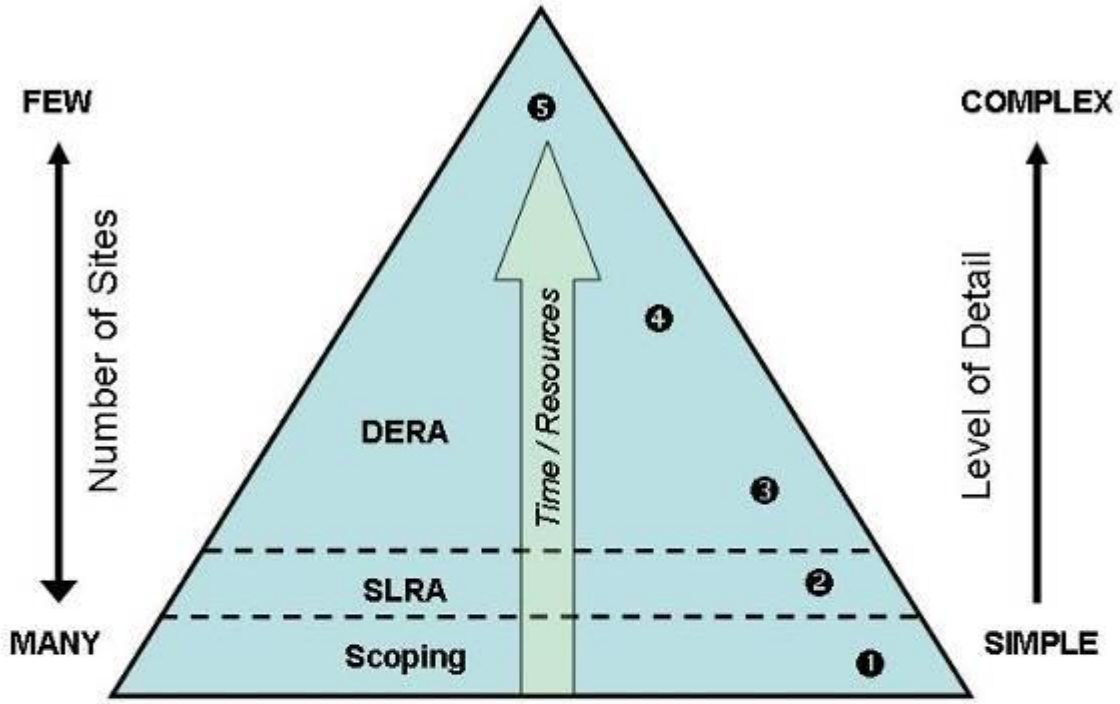
Note: ASTM – American Society for Testing and Materials; PSEP – Puget Sound Estuary Program; USEPA – United States Environmental Protection Agency.

FIGURES

FIGURE 1: Site Management Process under the Contaminated Sites Regulation

Note: Framework extracted from MOE (2003) and ERAGT (1998).

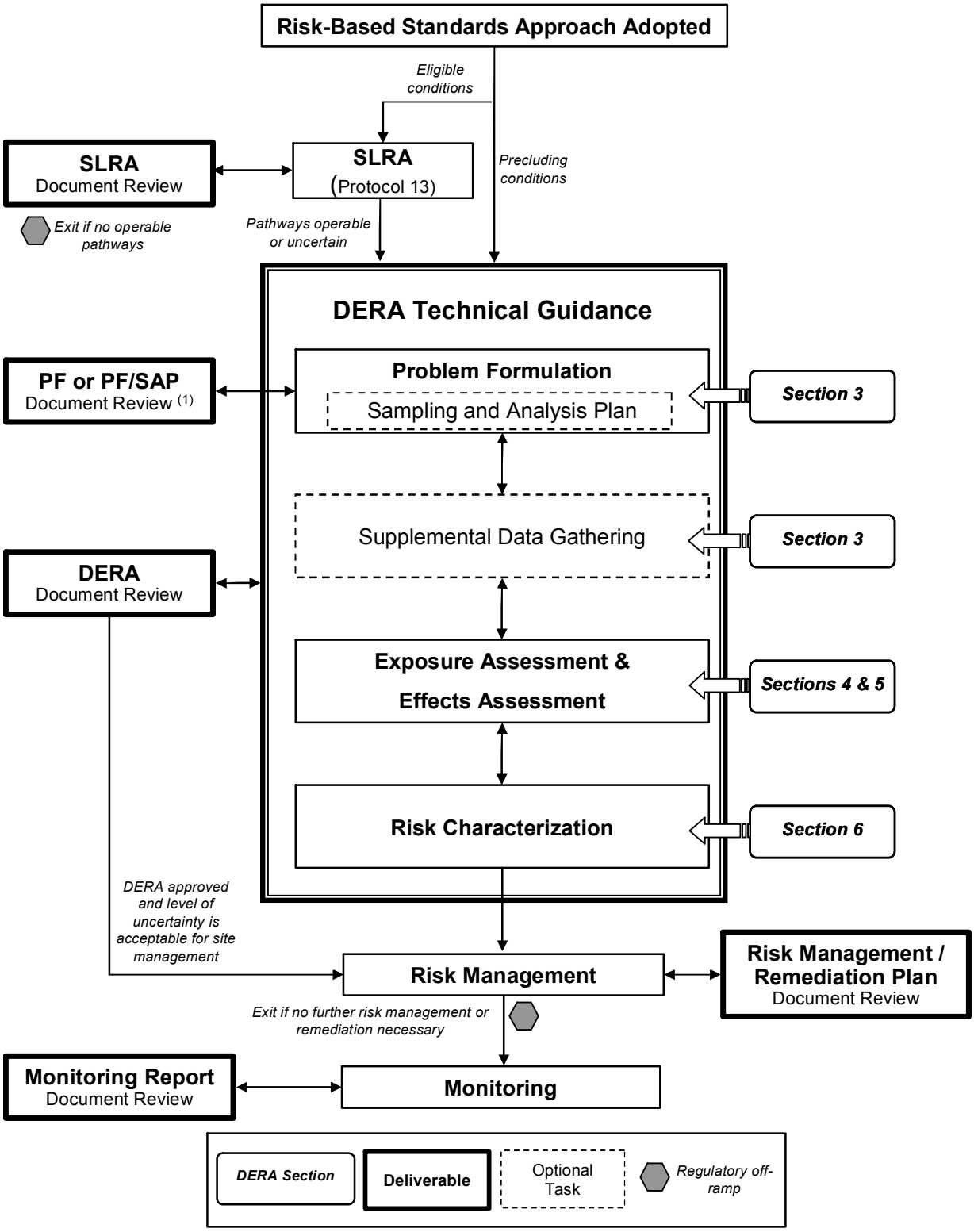
FIGURE 2: Function of DERA in the BC Risk Assessment Framework



Examples of Common Situations Encountered

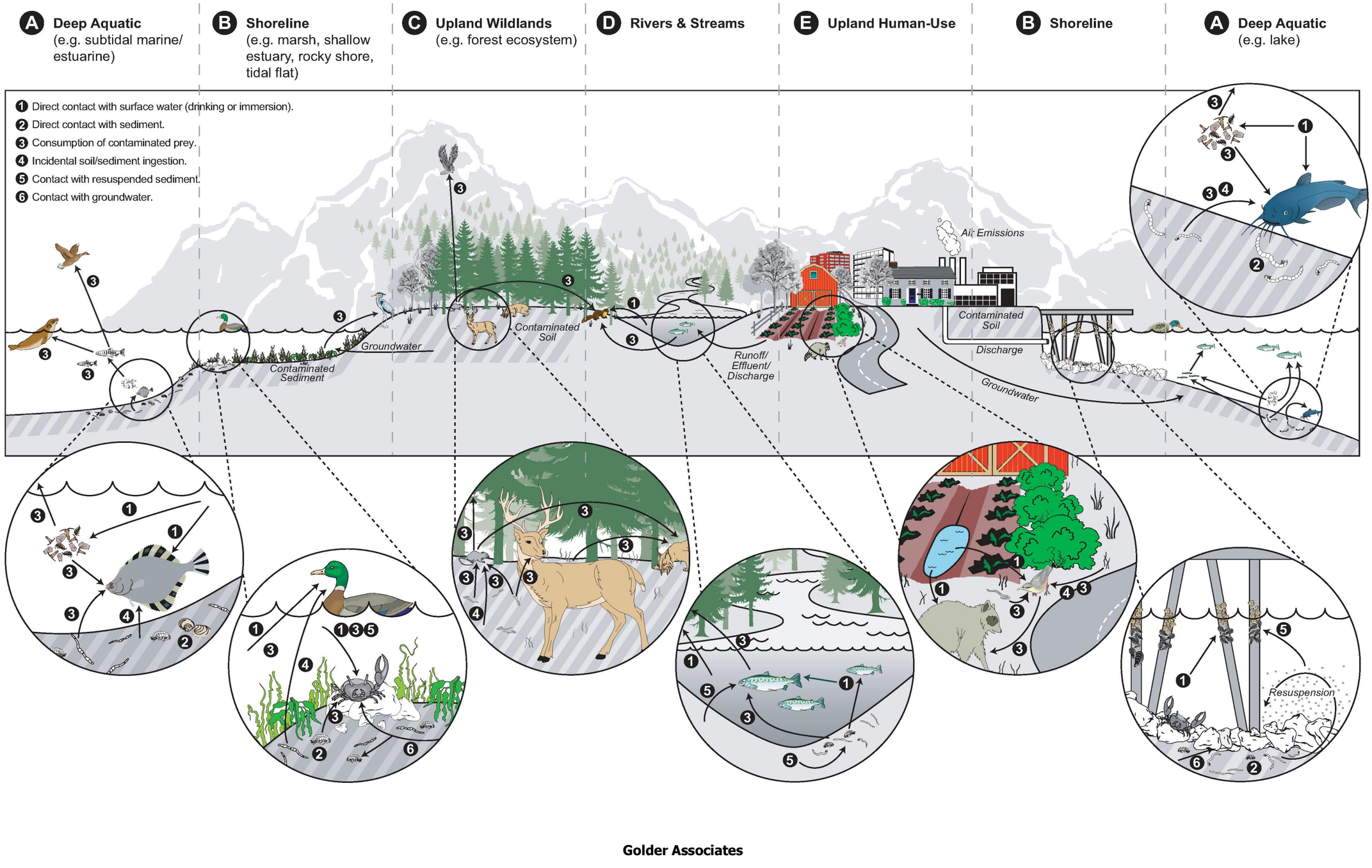
1. Scoping exercise to determine type of risk assessment considered appropriate for the site, if any.
2. SLRA sufficient to screen out all potential exposure pathways; no triggers for DERA present.
3. DERA problem formulation sufficient to bring closure to all outstanding ecological risk issues; no additional sampling or risk assessment required
4. Formal sampling and analysis plan implemented and results incorporated in DERA; subsequent risk estimates adequate to bring closure to outstanding ecological risk issues.
5. Multiple iterations of DERA sampling programs and/or risk assessment phases required to resolve all issues (*i.e.*, detailed ERA for complex site).

FIGURE 3: Process for DERA in British Columbia



(1) PF reviews are mandatory at high risk sites and are recommended (commensurate with the risk and complexity) at other risk sites. For the latter, PF review is at the discretion of the risk assessor and their client, but is strongly encouraged.

FIGURE 4: Broad Ecosystem Types in British Columbia



Golder Associates

FIGURE 5: Example of a Box-Style Conceptual Model

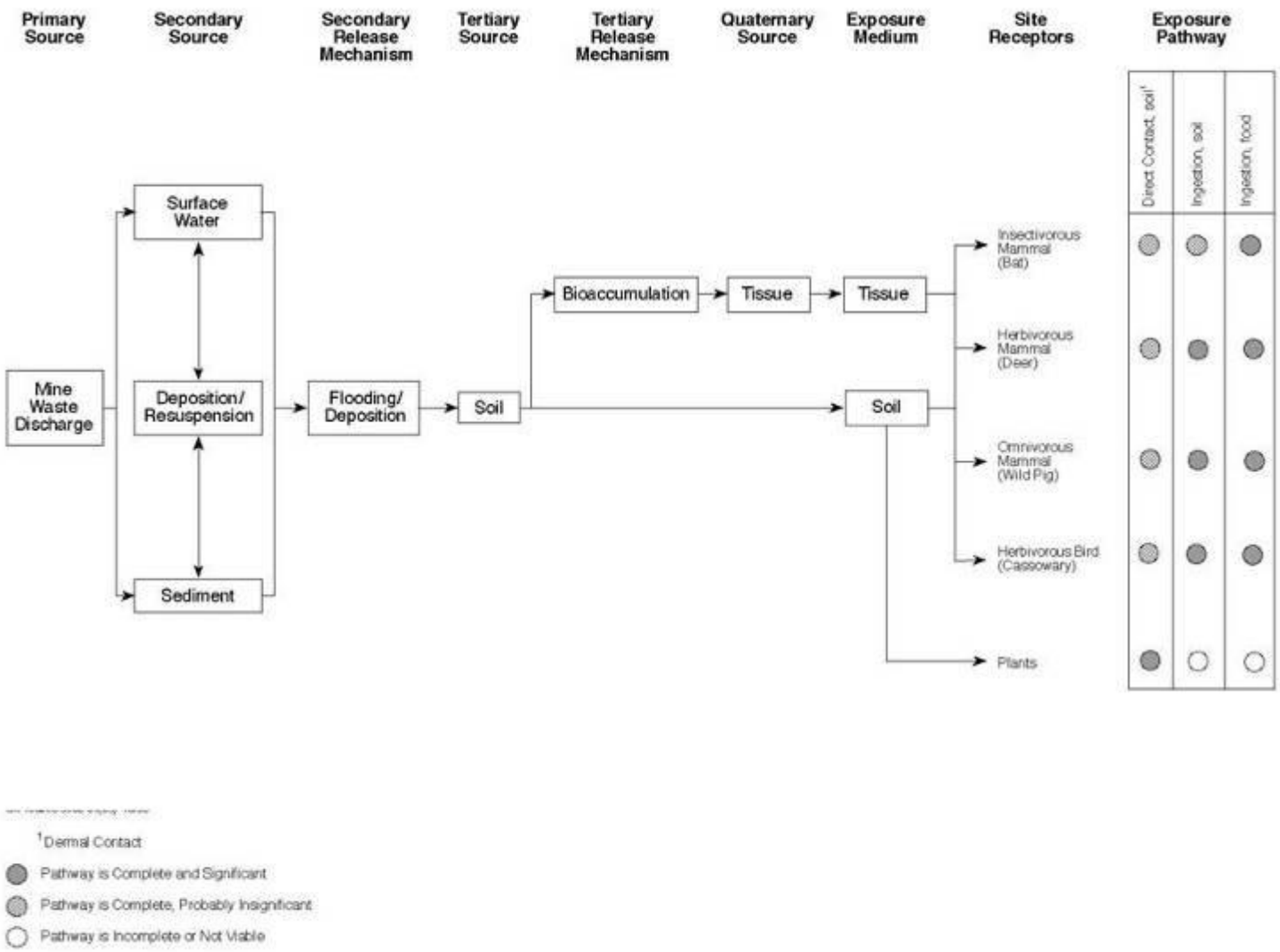


FIGURE 6: Example of Pictorial Conceptual Model

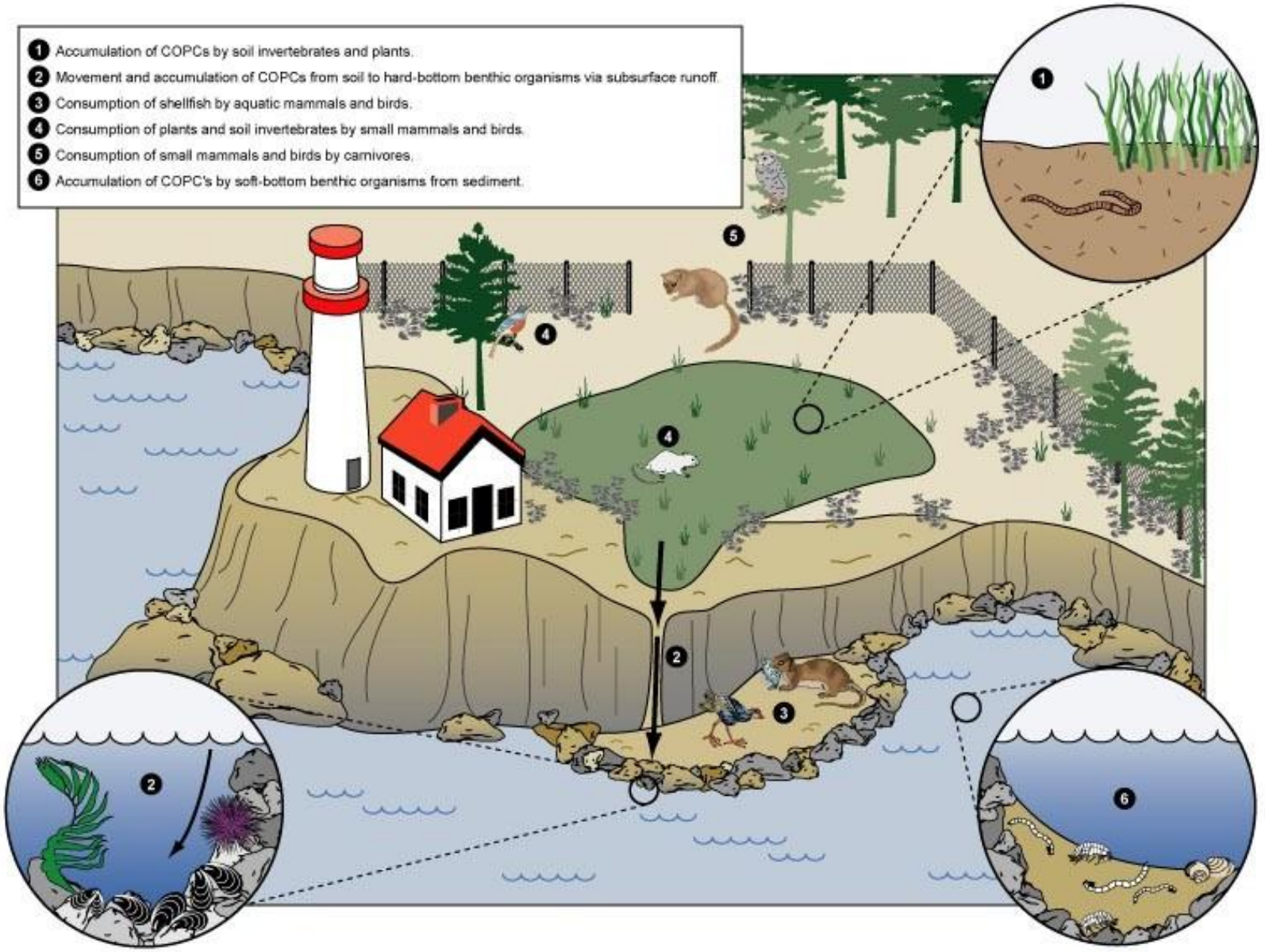


FIGURE 7: Illustrative Example of Tiering Risk Assessment Tools for Assessing Risks to Wildlife Receptors for the Wildlands Ecosystem Type

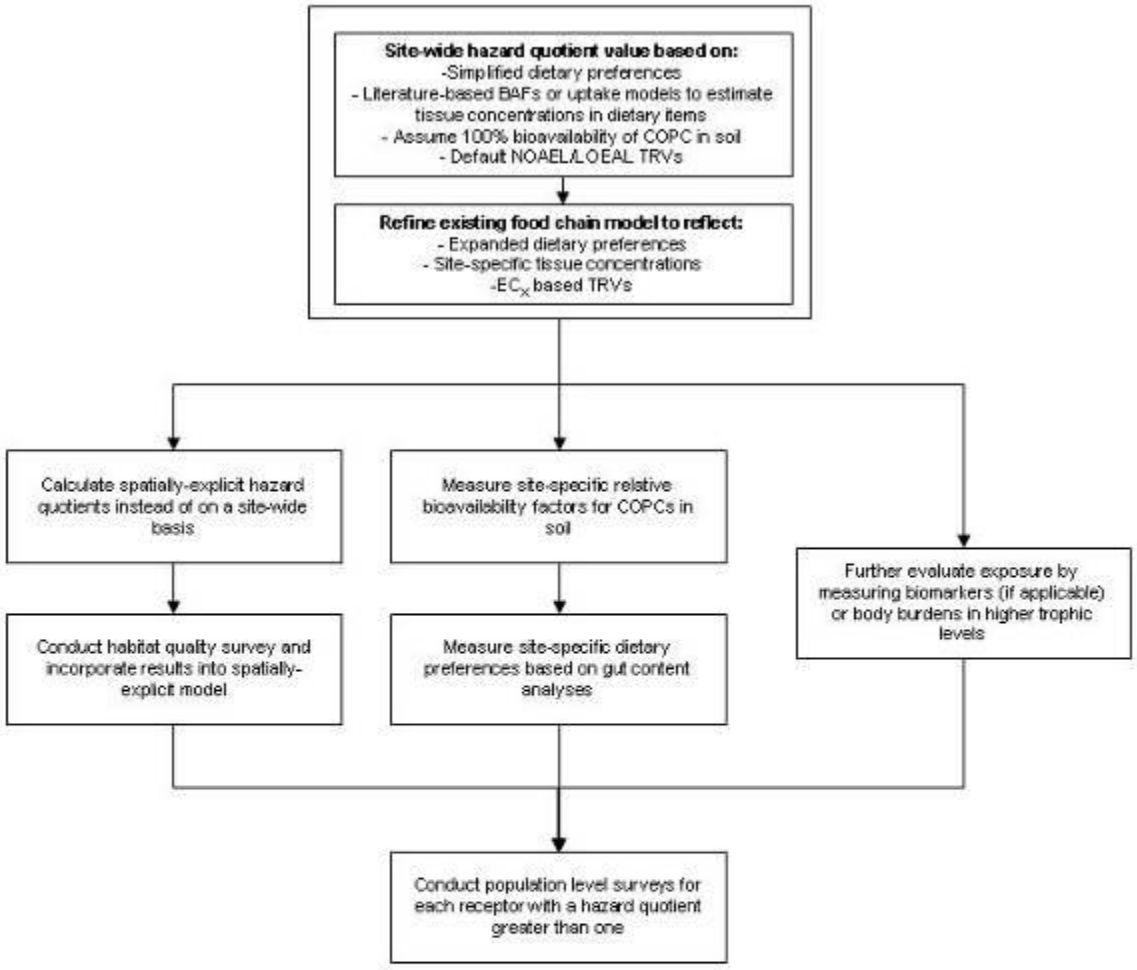
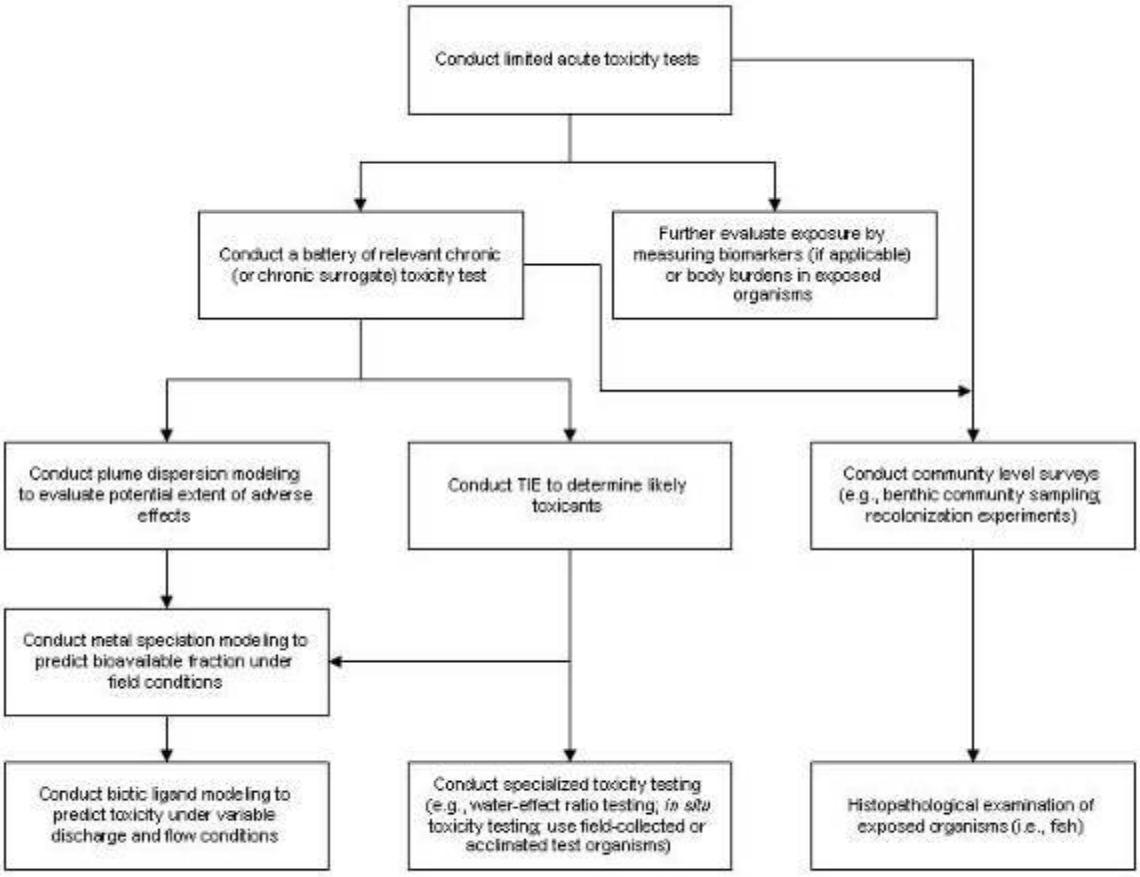


FIGURE 8: Illustrative Example of Tiering Risk Assessment Tools for Assessing Risks to Aquatic Receptors in the Stream and River Ecosystem Type



APPENDICES

APPENDIX I

DIRECT MEASUREMENT TOOLS

TABLE OF CONTENTS

APPENDIX I – TABLES

APPENDIX I – FIGURES

DIRECT MEASUREMENT TOOL #1

1.0 CHEMICAL ANALYSES OF SOIL, WATER AND SEDIMENT

What does this tool consist of? Measurement of the bulk concentration of contaminants in soil, sediment or water (surface water or groundwater) using analytical chemical techniques.

Which ecosystem(s) would this tool typically be applied in? All ecosystems.

How frequently is this tool used in a DERA? Common.

What are the benefits of using this tool in a DERA?

- Chemistry data provide direct measurements of COPC concentrations in the environmental media of concern.

- Concentrations in secondary media (*e.g.*, organism tissue) can be estimated using the results of chemistry data combined with modeling tools (Appendix II).

- Many jurisdictions have published environmental quality criteria/guidelines/standards against which bulk chemistry results can be screened to provide an initial list of COPCs (and an estimate of the magnitude of potential hazard).

- Remediation to numerical standards relies on COPC concentrations.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- Bulk chemistry results do not provide a measure of bioavailability of COPCs or site-specific potential for effects, and therefore must be used in conjunction with other lines of evidence within the DERA framework.

- Numerous ancillary parameters need to be measured to facilitate an appropriate interpretation the data (*e.g.*, *in situ* pH, hardness, TOC, AVS-SEM).

- The manifestation of biological effects may be influenced by the interaction of multiple COPCs (*e.g.*, some parameters may be antagonistic and moderate the effects of another parameter, whereas other contaminants may be additive or synergistic); these interactions cannot necessarily be predicted from bulk chemistry results.

- Not all COPCs can be analyzed with existing laboratory techniques, or if they can be measured, current laboratory techniques may not be able to detect environmental relevant concentrations (*i.e.*, method detection limits may be above concentrations that may cause effects).

Where can I find additional information about this tool? The following guidance manuals are available online:

- Environment Canada. 2002. *Metal Mining Guidance Document for Aquatic Environmental Effects Monitoring*. National Environmental Effects Monitoring (EEM) Office, Environment Canada, Ottawa, Ontario. Available at: http://www.ec.gc.ca/eem/english/MetalMining/Guidance/default.cfm

- Cavanagh, N., R.N. Nordin, L.W. Pommen and L.G. Swain. 1998. *Guidelines for Designing and Implementing a Water Quality Monitoring Program in British Columbia*. Field Test Edition. Ministry of Environment, Lands and Parks (currently the Ministry of Environment), Victoria, BC. Funding provided by Aquatic Inventory Task Force of the Resource Inventory Commission, Forest Renewal British Columbia. Document ID #7680000554. Available at: http://ilmbwww.gov.bc.ca/risc/pubs/aquatic/design/index.htm

- Caux, P.Y., D.R.J. Moore, and D. MacDonald. 1997. *Sampling Strategy for Turbidity, Suspended and Benthic Sediments - Technical Appendix Addendum*. Prepared for BC Ministry of Environment, Lands and Parks (now called Ministry of Environment) by Cadmus Group, Inc. and MacDonald Environmental Sciences Ltd. Funded by: Forest Renewal BC. April 1997. Available at: http://www.env.gov.bc.ca/wat/wq/BCguidelines/samp_strat/sampstrat.html

Practitioners should contact an accredited analytical chemistry lab for specific information about different analytical techniques.

DIRECT MEASUREMENT TOOL #2

2.0 CHEMICAL ANALYSES OF TISSUES

What does this tool consist of? Measurement of the bulk concentration of contaminants in sampled tissues (*e.g.*, whole benthic invertebrates; fish livers, whole bodies, or fillets; soil invertebrates, plants) using analytical chemistry techniques.

Which ecosystem(s) would this tool typically be applied in? All ecosystems.

How frequently is this tool used in a DERA? This tool is commonly used in cases for which persistent COPCs are known to bioaccumulate and/or biomagnify. Direct measurement of COPCs in field-collected tissues is also used when dietary ingestion is a relevant exposure pathway in the DERA. Field measurements of tissue chemistry can also be used to calibrate and/or validate mechanistic models of COPC bioaccumulation.

What are the benefits of using this tool in a DERA? The presence of a COPC in tissues provides an indication that the organism has been exposed, or that it may expose higher trophic levels to the COPC. Measured tissue concentrations of COPCs can be used to quantify exposure of receptors if the effects profile is expressed in terms of internal or dietary concentration. Measured tissue concentrations can also be used as input to a food-chain or ecosystem model, or to validate a food chain or ecosystem model.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- Bulk tissue concentrations do not provide a measure of effect; they provide only a measure of exposure. Many species have the ability to metabolize or otherwise sequester some COPCs (*e.g.*, copper is sequestered by metallothionein in fish; PAHs are metabolized and excreted in bile in fish and some invertebrates).

- Ancillary measurements such as lipid and moisture content may be necessary to interpret the data, as environmental quality guidelines may be presented as "normalized" concentrations. Practitioners should ensure that concentrations are clearly specified as dry-weight, wet-weight, or lipid-normalized measurements.

- The concentration of a given COPC in an organism may be affected by numerous biological factors such as: life stage and sex of the organism; physiological ability to detoxify and/or excrete the COPC; and/or the condition of the exposed organism. Variability in the tissue chemistry data may be high.

- For field-collected samples, there is usually significant uncertainty with respect to spatial pattern and duration of exposure. The uncertainty is greatest for migratory fish, and lowest for sessile species. As remedial targets can be based on concentrations in abiotic media, specification of the linkage between tissue concentrations and abiotic concentrations is often necessary.

Where can I find additional information about this tool? The following guidance manuals are available online:

- Cavanagh, N., R.N. Nordin, P. D. Warrington. 1997. *Freshwater Biological Sampling Manual*. Prepared for the Resources Inventory Committee. BC Ministry of Environment, Lands and Parks, Water Management Branch. Partial funding provided by: Aquatic Inventory Task Force of the Resources Inventory Committee. Document ID 7680000557. Available at: http://ilmbwww.gov.bc.ca/risc/pubs/aquatic/freshwaterbio/assets/freshwaterbio.pdf

- BC Ministry of Environment, Lands and Parks (BCMELP). 1997. *Fish Collection Methods and Standards*. Version 4.0. Prepared for the Resources Inventory Committee by the BC Ministry of Environment, Lands and Parks (currently the Ministry of Environment), Fish Inventory Unit for the Aquatic Ecosystems Task Force, Resources Inventory Committee. Available at: http://ilmbwww.gov.bc.ca/risc/pubs/aquatic/fishcol/assets/fishml04.pdf

- Environment Canada. 2002. *Metal Mining Guidance Document for Aquatic Environmental Effects Monitoring*. National Environmental Effects Monitoring (EEM) Office, Environment Canada, Ottawa, Ontario. Available at: http://www.ec.gc.ca/eem/english/MetalMining/Guidance/default.cfm

DIRECT MEASUREMENT TOOL #3

3.0 CHEMICAL ANALYSES OF POREWATER

What does this tool consist of? Measurement of the bulk concentration of contaminants in porewater (freshwater or marine) using analytical chemical techniques.

Which ecosystem(s) would this tool typically be applied in? Most commonly in Deep Aquatic and Shoreline ecosystems, but also applicable to Rivers and Streams.

How frequently is this tool used in a DERA? Occasionally used in aquatic ERAs.

What are the benefits of using this tool in a DERA?

- Measuring concentrations of COPCs in porewater provides direct information on the sediment-associated contaminant fraction that is likely to be most available to some sediment dwelling organisms.

- Porewater testing can provide a complimentary line of evidence to bulk sediment chemistry data.

- Many toxicity identification evaluations (TIEs) evaluate responses to organisms exposed to porewater because the aqueous fraction of a sediment sample is easily manipulated using common sample adjustments (treatments) that are more difficult to achieve using bulk sediments. Porewater chemistry is required to establish concentration-response relationships in these treatments.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- The choice of collection method should take into consideration the objectives of the sampling program. Porewater samples can be collected using *in situ* methods (*e.g.*, peepers) or ex situ (*e.g.*, centrifugation of bulk sediment), with advantages and disadvantages associated with each. There is a significant amount of literature on collection of porewater samples that should be accessed to determine the best methods for a given site or application.

- There can be difficulty in collecting a sufficient volume of porewater for analytical testing, especially when low detection limits are required, or where sediments contain low moisture content.

- Ancillary parameters often need to be measured to facilitate interpretation of the data (*e.g.*, *in situ* pH, organic carbon, ammonia [NH3], hydrogen sulphide [H2S]).

- It is nearly impossible to avoid artifacts and chemical changes during sampling, extraction and storage (*i.e.*, oxidation changes, *etc*.) of porewater samples.

- Porewater chemistry can vary seasonally.

Where can I find additional information about this tool?

- Winger, P.V., P.J. Lasier, B.P. Jackson. 1998. The influence of extraction procedure on ion concentrations in sediment porewater. *Arch. Environ. Contam. Toxicol.* 35:8-13.

- Carr R.S., Nipper M., Adams W.J., Berry W.J., Burton Jr. G.A., Ho K., MacDonald D., Scroggins R., Winger P.V. 2001. *Summary of a SETAC Technical Workshop: Porewater Toxicity Testing: Biological, Chemical and Ecological Considerations with a Review of Methods and Applications, and Recommendations for Future Areas of Research*. March 18 - 22, 2000, Society of Environmental Toxicology and Chemistry (SETAC), Pensacola, FL. 38 p.

DIRECT MEASUREMENT TOOL #4

4.0 SHORT-TERM/ACUTE TOXICITY TESTS

What does this tool consist of? Toxicity tests are studies specifically designed to determine whether exposure to test organisms via exposure to a given medium causes an adverse effect to those organisms. These tests may be conducted using water, sediment or soil samples, or combinations of these media. Acute toxicity tests are defined as being of short duration relative to the lifespan of the test organism, involving exposures ranging from minutes to a few days. Acute toxicity tests are defined as tests with duration of less than 10% of the lifespan of the test organism.

The endpoint most commonly measured in acute toxicity tests is lethality (or immobilization in the case of the cladoceran *Daphnia* sp.). Some toxicity tests that apply acute exposures also include measurement of sublethal endpoints (*e.g.*, echinoid fertilization, trout embryo viability), and are sometimes used as surrogates for estimating chronic toxicity. Acute toxicity tests are usually conducted in a laboratory under controlled conditions, although they may also be conducted *in situ*. Tests may be conducted using either single-concentration or multi-concentration experimental designs, although sediments are generally tested without dilution. A negative (clean) control must always be tested concurrently, to assess natural background variability in the test population and to determine test acceptability. Acute tests with daphnids or fish do not require replication, but other acute test methods do. Identical numbers of test organisms (of similar size/age) are exposed to the test material for a defined period of time under controlled laboratory conditions (*i.e.*, temperature, light, water quality); the number of surviving organisms in each treatment is determined at the end of the test. Responses in the test treatments are compared to the negative control. If a multi-concentration test was performed, then an LC_p (concentration estimated to be lethal to percentage "*p*" of the test population) can be calculated.

Which ecosystem(s) would this tool typically be applied in? Acute toxicity tests are applicable for all five ecosystem types. Standardized test methods are available for water (which may include groundwater, effluent, leachate, or receiving water), sediment and soil test species.

How frequently is this tool used in a DERA? Acute toxicity tests are commonly used in DERAs.

What are the benefits of using this tool in a DERA?

- Acute toxicity tests provide direct measurements of potential adverse effects to aquatic or terrestrial receptors of concern, information that cannot be obtained from chemistry measurements alone. Toxicity tests account for site-specific factors that may govern the bioavailability and/or toxicity of the substances to representative test organisms.

- Acute toxicity tests conducted in the laboratory are performed under controlled environmental conditions, so that the only variable under investigation is the test material. The use of standard test methods and test species facilitates repeatability and reproducibility, and allows for comparison of data generated by different laboratories for different sites.

- Toxicity tests are useful for evaluating the effects of mixtures of contaminants (including contaminants not measured using conventional analytical chemistry methods) and also provide an indication of the potential contaminant bioavailability under influence of modifying factors such as water hardness, organic carbon content or particle size.

- Toxicity tests can be used to predict potential adverse effects to receptors of concern; this differs from retrospective biological assessment tools such as benthic community structure, which can only show whether than alteration has already occurred.

- Toxicity tests can be useful for identifying whether alterations to biological communities are due to contaminant exposure or some other stressor when interpreted as part of a weight-of-evidence assessment.

- Acute toxicity tests are particularly useful in DERAs as a screening tool in a tiered testing approach. For example, there is little benefit to subjecting samples with high acute lethality to further chronic toxicity testing to evaluate potential sublethal effects. Samples exhibiting high acute lethality can be submitted for Toxicity Identification Evaluation (TIE) testing if identifying the stressor(s) causing the toxicity is important information for site management. Conversely, samples exhibiting little or no acute toxicity could be subjected to further evaluation, such as chronic toxicity testing, in the next tier of the DERA investigation.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- Acute toxicity tests do not provide information about the specific stressor(s) causing the observed toxicity, unless further evaluation is conducted using TIE manipulations.

- Acute toxicity tests do not provide information about sublethal effects such as growth, reproduction or development. On their own, they may provide enough information to identify areas that are not suitable for risk assessment (*i.e.*, high toxicity). However, in-place risk management typically requires other supporting lines of evidence to provide sufficient information regarding potential effects.

- Toxicity tests performed under laboratory conditions may not represent "real world" conditions found in the field. Sample collection, transport, storage and manipulation before and during testing may alter sample properties that influence contaminant bioavailability (*e.g.*, oxidation of an anoxic sample, mixture of micro-scale layering by sediments during homogenization).

- Toxicity tests are performed with a limited number of species. Linkage to the receptors of potential concern and the measurement endpoints in the DERA framework is necessary.

- The sensitivity of a given test species or test protocol to the detection of adverse biological responses is not fully known *a priori*. Although experience can be gained from experiments conducted at other sites, including "round-robin" tests of multiple species within a functional group and/or results of a "test battery" approach, each contaminated site is unique with respect to contaminant mixtures, substrate type, and environmental conditions. Some species and test methods are recognized for yielding significant responses more frequently than others (on average), but site-specific factors prevent universal statements on test sensitivity from being reliable.

Where can I find additional information about this tool?

- ASTM (American Society for Testing and Materials) International. 2004. *Standard Guide for Conducting Acute Toxicity Tests on Test Materials with Fishes, Macroinvertebrates, and Amphibians*. Method E729-96 (re-approved 2002). In: 2004 Annual Book of ASTM Standards, Water and Environmental Technology, Volume 11.05. ASTM International, West Conshohocken, PA.

- Landis, W.G. and M-H Yu. 2004. *Introduction to Environmental Toxicology: Third Edition.* Lewis Publishers, Boca Raton, FL. 328 pp.

- Toxicity test method protocols can be found electronically at:

  Environment Canada: http://www.etc-cte.ec.gc.ca/organization/spd_e.html
  USEPA: http://www.epa.gov/waterscience/WET/
  http://www.epa.gov/ost/library/sediment/

DIRECT MEASUREMENT TOOL #5

5.0 LONG-TERM/CHRONIC TOXICITY TESTS

What does this tool consist of? Toxicity tests are studies specifically designed to determine whether exposure to test organisms via exposure to a given medium causes an adverse effect to those organisms. These tests may be conducted on water, sediment or soil samples. Chronic toxicity tests are defined as being of relatively long duration, involving a substantial portion of the test organism's lifespan (10% or greater). Surrogates for chronic tests are also used (*i.e.*, test has a duration that is less than 10% of the organism's life cycle but measures a sensitive life stage, such as reproduction). In addition to the test duration, an important distinction between acute and chronic tests is the endpoints measured. Although lethality is often measured in both acute and chronic toxicity tests, it is the measurement of sublethal endpoints such as growth, development or reproduction that is most important in chronic toxicity tests. Chronic toxicity tests are usually conducted in a laboratory under controlled conditions in a manner similar to acute toxicity tests, although they may also be conducted *in situ*.

Which ecosystem(s) would this tool typically be applied in? Chronic toxicity tests are applicable for all five ecosystem types. Standardized test methods are available for water (which may include groundwater, effluent, leachate, or receiving water), sediment and soil test species. Test methods are more broadly developed for water-column testing than for sediment or soil testing.

How frequently is this tool used in a DERA? Chronic toxicity tests are likely to be commonly used in DERAs, although chronic "surrogates" are likely to be used more frequently than long-term tests involving full life-cycle exposures.

What are the benefits of using this tool in a DERA?

- All benefits of acute toxicity testing are also applicable to chronic toxicity tests.

- Chronic (and chronic surrogate) toxicity tests are particularly useful in DERAs when they are used in conjunction with acute toxicity tests in a tiered testing framework. Chronic toxicity tests generally require greater expense and effort than acute tests, so using acute testing as a screening tool to identify those samples that warrant further assessment using chronic toxicity testing is beneficial for prioritizing and focusing available resources.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- All pitfalls applicable to acute toxicity testing are also applicable to chronic toxicity testing.

- The potential for an artefact response can increase with test duration. Although most test species used widely in chronic toxicity testing are relatively reliable in terms of culturing and performance, there is always the possibility of a test "crash" due to unforeseen organism susceptibility or analytical error. Chronic toxicity tests entail a greater number of measurements and often require laboratory manipulation during the course of the experiment (*e.g.*, overlying water refreshes).

- Chronic toxicity tests afford greater flexibility in terms of sublethal test endpoints, but the ability of such sublethal endpoints to discern statistically and ecologically significant responses can be impaired by high endpoint variability. For some tests, the duration of exposure and range of endpoints must be balanced against the statistical power of the endpoint.

- The increased cost of chronic testing must be considered relative to the degree of uncertainty reduction afforded by the test. California State Water Resources Control Board (2005) demonstrated that for survival and growth endpoints, the *Leptocheirus* 28-day toxicity test was actually equal to or less sensitive (on average) relative to the 10-d version of the test. Longer is not necessarily better in terms of toxicity testing, particular if chronic testing is conducted at the expense of reduced representation of feeding types and test species.

Where can I find additional information about this tool?

- Toxicity test method protocols can be found electronically at:

Environment Canada:	http://www.etc-cte.ec.gc.ca/organization/spd_e.html
USEPA :	http://www.epa.gov/waterscience/WET/
	http://www.epa.gov/ost/library/sediment/

- USEPA (US Environmental Protection Agency). 2001. Methods for assessing the chronic toxicity of marine and estuarine sediment-associated contaminants with the amphipod *Leptocheirus plumulosus*. US Environmental Protection Agency, Office of Research and Development, Newport, OR. EPA/600/R-01/020. 104 pp.

- California State Water Resources Control Board. 2005. *Sediment Quality Objectives for California Enclosed Bays and Estuaries – Development of Toxicity Indicators*. Bay Protection and Toxic Cleanup Program, Presentation to Scientific Steering Committee Meeting, July 26, 2005.

DIRECT MEASUREMENT TOOL #6

6.0 MULTI-GENERATIONAL TOXICITY TESTS

What does this tool consist of? Toxicity tests are studies specifically designed to determine whether exposure to test organisms via exposure to a given medium causes an adverse effect to those organisms. These tests may be conducted on water, sediment or soil samples. Multi-generational toxicity tests are an extension of full or partial life-cycle chronic toxicity tests.

In chronic toxicity tests that measure reproduction (*e.g.*, three-brood *Ceriodaphnia dubia* cladoceran test, 28-d *Leptocheirus plumulosus* amphipod test), the number of offspring produced by the test organism is commonly used as the reproduction endpoint. The number of offspring (F1 generation) produced is used to quantify reproductive effects in the parents (P generation), without consideration of the quality or condition of the offspring themselves. In a multi-generational toxicity test, test organisms are exposed to the stressor(s) of concern for two full generations, from the egg stage of the P generation through to the production of juveniles of the F2 generation. The F1 and F2 generations are isolated and reared under the same exposure conditions that were used for the P generation. Each generation may be evaluated in terms of effects on survival, growth and hatching success; the P and F1 generations may also be evaluated in terms of endpoints such as time to maturity, sex ratios, fecundity, and development of secondary sex characteristics.

Which ecosystem(s) would this tool typically be applied in? Multi-generational chronic toxicity tests are applicable to all five ecosystem types.

How frequently is this tool used in a DERA? Multi-generational toxicity tests are rarely used in DERAs. They have been performed in experimental investigations of mammalian and avian toxicology, but are most commonly applied to the evaluation of specific contaminants rather than for contaminant mixtures representative of contaminated sites. As such, results must typically be extrapolated from the study to the site of interest.

What are the benefits of using this tool in a DERA?

- Multi-generational toxicity tests may be useful for the assessment of contaminants of potential concern associated with adverse teratogenic or endocrine-disrupting effects on receptors of concern. Such effects might not be apparent in the parent generation but would be manifested in the offspring of the first or subsequent generations.

- Multi-generational toxicity tests provide a rigourous measure of the potential for adverse chronic effects, because of their extended duration relative to the organism's lifespan.

- The spiking studies typically used for multigenerational studies can, if properly designed, provide a wide range of exposures suitable for development of a concentration-response curve.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- Life history characteristics of the candidate test organism need to be considered, and may limit the number of suitable test species. Ideally, the test species should have a fairly short life cycle with an early onset of sexual maturity (to reduce the overall length of the exposure period) and consistently produce large numbers of offspring to provide sufficient numbers of test organisms available from the F1 and F2 generations.

- As the exposure time increases for any toxicity test, the chance of an unexpected event (*e.g.*, equipment failure, reduced organism health) leading to a catastrophic loss of experimental data increases. Costs associated with multi-generational toxicity tests are likely to be high because of the increased degree of monitoring and need for measurement of test endpoints throughout the study.

- Multi-generational tests are not performed routinely and therefore the toxicology database is limited. Accordingly, there is greater uncertainty and variability variability associated with each endpoint. This may make interpretation of the test results more difficult.

- All chronic toxicity tests, including multi-generational tests, are sensitive to confounding or synergistic factors related to such factors as co-occurring contaminants, levels of essential nutrients, dietary type, and animal husbandry. Discerning the effects of contaminants from other experimental factors can be difficult.

Where can I find additional information about this tool?

- Lock, K. and C.R. Janssen. 2002. Multi-generation toxicity of zinc, cadmium, copper and lead to the potworm *Enchytraeus albidus*. *Environ. Pollut.* 117:89-92.

- Newsome, C.S. 1980. A multigeneration fish toxicity test as an aid in the hazard evaluation of aquatic pollutants. *Ecotox. Environ. Saf.* 4:362-369.

- Patyna, P.J., R.A. Davib, T.F. Parkertonb, R.P. Brownb and K.R. Coopera. 1999. A proposed multigeneration protocol for Japanese medaka (*Oryzias latipes*) to evaluate effects of endocrine disruptors. *Sci. Tot. Environ.* 233(1-3):211-220.

- Shellenberger, T.E. 1978. A Multi-Generation Toxicity Evaluation of p,p'-DDT and Dieldrin with Japanese Quail: I. Effects on Growth and Reproduction. *Drug and Chemical Toxicology* 1(2):137-146.

- Vandenbergh, G.F., D. Adriaens, T. Verslycke and C.R. Janssen. 2003. Effects of 17α-ethinylestradiol on sexual development of the amphipod *Hyalella azteca*. *Ecotox. Environ. Saf.* 54(2): 216-222.

DIRECT MEASUREMENT TOOL #7

7.0 *IN SITU* TOXICITY TESTS

What does this tool consist of? *In situ* toxicity tests involve conducting toxicity tests in the field (*i.e.*, at the location under investigation) rather than in the laboratory. These tests can be conducted to evaluate water and/or sediment toxicity, using techniques adapted from laboratory-based acute or chronic toxicity test methods. *In situ* exposures can also be designed to provide information on contaminant update and accumulation, similar to laboratory-based bioaccumulation tests.

Test organisms, of similar size/age and obtained from an uncontaminated location, are placed in screened enclosures that allow contact with the environmental compartment of interest. Concurrent placement of additional enclosures in uncontaminated reference locations (*e.g.*, upstream of the study area, or in a separate waterbody) is conducted to assess natural background responses of the test organisms. The enclosures may be suspended in the water column or anchored to be in contact with the sediment surface. *In situ* toxicity tests conducted with "eyed" eggs of salmonid fish may involve burying incubation enclosures in gravel and monitoring development to assess mortality and hatching rate. The size of the enclosures depends on the size and type of test organism being used, and the screen size needs to be such that organisms cannot escape, but such that water can flow through without the screen becoming fouled or clogged. At the end of the exposure period, surviving test organisms are recovered. If the experimental design includes assessment of sublethal endpoints (*e.g.*, growth) or tissue chemistry analyses, these measurements are made using the surviving specimens from each treatment. Responses among exposure and reference treatments are compared.

Which ecosystem(s) would this tool typically be applied in? *In situ* toxicity tests are applicable for all five generic ecosystem types, but are most common in the aquatic ecosystems. Fish and bivalves have been used most often for *in situ* testing, but other invertebrates (*Hyalella*, *Chironomus*, *Daphnia*, *Lumbriculus*, *etc*.) have also been used.

How frequently is this tool used in a DERA? *In situ* toxicity tests are used occasionally for DERAs. They tend to be applied in complex or detailed ERAs for which the extrapolation from laboratory to field responses is a significant concern.

What are the benefits of using this tool in a DERA?

- As with laboratory toxicity tests, *in situ* toxicity tests provide a direct measure of potential adverse effects of exposure on test organisms that cannot be determined from chemistry measurements alone.

- The primary benefit of *in situ* toxicity tests is that they allow direct exposure of test organisms to actual site conditions, and therefore eliminate the need for extrapolation of laboratory-based toxicity testing results to field conditions. *In situ* exposures integrate the environmental variables to which organisms would normally be exposed at a given location (*e.g.*, fluctuations in temperature, water flow, water quality, food supply) as well as factors that may affect the bioavailability of the contaminants of potential concern.

- *In situ* toxicity tests can be designed to use acute or chronic exposures, and to measure lethal and/or sublethal effects, provided that the test species chosen is able to tolerate the exposure without demonstrating adverse effects in the reference locations.

- *In situ* toxicity tests can accommodate simultaneous exposure via water and sediment, using flow-through chambers. Designs can be adjusted such that water-only exposures are simulated (*i.e.*, engineered obstruction of the sediment pathway in the test chambers), thus providing additional information on the source of any observed toxicity.

- Provided that stocks are available from an uncontaminated location, it may be possible to use native species for *in situ* testing, rather than surrogate species such as those that are typically used in standardized laboratory toxicity tests.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- *In situ* testing requires that approved transplant permits from applicable regulatory authorities be in place prior to conducting testing. Depending on the target test species, it is possible that permission to transplant test organisms in sensitive watersheds may not be granted. This may make it difficult to locate populations of naïve test organisms that have no previously been exposed to the stressor(s) of concern.

- Although *in situ* toxicity tests represent more realistic exposure scenarios than then controlled conditions associated with laboratory experiments, there is a higher degree of variability associated with the field exposures and that can make interpretation of *in situ* test results more difficult. Depending on the exposure duration, fluctuations in temperature and food supply may affect the health of the test organisms and their physiological response to the stressor(s) of concern.

- There is a risk of test chambers being lost or damaged during the exposure period, as a result of adverse weather conditions (storms, high or low water flows), predation or theft, and therefore the loss of associated data. Logistics associated with inspection

and monitoring of enclosures during the *in situ* exposure requires consideration of how the enclosures will be anchored, their accessibility, and how to inspect them without causing undue stress to the test organisms as a result of disturbance.

- *In situ* testing requires additional toxicology expertise relative to standardized toxicity test procedures. Costs are also generally higher due to the need for field mobilization and monitoring.

Where can I find additional information about this tool?

- ASTM International. 2004. *Standard Guide for Conducting In-situ Field Bioassays with Caged Bivalves*. Method E2122-02. In: 2004 Annual Book of ASTM Standards, Water and Environmental Technology, Volume 11.05. ASTM International, West Conshohocken, PA.

- BCMWLAP (British Columbia Ministry of Water, Land and Air Protection). 2003. *British Columbia Field Sampling Manual for Continuous Monitoring and the Collection of Air, Air-emission, Water, Wastewater, Soil, Sediment, and Biological Samples.* British Columbia Ministry of Water, Land and Air Protection, Water Air and Climate Change Branch, Victoria, BC. January 2003. 383 pp.

- Chappie, D.J. and G.A. Burton Jr. 2000. Application of aquatic and sediment toxicity testing *in situ*. *Soil Sed. Contam.* 9:219-245.

- Environment Canada. 1999. *Guidance Document on Application and Interpretation of Single-Species Tests in Environmental Toxicology*. Environmental Protection Series, Report EPS 1/RM/34, December 1999. Environment Canada, Method Development and Application Section, Environmental Technology Centre, Ottawa, ON.

DIRECT MEASUREMENT TOOL #8

8.0 BEHAVIOURAL TOXICITY TESTS

What does this tool consist of? Toxicity tests are studies specifically designed to determine whether exposure to a particular substance or exposure medium causes an adverse effect in a group of test organisms. These tests may be conducted using water, sediment or soil samples. Behavioural toxicity tests can be used to measure sublethal responses, and are considered separate from tests that evaluate growth, reproduction, and/or development. Examples of behaviours that can be assessed include changes in locomotion, respiration, habitat selection, feeding, avoidance (of predators or contaminants), competition, and reproductive behaviour. Changes in behavioural responses are compared to controls to determine whether the observed change in outside the typical range of variability for that species-behaviour combination. These tests can involve short-term or long-term exposures.

Which ecosystem(s) would this tool typically be applied in? Behavioural toxicity tests are applicable for all five ecosystem types. There is a standardized avoidance test method for soil using earthworms.

How frequently is this tool used in a DERA? Behavioural toxicity tests are likely to be used rarely in DERAs, except that earthworm avoidance tests are used occasionally. The main reason for the infrequent application of behavioural tests is that the ecological significance of the endpoints is often difficult to interpret.

What are the benefits of using this tool in a DERA?

- The 48-h acute avoidance test with earthworms has been found to be a useful tool for screening soil samples to be included for 56-d chronic toxicity tests.

- Behavioural toxicity tests provide an alternative mechanism for assessment of sublethal effects in receptors of concern.

- Behavioural endpoints can be used as surrogates for the survivability of offspring, and as such can be used as a substitute for multi-generational tests. For example, changes in flight-stimulus response and feeding behaviour can be used as indicators of health of young birds, without requiring a full second generation assessment.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- The behavioural characteristics of the test organism must be understood well before they can be used as measures of sublethal responses. Criteria for defining measured responses also need to be unambiguous, so that subjective judgement of behaviour by observers is avoided.

- Behavioural toxicity test results can easily be influenced by test organism health, care and handling, testing conditions, and prior exposure or experience with the stressor prior to testing.

- Interpretation of behavioural toxicity test results in the context of ecological effects is complicated because of uncertainty as to whether the observed behavioural change is likely to impact on relevant endpoints such as survival, growth or reproduction. If the behavioural change is associated with a short-term exposure, organisms may be able to recover without any long-term effects.

- In contrast to growth, reproduction, and/or development endpoints, behavioural endpoints generally have a weaker linkage to the ERA assessment endpoints.

Where can I find additional information about this tool?

- ASTM International. 2004. *Standard Guide for Behavioural Testing in Aquatic Toxicology*. Method E1604-94 (re-approved 2002). In: 2004 Annual Book of ASTM Standards, Water and Environmental Technology, Volume 11.05. ASTM International, West Conshohocken, PA.

- Environment Canada. 2004. *Biological Test Method: Tests for Toxicity of Contaminated Soil to Earthworms* (*Eisenia andrei*, *Eisenia fetida*, *or Lumbricus terrestris*). Environmental Protection Series, Report EPS 1/RM/43, June 2004. Environment Canada, Method Development and Application Section, Environmental Technology Centre, Ottawa, ON.

- Morgan, J.D., G.A. Vigers, D.M. Janz, A.P. Farrell and J. Manville. 1991. Acute avoidance reactions and behaviourial responses of juvenile rainbow trout to Garlon 4, Garlon 3A and Vision herbicides. *Environ. Toxicol. Chem.* 10:73-79.

DIRECT MEASUREMENT TOOL #9

9.0 TOXICITY IDENTIFICATION EVALUATION (TIE)

What does this tool consist of? Toxicity identification evaluations (TIEs) consist of side-by-side toxicity testing using manipulated and non-manipulated samples. Manipulations (chemical or physical) are selected to target specific toxicants (or groups of toxicants) known or suspected to be present in a sample. Differences in the toxicity between the manipulated and non-manipulated samples support inferences about chemical compounds or sample-related factors that are contributing to the original toxicity.

Which ecosystem(s) would this tool typically be applied in? TIEs can be applied in all five ecosystem types. TIE procedures are relatively well-developed for aqueous samples (*e.g.*, porewater, groundwater, overlying water, and effluent), somewhat less developed for whole sediments, and relatively limited for soils. Techniques for whole sediment TIEs have received increased attention in recent years.

How frequently is this tool used in a DERA? Rare for DERAs involving soils, but occasional for DERAs involving sediment and/or porewater. TIEs are more commonly applied to DERAs that evaluate aqueous samples, particularly where the sample consists of a discharged effluent, wastewater, contaminated groundwater, or stream.

What are the benefits of using this tool in a DERA? TIEs directly evaluate cause-effect relationships. TIEs can be used to determine the relative influence of physical versus chemical-related effects. Assessing the relative contribution of different chemicals also improves the ability of the risk characterization to guide appropriate risk management planning. TIEs are particularly useful for identifying contributions of ancillary chemicals (*e.g.*, ammonia, sulphide, dissolved oxygen) to observed toxicity. At many sites, effects are often incorrectly ascribed to contaminants (*e.g.*, metals, PAHs) on the basis of sediment quality value exceedances. TIEs address this problem by indicating the contaminant group(s) most likely responsible for the observed responses. A properly conducted TIE will increase the confidence of the study conclusion by using multiple lines of evidence (*i.e.*, multiple treatments showing consistent indications of potential cause-effect), thereby reducing the chance of a spurious result.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- TIEs are typically conducted after (or concurrently with) other toxicity testing. Careful consideration of how to integrate sample collection for both toxicity testing and a TIE is required. For example, sufficient sample volumes need to be collected in advance if a synoptic TIE is contemplated for a sediment quality Triad.

- TIEs are most effective when applied to samples that exhibit pronounced toxic responses. TIEs are less useful when toxicity is minor to moderate in magnitude.

- In heterogeneous environments, multiple conclusions can be reached using the results of different samples. Care must be exercises in the extrapolation of TIE results to other portions of the site that may have different physical and chemical conditions. For example, the changes in redox potential, pH, salinity, and other geochemical parameters as groundwater discharges, or as freshwater mixes with saline water, can elicit pronounced changes in toxic potential of samples.

- TIEs operate in an iterative fashion where the results of one type of manipulation lead to other potential manipulations that should be examined. The scope of the TIE cannot often be predicted in advance, although there should be consensus regarding the desired level of identification (*i.e.*, do you need to know which specific divalent metal is causing the toxicity, or is it enough for site management purposes to know that a contaminant group is responsible? [*e.g.*, divalent metals, non-polar organics]). The tiered approach, although cost-efficient, can be problematic in practical terms because site managers often require certainty in project cost and timelines at the beginning of a project.

- The TIE needs to consider a substantially broader range of potential contaminants and factors than would normally be measured to meet CSR requirements. Non-listed contaminants or physical factors may also be contributing to the toxicity.

- TIEs often require substantial professional judgment in interpreting the multiple lines of evidence. The physical and chemical manipulations of samples can cause complex interactions in the bioavailability of different sample constituents. For example, purging of sediments to reduce the influence of volatiles can have the side-effect of increasing the bioavailability of metals. The TIE investigator needs to be aware of the influence of different manipulations, and interpretation can be complex where multiple stressors of concern are present.

- TIEs are most easily conducted on aqueous samples, and for this reason, sediment assessments often apply TIEs to porewater extracted from sediments. The investigator needs to be aware of the physicochemical implications of processing sediments to obtain porewater, and understand the ecological relevance of porewater toxicity testing to the receptors of concern.

- TIEs are conducted on individual samples using individual test organisms, and as such, represent a "snapshot" of cause-and-effect relationships. Seasonal (and other) variations are not considered; additionally, the toxic mode of action may vary for different organisms. Interpretation of TIE results under these circumstances as indicative of the overall ecological effects at a site is problematic. TIEs may need to be repeated using multiple test organisms and samples.

Where can I find additional information about this tool? The following TIE guidance manuals are available online:

- Carr R.S., M. Nipper, W.J. Adams, W.J. Berry, G.A. Burton Jr, K. Ho, D. MacDonald, R. Scroggins, and P.V. Winger. 2001. *Summary of a SETAC Technical Workshop: Porewater Toxicity Testing: Biological, Chemical, and Ecological Considerations with a Review of Methods and Applications, and Recommendations for Future Areas of Research*. Summary of the SETAC Workshop on Porewater Toxicity Testing: Biological, Chemical, and Ecological Considerations with a Review of Methods and Applications, and Recommendations for Future Areas of Research; 18–22 March 2000; Pensacola, FL. Society of Environmental Toxicology and Chemistry (SETAC). Pensacola, FL. 38 p. Available at: http://www.setac.org/htdocs/files/PWSummary.pdf

- USEPA (U.S. Environmental Protection Agency). 1991. *Method for Aquatic Toxicity Identification Evaluations, Phase I Toxicity Characterization Procedures*. EPA/600/6-91/003. Office of Research and Development, Washington DC. February 1991. Available at: http://www.epa.gov/npdes/pubs/owm0330.pdf

- USEPA. 1993. *Methods for Aquatic Toxicity Identification Evaluations: Phase II Toxicity Identification Procedures for Samples Exhibiting Acute and Chronic Toxicity*. U.S. Environmental Protection Agency, Office of Research and Development, Duluth, MN. EPA 600/R-92-080.

- USEPA. 1993. *Methods for Aquatic Toxicity Identification Evaluations: Phase III Toxicity Confirmation Procedures for Samples Exhibiting Acute and Chronic Toxicity*. U.S. Environmental Protection Agency, Office of Research and Development, Duluth, MN. EPA 600/R-92-081.

- USEPA. 1996. *Marine Toxicity Identification Evaluation (TIE), Phase I Guidance Document*. U.S. EPA, ORD, EPA/600/R-95/054.

- USEPA. 2001. *Clarifications Regarding Toxicity Reduction and Identification Evaluations in the National Pollutant Discharge Elimination System Program*. U.S. Environmental Protection Agency, Office of Wastewater Management, Office of Regulatory Enforcement, Washington, DC. March 27, 2001. Available at: http://www.epa.gov/npdes/pubs/owmfinaltretie.pdf

- USEPA. 2007. *Sediment Toxicity Identification Evaluation (TIE) Phases I, II, and III Guidance Document*. Edited by K.T. Ho and R.M. Burgess (U.S. Environmental Protection Agency, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, Narragansett, Rhode Island) and D.R. Mount, T.J. Norberg-King, and J.R. Hockett (U.S. Environmental Protection Agency, National Health and Environmental Effects Research Laboratory, Mid-Continent Ecology Division, Duluth, MN). EPA/600/R-07/080. September 2007. http://www.epa.gov/nheerl/publications/files/Sediment%20TIE%20Guidance%20Document.pdf

DIRECT MEASUREMENT TOOL #10

10.0 HISTOPATHOLOGY ASSESSMENTS

What does this tool consist of? Histopathology involves microscopic examination of organism tissues (*e.g.*, gonads; liver). It entails detailed examination of tissue in order to study the manifestations of disease and cellular damage (*e.g.*, lesions). This tool is most frequently applied to fish health assessments, and is frequently combined with biomarker approaches.

Which ecosystem(s) would this tool typically be applied in? Deep Aquatic, and Rivers and Streams ecosystems.

How frequently is this tool used in a DERA? Infrequently.

What are the benefits of using this tool in a DERA?

- Provides information on adverse effects that may be occurring in individual organisms at exposure concentrations lower than those that result in adverse effects on growth or reproduction. Provides information regarding the "health" of organisms.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- Cause and effect relationships may not be clear; organisms may also suffer from diseases from background or natural sources that cause similar histopathological alterations.

- Histopathology is highly specialized: sample collection, preparation and analysis require substantial expertise.

- Substantial numbers of samples may be required to achieve the necessary statistical power.

Where can I find additional information about this tool?

- AETE (Aquatic Effects Technology Evaluation. 1998. *Technical Evaluation of Histopathology as an Environmental Monitoring Tool for the Mining Industry in Canada*. Report 2.2.2. Available online: http://www.nrcan.gc.ca/mms/canmet-mtb/mmsl-lmsm/enviro/metals/aete.htm

- USEPA. 1987. Guidance for Conducting Fish Liver Histopathology Studies During 301(H) Monitoring. EPA 430/987/004. June 1987.

- Vethaak, A.D. 1992. Gross Pathology and Histopathology in Fish: Summary. Ministry of Transport and Public Works, Tidal Waters Division, Ecotoxicology Section, The Hague, Netherlands. *Mar. Ecol. Prog. Ser. 91:171-172*. Available at: http://www.int-res.com/articles/meps/91/m091p171.pdf

DIRECT MEASUREMENT TOOL #11

11.0 DEFORMITY ASSESSMENTS

What does this tool consist of? Deformity assessments entail visual inspection of organisms (usually larval fish or amphibians) from either chronic toxicity testing or from field sampling. The frequency and magnitude of deformities (*e.g.*, edema, ocular or skeletal malformation) are measured. Deformity assessments are usually limited to those DERAs involving compounds with a known tendency to cause deformity (*e.g.*, PCBs or pesticides for amphibians; selenium for larval fish). Deformity assessments are also conducted in mammalian studies, but are generally used in DERA as supporting lines of evidence rather than used directly in setting effects thresholds.

Which ecosystem(s) would this tool typically be applied in? Deep Aquatic or Rivers and Streams ecosystems. Deformities and disease are used in rapid bioassessment protocols to depict the health and condition of individual fish. Barbour *et al.* (1999) recommend reporting proportion of individuals with disease, tumors, fin damage, and skeletal anomalies as indicators of the subacute effects of chemical pollution and the aesthetic value of game and non-game fish.

How frequently is this tool used in a DERA? Occasional.

What are the benefits of using this tool in a DERA?

- Malformations may occur at concentrations that are lower than thresholds for reproductive or growth effects; it is potentially a more sensitive toxicological endpoint.

- In some instances, evaluation of deformities can be used to provide a more specific cause-effect linkage than is possible through evaluation of growth, reproduction, and development data. For example, research has shown that ingestion of coplanar PCBs by juvenile mink (kits) causes a specific lesion in the jaw mandible and maxilla that leads to loose and displaced teeth. Confirmation of these jaw lesions in experimental organisms exposed to PCBs has been used to confirm that observed reductions in laboratory mink kit survival are attributable to PCBs.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- Toxicological testing designs need to explicitly consider requirements of malformation endpoint data in order to properly address statistical power considerations. Adding malformation endpoints to existing chronic toxicity testing protocols is not appropriate.

- Malformation is expressed through a complex mode of toxic action involving the interaction of the contaminant with various stages of organism development. Timing of the exposure may be a significant confounding factor. For example, toxicity to amphibians (*e.g.*, ranid frogs) is often enhanced during tail resorption and other physiological changes that occur during metamorphosis from larval stage (tadpole). Accordingly, exposure conditions in a test must simulate the exposure pathway and timing relevant to the species of interest.

- Not all malformations are equal in terms of their ecological relevance. To date, few studies have explored the ecological relevance of malformation to larval organisms in terms of their population level impacts. Some malformations (*e.g.*, external swim bladder, severe scoliosis of spine) can be expected to cause mortality either directly or indirectly. Other malformations, such as fin rot, disfiguring lesions, or enlarged liver, have a more uncertain impact for the survival and reproduction of affected fish.

Where can I find additional information about this tool? Because the nature of deformities is highly species- and contaminant-specific, a literature search for the compound and organism of interest is recommended prior to planning or conducting a deformity assessment. The following references provide information on health assessment protocols for fish deformity analyses.

- Adams, S.M., A.M. Brown and R.W. Goede. 1993. A quantitative health assessment index for rapid evaluation of fish condition in the field. *Trans. Am. Fish Soc.* 122:63-73.

- Adams, S.M. and M.G. Ryon. 1994. A comparison of health assessment approaches for evaluating the effects of contaminant-related stress on fish populations. *J. Aquat. Ecosystem Health* 3:15-25.

- Adams, S.M., K.D. Ham, M.S. Greeley, R.F. LeHew, D.E. Hinton, and C.F. Saylor. 1996. Downstream gradients in bioindicator responses: point source contaminant effects on fish health. *Can. J. Fish. Aquat. Sci.* 53(10):2177–2187.

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- Goede, R.W. and B.A. Barton. 1990. Organismic indices and an autopsy-based assessment as indicators of health and condition of fish. *American Fisheries Society Symposium* 8: 93-108.

- Munkittrick, K. R. 1992. A review and evaluation of study design considerations for site-specifically assessing the health of fish populations. *J. Aquat. Ecosystem Health* 1: 283-293.

DIRECT MEASUREMENT TOOL #12

12.0 STABLE ISOTOPE ANALYSES

What does this tool consist of? Isotope analysis is the identification of isotopic signature, the distribution of certain stable isotopes and chemical elements within chemical compounds. Stable isotope analysis can be applied to a food web to make it possible to draw direct inferences regarding diet, trophic level, and subsistence.

Ratios of the stable isotopes of carbon (^{13}C *versus* ^{12}C) and nitrogen (^{15}N *versus* ^{14}N) can be used to infer feeding relationships, which can in turn be used to model the trophic transfer of contaminants. Stable isotope ratios are expressed in "delta" units (written $\delta^{13}C$ and $\delta^{15}N$). The combination of $\delta^{13}C$ and $\delta^{15}N$ is sometimes referred to as an organism's "stable isotope signature". $\delta^{13}C$ reflects the carbon source at the base of an organism's food web (*e.g.*, benthic algae *versus* phytoplankton, or a mixture of the two). $\delta^{13}C$ typically changes very little between diet and consumer (*i.e.*, "you are what you eat"). $\delta^{15}N$ reflects the organism's trophic level (TL), and tends to increase between diet and consumer. An organism's $\delta^{15}N$ must be interpreted relative to the $\delta^{15}N$ of the base of the food web, commonly by using clams or some other herbivore to provide a long-term average $\delta^{15}N$ for the basal resource:

$$TL = TL_{baseline} + (\delta^{15}N_{organism} - \delta^{15}N_{baseline})/3.4\ ‰$$

where TL_{clam} is the trophic level of the species used as a baseline (2.0 for clams or other herbivores, 1.0 for plants) and 3.4‰ is the average enrichment in $\delta^{15}N$ between diet and consumer (called "trophic fractionation").

Which ecosystem(s) would this tool typically be applied in? Stable isotope analysis can be applied to establish feeding links and diet compositions in the food web of any ecosystem type.

How frequently is this tool used in a DERA? Infrequent.

What are the benefits of using this tool in a DERA? Stable isotope analysis provides site-specific, time-integrated diet information for receptors, and can be invaluable in estimating the exposure of these receptors to COPCs via their diets. Trophic level, inferred from $\delta^{15}N$, is particularly important for estimating exposure to chemicals that biomagnify. Stable isotope analysis requires only a small amount of material (typically < 1 mg) and is inexpensive (~$10-20/sample).

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- Site-specific measurements are essential. There can be tremendous spatial variation in the stable isotope signatures of basal resources (phytoplankton, vascular plants, detritus), produced by local variation in nutrient sources, currents and mixing, growth rates, *etc*. This will produce spatial variation in stable isotope ratios of the animal species farther up the food chain. It is not appropriate to assume that the stable isotope signature of an organism is the same as that measured in other areas.

- Small and short-lived species can exhibit large temporal variation in stable isotopes, which can make it difficult to correctly interpret feeding relationships from a 'snapshot' study. This problem can be circumvented for phytoplankton by sampling large-bodied, long-lived herbivores (*e.g*., clams) and then inferring the mean phytoplankton signature from this. A similar approach can be used for zooplankton if strictly zooplanktivorous fish are available; otherwise, it is best to have repeated (*e.g*., seasonal) sampling to capture this temporal variability.

- There can be substantial variation in stable isotope signatures among tissues within an animal. For small animals that are consumed whole (*e.g.*, insects), it is appropriate to use a whole-body analysis. For larger animals (*e.g.*, fish), analysis is typically done on muscle tissue. If non-lethal sampling is desired, it is possible to use scales, hair or feathers, but it is then necessary to know how the stable isotope signature of this tissue relates to that of the animal's bulk muscle tissue (the edible part, and the main repository of nitrogen in animals).

Where can I find additional information about this tool?

- Cabana, G., and J.B. Rasmussen. 1994. Modeling food chain structure and contaminant bioaccumulation using stable nitrogen isotopes. *Nature* 372:255-257.

- Peterson, B. and B. Fry. 1987. Stable isotopes in ecosystem studies. *Annu. Rev. Ecol. Syst.* 18:293–320.

- Post, D.M. 2002. Using stable isotopes to estimate trophic position: models, methods, and assumptions. *Ecology* 83:703-718.

- Vander Zanden, M. J. and J. B. Rasmussen. 1999. Primary consumer δ13C and δ15N and the trophic position of aquatic consumers. *Ecology* 80:1395–1404.

DIRECT MEASUREMENT TOOL #13

13.0 BIOMARKER STUDIES

What does this tool consist of? Biomarkers are measurable biological or biochemical parameters that change in response to xenobiotic exposure and other environmental or physiological stressors, and can be indices of toxicant exposure or effects. Examples include bile fluorescent aromatic compounds (FACs), liver enzyme induction (EROD, CYP1A), hematological parameters, and steroid hormone levels. The use of biomarkers is often combined with histopathology assessment.

Which ecosystem(s) would this tool typically be applied in? Any.

How frequently is this tool used in a DERA? Rare. Biomarkers are often used as supporting lines of evidence, but are rarely used to establish effects thresholds in DERAs.

What are the benefits of using this tool in a DERA? If the biomarkers are sufficiently specific and well characterized, they can provide meaningful data for the risk assessment process by providing an indication of the degree of exposure of humans or animals in natural populations to a specific xenobiotic or class of xenobiotics.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- Most biomarkers are effective as indices of exposure, but adequate information is rarely available on the underlying dose-response curves. Biomarkers are rarely useful in providing information about effects.

- Biomarkers tend to measure changes in sub-organism parameters (*e.g.*, biochemistry; enzyme activity) that do not necessarily translate into a relevant endpoint for DERA purposes (*e.g.*, organism-level endpoint such as survival, growth, deformity and reproduction).

- The degree of a change in a biomarker parameter can be influenced by multiple endogenous factors (*e.g.*, age) and exogenous factors (*e.g.*, chemical exposures). Many biomarkers respond to multiple COPCs or groups of COPCs (*e.g.*, CYP1A responds to multiple types of chemicals), making it difficult to correlate the degree of change in the biomarker with COPC exposure. In general, most biomarkers are not sufficiently specific for DERA purposes.

- Caution is urged with respect to utilization of biomarkers in the risk assessment process until more complete documentation is available on the specificity, sensitivity, and time course of changes, and on the impact of multiple exposures or the time of exposures (Chambers *et al*. 2004).

Where can I find additional information about this tool?

- McCarty, L.S., M. Power and K.R. Munkittrick. 2004. Bioindicators versus biomarkers in ecological risk assessment. *Hum. Ecol. Risk Assess.* 8:159-164.

- Chambers, J.E.; J.S. Boone, R.L. Carr, H.W. Chambers and D.L. Straus. 2004. Biomarkers as predictors in health and ecological risk assessment. *Hum. Ecol. Risk Assess.* 8:165-176.

- Fossi, M.C. 1994. Nondestructive biomarkers in ecotoxicology. *Environ. Health Perspect.* 102 (S12):49-54.

DIRECT MEASUREMENT TOOL #14

14.0 BENTHIC COMMUNITY SURVEYS

What does this tool consist of? Benthic community surveys entail taxonomic identification and enumeration of benthic organisms collected using standardized sampling techniques. Diversity, abundance, and multiple other indices (individual and multivariate) can be calculated from this data. General types of indices include:

- Individual metrics – Diversity, abundance, richness, dominance, evenness, Modified Hilsenhoff Biotic Index (HBI), *etc.*

- Combined metrics – Combined abundance of EPT taxa (mayflies, caddisflies, and stoneflies), multivariate profiling (*e.g.*, non-metric multidimensional scaling) of assemblages, cluster analyses, *etc.*;

- Biological Response Indices – Rapid Bioassessment Protocols (Plafkin *et al.*, 1989) integrate numerous lines of evidence in the evaluation of benthic community heath. The Benthic Index of Biotic Integrity (B-IBI) defines expected conditions at reference sites relatively free of anthropogenic stress, and then assigns categorical values for various descriptive metrics by comparison with observations at these reference sites. Metrics considered in biological response indices include different aspects of stream biology, including taxonomic richness and composition, tolerance and intolerance, habit, reproductive strategy, feeding ecology, and population structure.

Which ecosystem(s) would this tool typically be applied in? Deep Aquatic and Rivers and Streams

How frequently is this tool used in a DERA? Common. This tool is part of the Sediment Quality Triad approach to sediment risk assessment. Numerous sampling and evaluation protocols exist for this tool.

What are the benefits of using this tool in a DERA?

- The tool provides a direct measurement of potential long-term toxicant-related effects under actual field conditions. Long life cycles integrate effects of short-term or intermittent impacts, as well as long-term perturbations. Sensitive life stages of individuals respond quickly to stress, with the effect on the entire community occurring over longer timeframes.

- The methods of sample collection, processing, enumeration, and data analysis are well standardized. Sampling can be performed using simple, inexpensive equipment and a small field crew.

- Benthic macroinvertebrates are abundant in most streams and large water bodies, and the relatively sedentary nature of benthic invertebrates allows for quantifiable spatial analysis of environmental effects.

- Taxonomy of many invertebrate groups is well known and numerous identification keys are available.

- Responses of many common species to different types of pollution have been established (*i.e.*, organisms have been classified based on their level of "pollution sensitivity").

- Benthic macroinvertebrates constitute a primary food source for recreationally and commercially important fish species that are protected under the federal Fisheries Act.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- Benthic community data are sensitive to habitat alteration and other physical factors not related to the COPCs. If there is sufficient stream velocity, the tendency for individual organisms to drift downstream may offset advantages of being sedentary.

- Micro-scale variation in contaminant distribution can result in substantial variation in benthic community data. Synoptic sampling for chemistry and benthic community data is essential, but even synoptic sampling will not address natural stochasticity in resident biological communities.

- Statistical power is often limited when single benthic community replicates are collected from each station. Multiple replicates are recommended to decrease uncertainty and improve statistical power. Quantitative sampling can require large numbers of samples, with increased costs (attributed to taxonomic identification).

- Specific benthic community metrics are the subject of significant debate in the scientific community. Rather than emphasize a single benthic community metric, it is recommended that practitioners evaluate the data using multiple approaches.

- Seasonal variation in community structure may complicate interpretations or comparisons.

Where can I find additional information about this tool?

- Environment Canada provides detailed guidance for the use of benthic community surveys for environmental effects monitoring programs (EEM) for pulp and paper mills and metal mines. Technical documents that summarize the guidance are available at: http://www.ec.gc.ca/eem/english/Publications/default.cfm

- Barbour, M.T., J.B. Stribling, and J.R. Karr. 1995. The multimetric approach for establishing biocriteria and measuring biological condition. Pages 63-76 in W. S. Davis and T. P. Simon (editors). *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making*. Lewis Publishers, Ann Arbor, Michigan.

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- Beatty, J.M., L.E. McDonald, and F.M. Westcott. 2003. *Guidelines for sampling benthic invertebrates in British Columbia streams*. Prepared for BC Ministry of Land, Water and Air Protection.

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- Bode, R.W. and M.A. Novak. 1995. Development and application of biological impairment criteria for rivers and streams in New York State. Pages 97-107 in W. S. Davis and T. P. Simon (editors). *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making*. Lewis Publishers, Ann Arbor, Michigan.

- EVS (EVS Environment Consultants). 2003. GVRD Benthic Macroinvertebrate B-IBI Guide. Prepared for the Greater Vancouver Regional District, Burnaby, BC by EVS Environment Consultants, North Vancouver, BC. Available at: http://www.gvrd.bc.ca/sewerage/pdf/bib_guide.pdf

- EVS (EVS Environment Consultants). 1992. *Guidelines for Monitoring Benthos in Freshwater Environments*. Final Report. Prepared for Environment Canada, North Vancouver, BC. 81 pp.

- Gibson, G.A., M.T. Barbour, J.B. Stribling, J. Gerritsen, and J.R. Karr. 1996. *Biological criteria: Technical guidance for streams and rivers*. EPA/822-B-94-001. U. S. Environmental Protection Agency, Office of Science and Technology, Washington, DC.

- Karr, J.R and E.W. Chu. 1997. *Biological monitoring and assessment: using multimetric indices effectively*. EPA 235-R97-001. Seattle: University of Washington.

- Kleindl, W.J. 1998. *A Benthic Index of Biotic Integrity for Puget Sound Lowland Streams, Washington, USA*. Masters of Science thesis, University of Washington, Seattle, WA.

- Plafkin, J.L, M.T. Barbour, K.D. Porter, S.K. Gross, and R.M. Hughes. 1989. *Rapid Bioassessment Protocols for Use in Streams and Rivers: Benthic Macroinvertebrates and Fish*. EPA/440/4-89-001. U.S. Environmental Protection Agency, Office of Water, Washington, DC. Available at: http://www.epa.gov/owow/monitoring/rbp

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- Rosenberg, D.M., I.J. Davies, D.G. Cobb, and A.P. Wiens. 2001. *Protocols for Measuring Biodiversity: Benthic macroinvertebrates in Freshwaters*. Environment Canada Ecological Monitoring and Assessment Network. Freshwater Institute, Winnipeg MB. May 2, 2001. Available at: http://www.eman-rese.ca/eman/ecotools/protocols/freshwater/benthics/intro.html

- Rosenberg, D.M. and V.H. Resh. 1993. Introduction to freshwater biomonitoring and benthic macroinvertebrates. pp. 1-9 In Rosenberg, D.M. and V.H. Resh (Eds.), *Freshwater Biomonitoring and Benthic Macroinvertebrates*. Chapman and Hall, New York.

- Stribling, J.B., S.K. Moulton III, and G.I. Lester. 2003. Determining the quality of taxonomic data. *Journal of the North American Benthological Society* 23(4):621-631.

- U.S. EPA. 1998. *Lake and Reservoir Bioassessment and Biocriteria: Technical Guidance Document*. EPA 841-B-98-007. EPA Office of Water, Washington, DC.

DIRECT MEASUREMENT TOOL #15

15.0 INTERTIDAL COMMUNITY SURVEYS

15.1 Overview

What does this tool consist of? The intertidal flora and fauna of rocky shorelines typically consist of macroalgae and encrusting invertebrates, as well as motile macroinvertebrates and fish. Intertidal community surveys can be useful indicators of the effects of contaminants, and can complement other lines of evidence (*e.g.*, toxicity tests) in the risk assessment process. Macroalgae and encrusting benthic organisms are particularly useful in this regard because:

- They are sessile (*i.e.*, fixed in place), and therefore exposed to any COPCs that are present in groundwater seepage along the shoreline;

- They colonize a range of hard substrates, including sites heavily disturbed by human activities; and,

- Localized impacts of COPCs on intertidal communities may be readily apparent through shifts in community structure.

In short, intertidal community surveys can be used in situations where hard, intertidal substrates are present, often adjacent to upland properties that are the sources of contamination. The resulting data and information (which can range from quantitative to semi-quantitative) can be used, in conjunction with other lines of evidence, to assess the effects of contaminant exposure on intertidal communities. To the extent possible and as merited based on the site conditions and management objectives, such studies should be as quantitative as possible.

Which ecosystem(s) would this tool typically be applied in? Shoreline (marine or estuarine).

How frequently is this tool used in a DERA? Fairly common for risk assessments on properties with hard substrates and adjacent to marine systems.

What are the benefits of using this tool in a DERA?

- The tool provides direct measurement of toxicant-related effects under actual field conditions.

- Using consistent survey methods, sampling can identify effects of chemical exposure to hard bottom substrate communities.

- Intertidal invertebrates are abundant in their habitats, and their relatively sedentary nature of allows for spatial analysis of environmental effects.

- Taxonomy of many invertebrate groups is well known and numerous identification keys are available.

- Intertidal communities constitute a food source for recreationally and commercially important fish species that are protected under the federal Fisheries Act.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- The often heterogeneous nature of intertidal habitats, particularly with hard substrates, can challenge the sampling design of such studies. Factors such as slope, orientation, substrate type, and wave action can vary across a site, complicating interpretation of survey data. This issue is discussed further in Section I-15.3.

- Achieving statistical power can be challenging and alternative interpretation approaches may be required (see Section I-15.5).

- It can be very challenging to establish reference or background conditions (alternatives are described in Section I-15.4).

- The details of the sampling design will vary from site to site depending on the nature of the site and the assessment endpoints of the risk assessment. Section I-15.5 presents a strawman study design that would be useful for many DERAs; however, individual cases may call for alternative study designs. Such is acceptable provided that rationales are provided for the connection between the "how" (measurement endpoints and methodological details) and the "why" (assessment endpoints and risk hypotheses).

- As with all direct assessments of biological communities, natural variability and stochasticity, combined with incertitude regarding ecological processes, will limit the extent to which results may be interpreted in a rigorous quantitative fashion. Not all known physical factors can be controlled in a practical study design, and unknown factors (ecological uncertainties) will persist. Investigators should be realistic about the level of quantitative precision and/or certainty that can be expected from studies of this type.

15.2 Relevant Intertidal Organism Types

Intertidal organisms can be grouped into broad categories. Depending on the details of the sampling design, it may be necessary to identify organisms to a higher level of detail than the categories below; however, these represent the primary biota of interest in an intertidal survey. Descriptions were derived from BC WLAP (2002), Bates (2007), BC ILMB (2008), and other scientific references. Additional references useful for identification of biota are summarized in Section I-15.6.

- *Barnacles* – Barnacles are small filter feeding crustaceans that feed on plankton in the water around them. In British Columbia, common barnacle species include the familiar acorn barnacle (*Balanus glandula*), the little brown barnacle (*Chthamalus dalli*), the gooseneck barnacle (*Pollicipes polymerus*) and the thatched barnacle (*Semibalanus cariosus*). They are common in areas subject to high wave energy.

- *Mussels* – Mussels are shellfish (mollusks) with two wing-shaped shells hinged together. They are filter feeders that strain their food, mostly small algae, from the water. An example of a common mussel genus in British Columbia is the blue mussel (*Mytilus* spp.), particularly in the southern Strait of Georgia, where the influence of the Fraser River plume is strong on the rocky intertidal community.

- *Oysters* – Oysters are filter-feeding bivalves that live in the intertidal and subtidal habitats. Oyster beds provide habitat for hundreds of animals such as anemones, barnacles, and hooked mussels. The introduced Japanese oyster (*Crassostrea gigas*) is found in the Strait of Georgia and is known to complete with blue mussels and native British Columbia oysters (*Ostrea conchaphila*).

- *Gastropods* – Snails, limpets, and dogwhelks are gastropods that are related to mussels (*i.e.*, mollusks) but they do not have hinged shells; rather they typically have a single coiled shell composed of calcium carbonate. Their primary food consists of algae rasped off of rock or sand, but a few species are scavengers or predators. Periwinkle snails (*Littorina sitkana* and *Littorina scutulata*) and the moon snail (*Pollinices lewisii*) are common along British Columbia shorelines, often inhabiting crevices under rocks and seaweeds. Limpets possess small, hat-shaped shells attached to rocks and other hard surfaces. Common limpet species in British Columbia include the ribbed limpet *Lottia digitalis*, the mask limpet *Tectura persona*, and the shield limpet *Tectura scutum*. Other intertidal gastropods in BC include the dog whelk (*Nucella lamellosa*) that inhabits tide pools and rock crevices in the intertidal zone.

- *Sea Anemones* - Anemones are stinging animals belonging to the phylum Cnidaria, They have sticky and poisonous tentacles that paralyze their prey, and bring their prey towards their mouth. Anemones have no shell, and appear flower-like as they are built with a trunk-like body with petal-like tentacles. Many species of sea anemones inhabit rocky shores, especially where there are tide pools in which they can remain submerged when the tide goes out. The giant green anemone *Anthopleura xanthogrammica* is an example of an anemone found in the intertidal zone within British Columbia. Clusters of pink-tipped anemones in BC are often aggregate anemones (*Anthopleura elegantissima*).

- *Sea Stars* – Starfish (and brittle stars) are common on rocky shores in the water just below the tide lines. They have five arms (or multiples of five arms) that radiate out from a central disc that houses the mouth. Sea stars feed by using their arms to force open shellfish, pushing their stomach out their mouth, and digest the meat of the organism. The ochre sea star *Pisaster ochraceus* and the sunflower star *Pycnopodia helianthoides* are examples of common sea star species found in the intertidal zone within British Columbia.

- *Sea Urchins* – Sea urchins are grazers that feed on seaweed and small animals. In the intertidal zone, they can be found in wetted areas in great numbers. They are puffed, hard shelled creatures with rigid spines that forage for dead and decaying matter, algae, sponges, mussels, and barnacles. The green sea urchin *Strongylocentrotus droebachiensis* is a common species in sheltered, rocky shores and tide pools within British Columbia.

- *Crabs* – Crabs are crustaceans, have jointed appendages, no backbone and a chitonous exoskeleton. Although there are nearly 100 crab species in British Columbia, the two species most common are the Dungeness crab *Cancer magister* and red rock crab *Cancer productus*. The red rock crab is a small dark red crab found on rocky shores, whereas Dungeness crabs prefer sandy or gravel environments. Dungeness crabs juveniles remain in intertidal and shallow subtidal areas hiding beneath or among plants, rocks and shell debris. Other important intertidal crab species include the hermit crab (*Pagurus* spp.), shore crabs (*Hemigrapsu*s spp.), northern kelp crab (*Pugettia producta*) and the decorator crab (*Oregonia gracillis*).

- *Isopods* – Intertidal isopods are scavengers related to their land-dwelling cousins, the sow bugs. Intertidal species in British Columbia include the rockweed isopod (*Idotea wosnesenskii*), the Oregon pill bug (Gnorimosphaeroma oregonensis), and the gribble (*Limnoria lignorum*).

- *Chitons* – This soft-bodied animal is protected by tough, overlapping plates, and moves via a large, undulating foot.

- *Tidepool Fish* – Bullheads or tide pool sculpins (*Oligocottus maculosus*) are abundant near rocky shores. In rocky habitats, eel-like high cockscomb (*Anoplarchus purpurescens*) specimens are also found.

- *Polychaetes* – Polychaetes are a diverse group of annelids that can be found in various habitats, including underneath cobbles, attached to rocks in calcareous tubes, in burrows or free-living. Some are predators, while others are detritivores or planktivores.

- *Sponges* - A few species are found in intertidal rocky habitat in BC, mostly along the lower intertidal zone in protected and shaded areas. These filter feeders are predated by some nudibranchs and limpets. Some sponges have been used as bioindicators of metal contamination.

- *Vascular Plants and Algae* – Eelgrass (*Zostera* spp.) is a flowering plant adapted to living in saltwater and is one of the few vascular plants found in the intertidal zone. Eelgrass beds are highly valued as nursery areas for fish. Surfgrass (*Phyllospadix* spp.) is often found attached to immobile rock substrate and at higher wave exposures relative to eelgrass. The majority of intertidal "plant" species are macroalgae. Common British Columbia macroalgae include the following taxa (organized broadly into green, brown, and red algae):

  - sea lettuce (*Ulva* spp.) – thin and fast growing foliose green algae often found in areas with elevated nutrients and low to intermediate wave exposures. At least four species of *Ulva* are found in BC. It is eaten by a wide diversity of species, including geese, chitons, snails, polychaetes, crabs, echinoderms and fish;

  - *Enteromorpha* – tube-like green algae closely related to *Ulva*, the *Enteromorpha* genus contains seven species in the Northeast Pacific. It can tolerate a wide range of salinity, and is often seen at freshwater seeps, even above the high tide line;

  - *Cladophora* – *Cladophora* is a green alga genus represented by 8 species in the Northeast Pacific, with appearing as tufts of moss in the intertidal;

  - giant kelp (*Macrocystis integrifolia*) - a large, leafy brown algae, known as seaweed, that grows along colder coastlines;

  - bull kelp (*Nereocystis luetkeana*) – a bladed kelp (brown algae) that beds on rocks;

  - rockweed (*Fucus* spp.) – a bushy clump (brown algae) with floating air bladders;

- nori (*Porphyra* spp.)- purplish-brown seaweed in the red algae category; and,

- Turkish towel – *Mastocarpus papillatus* is a small red encrusting alga with a bifurcating dark brown blade.

In addition to invertebrates, plants, and algae, intertidal communities provide habitat for numerous bird or mammal species that may prey on the organisms listed above. Rocky coastal habitats in British Columbia include numerous bird species including loons, grebes, cormorants, herons, swans, geese, ducks, shorebirds, gulls, terns, auks, oystercatchers, pigeon guillemots, and raptors. Marine mammals such as harbour seals, California and Steller's sea lions also frequent these habitats, but use intertidal mainly for resting rather than feeding. Harbour seals (*Phoca vitulina*) are typically seen in small groups resting on tidal reefs, boulders, and sandbars. The Steller sea lions *Eumetopias jubatus* congregate at rookeries in Cape St. James, North Danger Rocks and on the Scott Islands, and disperse locally along the coast to numerous wintering sites. The California sea lion *Zalophus californianus* winters off southern Vancouver Island, and feeds mainly upon mid-water schooling fishes such as hake, herring, dogfish, and salmon. The northern elephant seal (*Mirounga angustirostris*) is the largest pinniped species inhabiting the Northern Hemisphere; they occasionally come ashore to rest along rocky coastlines in BC. Procedures for identification of these species at specific sites are discussed in Appendix I-18.

15.3 General Considerations for Intertidal Studies

Care should be exercised in conducting an intertidal community surveys because the composition and health of a site's resident intertidal community is influenced by a wide range of interacting factors. The most important of these include:

- ecological factors;
- phenology; and,
- pollution tolerance.

These broad factors are discussed in turn below, with discussion of the implications for conducting intertidal surveys to support ecological risk assessment.

Ecological Factors – The composition and abundance of organisms in an intertidal community is influenced by biotic and abiotic factors, and the communities on two superficially similar sites may be very different because of subtle differences in these factors. From a risk assessment perspective, the main objective is to discriminate between factors attributable to natural conditions and those attributable to human alterations. In addition, identification of human influences related to physical disruption of the site must also be distinguished from the primary stressors of concern in the risk assessment

(usually contaminants). Biotic influences may be intraspecific or interspecific. Intraspecific relations exist between individuals of the same species within a population. For example, crowding may limit the number of mussels or barnacles that can colonize a given rock surface; this is a type of density-dependence. In contrast, interspecific relations are among different species, and include interspecific competition, predation, parasitism, and disease. For example, starfish predation may dramatically effect mussel distribution on hard substrates. Abiotic influences include chemical and physical factors (*e.g.*, depth, sunlight, climate, wave exposure, shore drainage). Anthropogenic activities (*e.g.*, shoreline modification or stabilization, propeller wash, pier construction) also cause abiotic perturbations.

Phenology – The "growing season" is the portion of the year during which conditions are favorable for growth. In the temperate waters of coastal British Columbia, intertidal growth occurs primarily between May and October. The timing of a site visit in relation to the growing season is an important consideration when interpreting the results of an intertidal survey, as certain types of macrophytes (*e.g.*, eelgrass [*Zostera* spp.]) die back at the end of the season. This is not to say that an intertidal survey completed outside this time window has no risk assessment value; rather, the emphasis should be on considering phenology in data interpretation, and targeting the growing season if there is flexibility in the risk assessment scheduling.

Tolerance to pollution – Different types of organisms may have very different tolerances to organic and inorganic contaminants. For example, it is not unusual to find mussels (*Mytilus* spp.), barnacles (*Balanus* spp.), shore crabs (*Hemigrapsus* spp,) and amphipods on relatively polluted shorelines, whereas sea anemones, tube worms, and nudibranchs are usually found at relatively uncontaminated sites. Further discussion of sensitivity is found in Section I-15.4, below.

In designing an intertidal survey, the implications of these three broad factors, and their potential interaction, should be carefully considered.

15.4 Specific Considerations for Intertidal Studies

Field study design requires a number of specific considerations, including toxonomic resolution, identification of control/comparison sites, seasonal effects and pollution sensitivity.

Degree of taxonomic resolution – In conducting intertidal community assessments, an investigator will encounter a wide array of species. Unlike benthic community assessments in subtidal environments, it is not common practice to evaluate intertidal taxa to the lowest possible level of taxonomic resolution (*i.e.*, species level). Part of the reason for this is that the sessile nature of intertidal flora and fauna precludes the

sampling and transport of the organisms to a laboratory setting. Instead, it is useful and defensible to conduct an evaluation at an intermediate level of classification; for intertidal communities is the lowest practical level determinable in the field by a qualified and experienced biologist. For some organisms that are readily identifiable, the lowest practical level will likely include some species-level determinations. However, other taxonomic groups (*e.g.*, *Ulva* spp., Polychaeta) are more difficult to speciate in the field, and an intermediate level of classification is acceptable for these cases. The guiding principles for taxonomic identifications are:

- An experienced and qualified biologist should be able to directly assess taxonomy in the field (*i.e.*, microscope evaluations should not be necessary). Accordingly, the lowest practical level for field assessments of intertidal fauna is somewhat less detailed than for subtidal assessments.

- Where detailed determinations are known, they should be documented, such that information is not lost during data logging and transcription.

- Provided the ecological role is understood for risk assessment purposes, it acceptable to evaluate some taxa at an intermediate level of classification (family or genus level).

- Practitioners should maintain a photographic record of each transect or quadrat, with close-ups of representative species kept for identification, in case more detailed information or confirmation is required at a later date.

- The risk assessment should provide a technically-defensible rationale for the approach used in any given study.

Vertical Zonation – Most aquatic community assessments are sensitive to the depth (relative to sea level) at which organisms and communities occur. Such is particularly the case for intertidal assessments, as the communities are sensitively dependent on the physical conditions (wave action, proportion of time wetted, salinity, temperature, *etc.*) that are a function of elevation. Accordingly, the intertidal area or littoral area is divided into vertical zones including the spray zone, high tide zone, middle tide zone, and low tide zone, each of which has a characteristic community (Figure I-15-1). In designing an intertidal community assessment, it is essential that comparisons be made across similar elevations, such that observed differences are not incorrectly attributed to contaminant influences. In practice, sampling is often conducted along transects perpendicular to the shoreline, with sampling at prescribed elevations, and elevation is considered in the data assessment and interpretation phases.

Reference Conditions – Due to the large number of non-contaminant factors that govern the distribution of intertidal organisms, it is preferable (but not mandatory) to obtain reference or background information against which to evaluate the potentially affected community. A suitable reference location should include consideration of physical and biological factors, and attempt to match these as closely as possible to the exposed condition at the site. Because of the high sensitivity of intertidal assemblages on light, aspect, slope, and wave conditions, and on the ecological characteristics in which the community is located, it important to assess the degree to which the reference condition may be different from exposed conditions (for reasons unrelated to site contamination).

Adjacent or Background Conditions – Where comparison of exposed conditions to a pristine reference condition is not possible, an alternative approach is to evaluate the community at a site relative to adjacent lands (*i.e.*, lateral to the site along the foreshore). In applying such an approach, the investigator must consider: (1) the potential for differences in human-induced modification at all locations; (2) the potential for site contamination to have affected the foreshore of adjacent lands via contaminant transport; and (3) the comparability of physical features such as substrate type. An example of this approach was applied to the assessment of acid-rock drainage near Britannia Creek, British Columbia (Marsden *et al.*, 2002; Zis *et al.* 2004); the surrounding intertidal zone was found to be devoid of rockweed, *Fucus gardneri*, a seaweed that dominates nearby shores.

Contamination Gradient – Where comparison of exposed conditions to a reference condition (either local background or pristine) is not possible, an alternative approach is to sample size representing multiple levels of exposure to contaminants. This approach is best suited to sites that have a wide range of exposure conditions that can be clearly delineated. If the range of exposures is narrow, the stochasticity and other variations of intertidal communities are more likely to overwhelm an effect of site contamination.

Seasonal Influences – In the design and interpretation of intertidal sampling data, it is important to standardize the data for seasonal influences. The simplest way to do this is to sample reference and exposed conditions near-simultaneously. If data are collected over multiple seasons, the potential influence of seasonal or ephemeral factors should be recognized explicitly. For example, blooms of the blade and filamentous green seaweed classified as sea lettuce (*Ulva* spp.) tend to be variable, both between seasons and within the summer growing season. Algae often show signs of natural decay after the spring bloom, and investigators should consider such transient effects in the interpretation of site data.

Contamination/Pollution Sensitivity – Whereas intertidal assessments should emphasize the broad composition of communities, there are some individual taxa and/or specific indicators that can be assessed as potential indicators of contamination (*i.e.*, sentinel species):

- Green filamentous algae dominance - *Enteromorpha* is a genus of green filamentous algae that can be found to depths of five metres in shallow brackish areas, or marine habitats in close proximity to freshwater seeps. *Enteromorpha* species are frequently found in areas influenced by municipal or industrial waste discharges. *Enteromorpha*, as well as some *Ulva* species, develop abundantly in zones directly affected by pollution, even as the abundance of other genera decreases (Żbikowski *et al.*, 2007; Marsden and DeWreede, 2000). In areas affected by pollutive discharge, *Enteromorpha* become a highly successful fouling organism (Castilla, 1996).

- Dog whelk presence and condition – The dog whelk, *Nucella lapillus*, is a species of predatory sea snail; a carnivorous marine rocky shore gastropod mollusk in the family Muricidae, the rock snails. Numerous studies have shown sensitivity of this species to the reproductive effects of tributyltin (Skarphedinsdottir *et al.*, 1996) which include imposex (*i.e.*, male sex characteristics form in female gastropods).

- Rockweed presence – Some intertidal macroalgae, such as *Fucus*, may be reduced in abundance or absent in contaminated areas. Acid mine drainage can result in the intertidal zone being devoid of rockweed, *Fucus gardneri*, a macroalga that dominates nearby shores (Marsden *et al.*, 2002).

Health of Organisms – The investigator should record whether dead or dying encrusting organisms are present. Barnacles and mussels are common, conspicuous encrusting organisms that form distinctive bands in the intertidal zone of temperate rocky habitats. Because the barnacle and mussel shells are anchored to the substrate, they can remain in place long after the organism inside has died. If areas of dead barnacles or mussels are observed at a site, this could be indicative of the presence of a COPC. Green and brown macroalgae can also be evaluated for clear signs of organism health or survival. Figures I-15-2 through I-15-5 indicate how site observations can be used to make determinations of algal health. To interpret the findings, it is important to look for broad patterns rather than individual observations of alga. The influence of seasonal life-cycles should also be considered when assessing the apparent health of organisms (*e.g.*, consider possibility of algal decay following spring bloom).

Bacterial Mats – The investigator should record whether bacterial mats are present. A bacterial mat is a layer of bacteria, sufficiently thick to be visible to the naked eye, that typically forms in environments where other organisms are unable to thrive. For example,

surface mats of *Beggiatoa* spp. have been observed in areas with low oxygen levels in the water column and high sediment sulfide concentrations. The presence of these mats has also been identified as an indicator of organic enrichment from either natural or anthropogenic sources (fish net pens and wood waste deposits) (Elliott *et al.*, 2006).

Anthropogenic Disturbances – In rocky intertidal communities, macroalgae grow best on stable, southward-facing shorelines that are sheltered from excessive wave action. Anthropogenic disturbances include shading from piers and other above-water structures, shoreline erosion, and propeller wash from ships. The age of rip-rap should be considered to determine whether timing of rip-rap placement may have influenced community structure. Community diversity may be reduced in areas with these influences, and these factors should be considered during study design and data interpretation.

15.5 Suggested Monitoring Methods

The detailed designs of intertidal surveys will vary depending on the assessment of the general and specific considerations described above. However, there are some common procedures that can be incorporated into a sampling design to maximize consistency across sites. This section describes a generic study framework that can be applied as a logical starting point for intertidal assessments. The practitioner should always evaluate the site conditions and study objectives to evaluate whether additional study components or revisions are warranted. Finally, there are a number of references in Section I-15.7 that may be used to develop or refine an intertidal sampling methodology.

Intertidal community monitoring should be conducted at or near "low tide". In British Columbia, coastal tides are semidiurnal (two high waters and two low waters each day). In addition, the elevation (relative to datum) of the lowest tide varies on a seasonal basis, and occurs at different times of day throughout the year. To strike a compromise between the need for maximum exposed shoreline and the need for practicality and repeatability of sampling, it is recommended that intertidal community monitoring be conducted in conditions close to the typical or average daily low tide, rather than the lowest low tide. The sampling design should acknowledge the limited time window available for low tide sampling. If a contaminant discharge plume is known or suspected to daylight at a specific elevation range, the sampling program should emphasize those elevations.

A target evaluation area is selected and divided into sections (*e.g.*, transects, quadrants), with attention to shore zonation so that representative coverage is obtained from each zone. A square-meter quadrat is randomly placed in each of the sections, maintaining consistent elevation relative to datum at all sections. The evaluation area should be consciously selected to allow for repeatability, but the quadrat sample positioning is random (subject to the requirement to control for elevation). Some organisms are

relatively rare or in particular habitats (*e.g.* sponges on sheltered vertical rock faces) so can be easily missed by a random quadrat, or only picked up at one site whereas the distribution is similar at all sites. These species should be noted in the data analysis.

For each quadrat, all taxa present are recorded on a separate data sheet. Abundance estimates are used for some common and often prolific species such as snails. An actual count is taken for other invertebrates, such as the crabs, sea stars and sea urchins. Sessile organisms (seaweeds, mussels, barnacles) are assigned percent coverage estimates, while mobile invertebrates are either counted or assigned abundance estimates. The field assessor should gently turn over medium-sized rocks to determine if the underside provides significant habitat, and replace, taking care to not harm the organisms.

For a rigorous study, monthly sampling during spring, summer, and fall is a good target. Monitoring during the winter months is often not possible because of weather conditions, and is less desirable than sampling during the growing season (and shoulder seasons) due to organism phenology considerations. Monitoring in teams of two to four people is recommended, with results validated or averaged among investigators. The safety of the monitors is always of paramount importance, and people should never put themselves in danger to collect data. This method may be used for tide pools with one data sheet per monitoring session.

The following methods were adapted from SSCW (2008) and supplemented with concepts from other technical references (see Section I-15.7). Additional detailed guidance, documentation forms, and technical references are available in Desrochers *et al.* (2006).

Sampling Layout

1. Establish a fixed evaluation area between the mid and low tide lines by locating the largest and best suitable area of rocky or cobble habitat in a particular region. The ideal location to conduct a survey is a rocky intertidal bench that is at least 30 meters wide (along the shore) and gently sloping from the high zone to low. It is important to select an area where the composition of the community is representative of the site (Jamieson *et al.*, 1999; University of California Santa Cruz, 2008). If possible link the survey with sampling sites used for chemical or biological analysis, *e.g.* mussel tissue contamination, sediment chemistry, *etc.*

2. Consult tide charts (http://www.waterlevels.gc.ca) to determine the time of low tide.

3. Divide the area into four quadrants by placing a metre tape parallel to the water's edge for a distance of 20 m and perpendicular to water line for 20 m. The two tapes must cross each other at the mid-intersection, in this case, at 10 m. The ideal set-up is 20 m × 20 meters.

4. At the initial setup, map the area and key landmarks for repeatability. Ascertain the site coordinates using a GPS or from a topographic map. Identify a benchmark for vertical control.

Data Collection

1. Return to your chosen evaluation site within one to two hours of low tide.

2. Record the names of all monitors present, date, time, site name, time of low tide, weather conditions, and air and water temperature.

3. Choose a quadrant near the low water edge to begin sampling.

4. Place a 1 m^2 quadrat on the substrate and take photos of each quadrat.

5. If the low tide zone contains sediment in rock interstices, it is useful to overturn the top 15 cm to note the presence of infaunal organisms, presence of sulfide odour and the composition of the substrate.

Data Analysis

1. Look for broad spatial patterns before conducting detailed exploratory statistics.

2. Consider factoring out or controlling for background/reference/adjacent conditions.

3. Consider habitat differences among transects/quadrats – note the slope, aspect and condition of the substrate type. In evaluation, investigator may control for it (stratify the design) or include variation (but factor out abiotic factors either quantitatively or qualitatively).

4. In addition to individual metrics, consider cumulative measures of response. For example, a "percent healthy community" metric can be summary percent cover for healthy biota (*i.e.*, excluding unhealthy algae, bare substrate, dead barnacles, or *Enteromorpha*).

5. Describe food pathways evident from the species composition.

6. Assess differences in species composition in reference to potential COPC sensitivity.

7. Weight of evidence – Compare biological patterns to other lines of evidence such as chemistry or toxicity of groundwater and/or seep water.

8. Reporting should include a diagram of the transects/quadrats with indications of factors that could influence the observed species composition (*e.g.*: contaminants, drainage, substrate, *etc*.) Photographs of each quadrat should be provided.

An example (hypothetical) of an analytical tool is provided in Figure I-15-6. In the figure, the cumulative metric of "percent healthy" is used in conjunction with chemistry and groundwater toxicity data. Despite the high variability of the community metrics, there is some indication of an association among the main lines of evidence.

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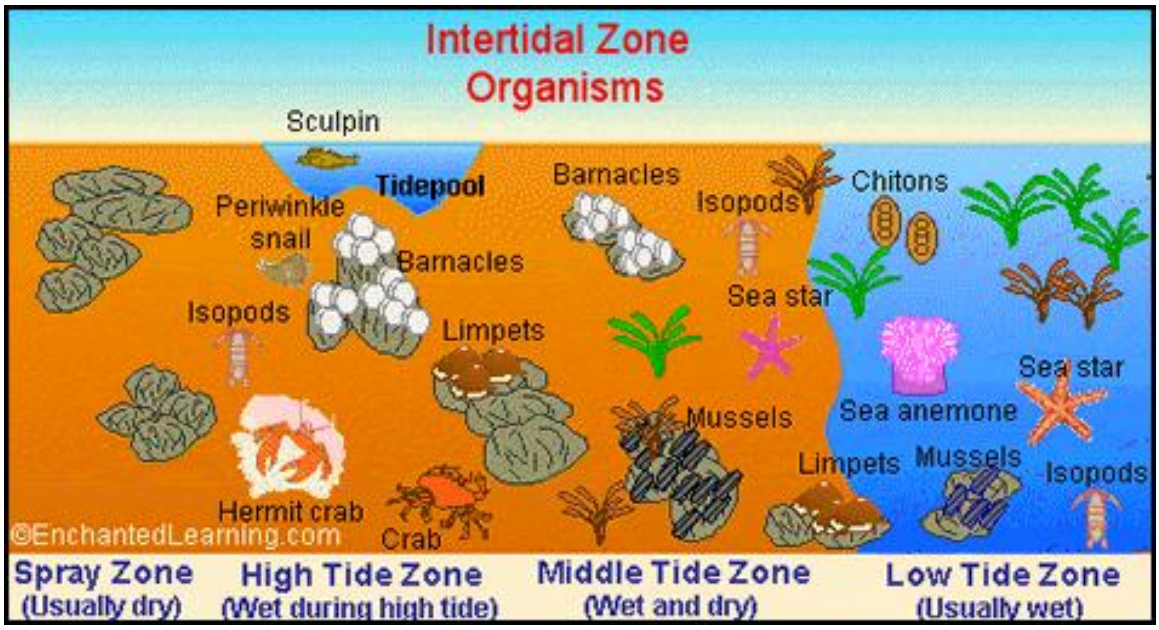

FIGURE I-15-1: Conceptual Diagram of Intertidal Zone, Showing Organism Types and Vertical Stratification

Figure taken from: http://www.enchantedlearning.com/biomes/intertidal/intertidal.shtml

FIGURE I-15-2: Example of Macroalgae Categorized as "Unhealthy" in BC Intertidal Survey

FIGURE I-15-3: Second Example of Macroalgae Categorized as "Unhealthy" in BC Intertidal Survey

FIGURE I-15-4: Example of Macroalgae Categorized as "Healthy" in BC Intertidal Survey

FIGURE I-15-5: Second Example of Macroalgae Categorized as "Healthy" in BC Intertidal Survey

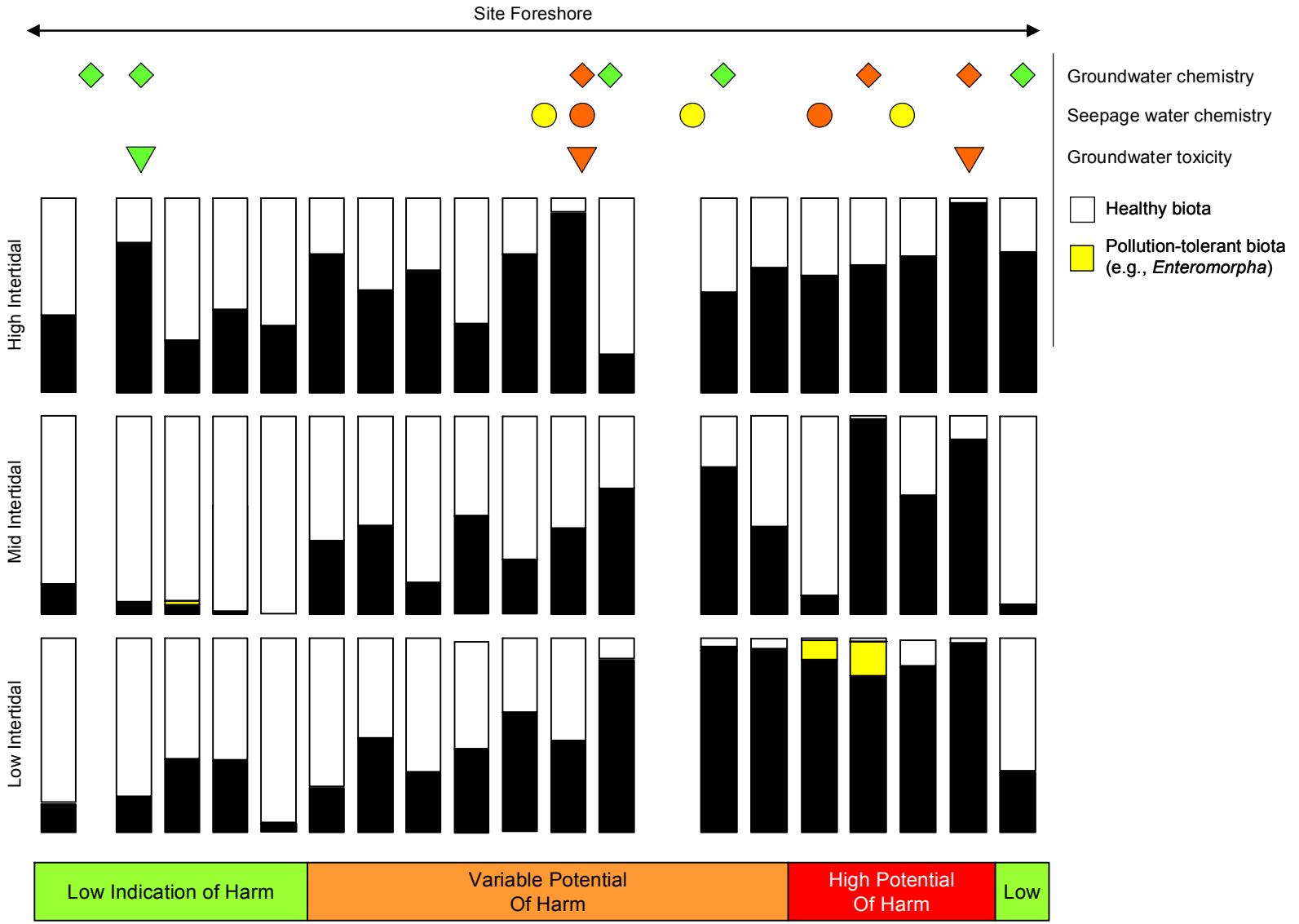

FIGURE I-15-6: Example of an Interpretation of Intertidal Benthic Community Survey Data Using Multiple Lines of Evidence

DIRECT MEASUREMENT TOOL #16

16.0 VASCULAR PLANT COMMUNITY SURVEYS

16.1 Overview

What does this tool consist of? Resident plant communities can be valuable indicators of the presence of contaminants on a site and can complement other lines of evidence (*e.g.*, soil toxicity tests) in the risk assessment process. Plants are particularly useful in this regard because:

- They are sessile (*i.e.*, fixed in place), and are constantly exposed to any contaminants that are present in the soil or groundwater;
- Localized impacts of contaminants on plant health may be readily apparent through mortality, retarded growth, or discoloration; and,
- They colonize many different habitat types, including sites heavily disturbed by human activities.

In short, vascular plant community surveys can be used to assess the effects of contaminant exposure. To the extent possible and as merited based on the site conditions and management objectives, such studies should be as quantitative as possible.

Which ecosystem(s) would this tool typically be applied in? Upland Terrestrial (Wildlands and Human Use). The tool is more commonly applied in natural environments due to the confounding effect of introduced species, weeds, and ornamentals in populated areas.

How frequently is this tool used in a DERA? Not commonly used.

What are the benefits of using this tool in a DERA?

- With consistent survey methods and a good understanding of exposure conditions, vascular plan surveys can identify effects of chemical exposure.
- For some sites (*e.g.*, wildlands settings), plant communities are an important ecological component; because they are sessile they allow spatial analysis of environmental effects.
- Identifications of vascular plants are well known and numerous identification keys are available.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- The often heterogeneous nature of plant habitats, particularly with anthropogenic influences such as soil compaction (*e.g.*, roads) can challenge the sampling design of such studies. Factors such as slope, orientation, drainage, soil type, and light exposure can vary across a site, complicating interpretation of survey data. This issue is discussed further in Section I-16.3.

- It can be very challenging to establish reference or background conditions (alternatives are described in Section I-16.4).

- Achieving statistical power can be challenging and alternative interpretation approaches may be required (see Section I-16.4).

- Environmental disturbances, such as drought, fire, and physical habitat alteration may confound data collection and interpretation (see Section I-16.4).

- The details of the sampling design will vary from site to site depending on the nature of the site and the assessment endpoints of the risk assessment. Section I-16.5 presents a strawman study design that would be useful for many DERAs; however, individual cases may call for alternative study designs. Such is acceptable provided that rationales are provided for the connection between the "how" (measurement endpoints and methodological details) and the "why" (assessment endpoints and risk hypotheses).

- As with all direct assessments of biological communities, natural variability and stochasticity, combined with incertitude regarding ecological processes, will limit the extent to which results may be interpreted in a rigorous quantitative fashion. Not all known physical factors can be controlled in a practical study design, and unknown factors (ecological uncertainties) will persist. Investigators should be realistic about the level of quantitative precision and/or certainty that can be expected from studies of this type.

16.2 Relevant Vegetation Organism Types

Vegetation descriptions may be based on either physiognomic (structural) features or on floristic (taxonomic) analyses. The former description type entails large area assessments of morphology, life-form, and other broad features, whereas the latter is based on the presence and abundance of species in a study area (Kent and Coker, 1994). Both types of analysis may be used in ecological risk assessments, however floristic analysis is generally more appropriate for the scale that is typical of contaminated sites. There are contaminated sites situations where the physiognomic analyses may be appropriate, but the choice is at the discretion of the risk assessor and they should provide their rationale.

General Vegetation Types

A primarily physiognomic scheme has been used by Klinkenberg (2007) to describe the broad plant assemblages found in British Columbia. A brief summary of each vegetation zone is provided below, and representative dominant taxa are listed in Table I-16-1.

- *Coniferous forest* – Evergreen coniferous forest dominates the province's vegetative cover.

- *Deciduous forest* – Trembling aspen (*Populus tremuloides*) is the most widespread and abundant deciduous tree species in British Columbia, with alders, cottonwoods, poplars, willows, and birches also important regionally.

- *Scrub* – Shrubby vegetation dominates the landscape in the dry southern Interior, the North, the outer northern Coast, and portions of high elevation habitat throughout the province.

- *Grass* – Grass-like plants are found in marshes and grassy tundra as well as in typical grassland ecosystems (*e.g.*, dry southern interior valleys).

- *Broad-leaved herb* – Timberline meadows are the only widespread, natural, broad-leaved herbaceous vegetation type in British Columbia.

- *Bryoid* – Bryoid vegetation consists of mosses, liverworts, and/or lichens, and usually occurs in environments too harsh for vascular plants.

During problem formulation, the practitioner should identify which of the above habitats best represents the site. There might be a combination of habitats, or perhaps none are representative if the site is anthropogencially altered to a large degree (*e.g.*, fully landscaped with ornamentals). In these cases, the risk assessor may want to involve a specialist to develop a strategy appropriate for a given site.

Rare Species

British Columbia is home to more than 2300 native vascular plant taxa, of which more than 600 are considered rare in the province (Klinkenberg, 2007). Of the rare plant taxa, more than 200 of these are species that are rare because of human activities (Douglas *et al.*, 2002). As indicated in Section I-16.5, assessment of rare species requires special considerations.

16.3 General Considerations for Vegetation Surveys

Kent and Coker (1994) identified four initial considerations in conducting vegetation descriptions:

- Purpose of Survey – the features and characteristics described in a vegetation survey will vary depending on overall aims and objectives. In ecological risk assessments, the purpose of the study is typically to evaluate the survival, growth, and reproduction of resident plant species. Accordingly, the features of interest relate mainly to species presence/absence, species density, visual evidence of health, and comparisons to reference conditions. Such studies can be paired with evaluation of contaminant update from plants to wildlife.

- Scale of Study – different methods apply to small and large sites. For the purpose of this Appendix, it is assumed that the site is small to medium in size (*i.e.*, tens to hundreds of metres across[1]) and complexity. Larger, complex, or wide area sites will typically require different approaches.

- Overall Habitat Type – different techniques will apply for each major vegetation unit described in Section I-16.2. For example, a mid-successional forest requires evaluation of the vertical dimension (*e.g.*, canopy, understory, ground cover) whereas a grassland assessment generally does not.

- Resources – The time, effort, and other resources will be proportional to the size of the site and level of detail required. In risk assessments, vegetation surveys are usually considered as part of weight-of-evidence for assessing potential impairment. As such, the costs of conducting detailed field assessments must be traded off against other lines of evidence, such as soil toxicity tests and contaminant concentrations in soil or plant tissue. As with other DERA tools, plant community assessments are often conducted in tiers, with the level of effort and detail increased in subsequent phases (Morris *et al.*, 1995).

All vegetation assessments conducted to support ERAs should, at minimum, identify and describe major habitat/vegetation types and their character in relation to contaminant concentrations. Additional studies[2] may be required "to obtain detailed information on

[1] Site sizes may be defined as small (<1,500 m^2), medium (1,500-<12,000 m^2), and large (>12,000 m^2) using thresholds previously developed for contaminated sites. However, prescriptive cutoffs are not recommended for discriminating between site "sizes" in vegetation assessments because the scale will depend on both areal extent and vegetation complexity (heterogeneity and diversity). The practitioner is advised to exercise professional judgement in determining whether a site is classified as a large or complex site requiring a different study design than discussed in this Appendix.

[2] Additional studies may include formal hypothesis testing and/or detailed vegetation descriptions including Raunkaier's life-form classification, structural-physiognomic classifications (Dansereau, Kuchler, or Fosberg techniques), or floristic quantifications based on quadrats or transects (Kent and Coker, 1992).

the distribution and abundance of selected species, elucidate a complex community pattern, or determine the relationships between species or communities and one or more critical factors". Alternatively, initial screening assessments may indicate that additional analyses are unlikely to be worthwhile due to the obscuring effect of confounding factors (human use, natural biological variations).

Although the presence of "unnatural" looking vegetation on a site may suggest the presence of a contaminant, the composition and health of a site's resident plant community is influenced by a wide range of other factors, including:

- Ecological factors, including anthropogenic abiotic stressors or disturbances;
- Plant phenology;
- Variable growth forms;
- Tolerance to contaminants (toxicokinetic and/or toxicodynamic tolerance); and,
- Introduced weed species.

These broad factors are discussed in turn below, with discussion of the implications for conducting terrestrial plant surveys to support ecological risk assessment.

Ecological Factors – A range of biotic and abiotic factors influence plant growth and community structure. Communities on two superficially similar sites may be very different because of subtle differences between the sites with respect to one or more of these factors.

- Biotic Influences – these may be intraspecific or interspecific. Intraspecific relations are among individuals of the same species within a population. An example of intraspecific competition is when shading from a thick forest canopy prevents new trees of the same species from growing on the forest floor. Interspecific relations are between different species, and include interspecific competition, predation (*e.g.*, grazing or browsing, insect pests), parasitism, and disease. A positive intraspecific relation occurs when the root mass of a pioneering species creates a stable habitat for colonization by another species.

- Abiotic Influences – These include chemical and physical factors (*i.e.*, sunlight, oxygen, temperature, soil and climate), as well as perturbations (*e.g.*, fire, landslides, natural disasters). Anthropogenic activities (*e.g.*, land clearing, vehicular or pedestrian traffic) also cause perturbations. On a disused industrial site, vegetation growth may be greater around the site perimeter than at the center of the site due to the influence of vehicular traffic.

Phenology – Phenology is the study of timing of periodic phenomena such as flowering, growth initiation, growth cessation, litter fall, *etc.*, especially as related to seasonal changes in temperature, photoperiod, *etc.* In the context of plant communities, the main phenological attributes of interest are the growing season, seasonality, and seral succession.

- Growing Season – The growing season is the portion of the year during which conditions are most favorable for plant growth. Growing season length is affected by latitude (as one proceeds northward from the equator, the growing season generally gets shorter), altitude (growing season generally decreases with elevation above sea level, particularly in mountainous regions), and coastal influences (large water bodies have a moderating effect on climate, and growing seasons are often longer in coastal versus inland areas). The timing of a site visit in relation to the local growing season is an important consideration when interpreting the results of a plant survey, as a survey conducted outside the growing season will yield fewer apparent plant species than one conducted during the growing season. Also, it is more difficult to assess plant health when plants are dormant. The onset and end of the growing season will vary depending on the location of the site to be assessed. This will influence the stage in the seasonal community succession during which sampling occurs. At the end of the growing season, frost-resistant plants may persist considerably longer than species that die off quickly with the first frost.

- Seasonality – Within the growing season, plant species display a seasonal succession in their times of sprouting, growth, fruiting, and senescence. The plant species observed during a site visit conducted at the beginning of the growing season may be very different than the species visible at the end of the season. Flowering and fruiting can be sensitive indicators of plant health and the scheduling of vegetation surveys should consider the value of surveys that coincide with the expected timing of flowering and fruiting.

- Seral Succession – The composition of a plant community does not typically remain static over time. Apart from the regular fluctuations in species abundance related to seasonal changes, communities develop progressively over time through a recognizable sequence known as the sere. Pioneer populations are replaced by successive colonists along a more-or-less predictable path toward a relatively stable community. This process of succession results from interactions between different species, and between species and the environment, which govern the sequence and the rate with which species replace each other. In some cases, seres may take hundreds of years to complete. Adjacent sites may be identified as successively older stages of the same sere, if it is assumed that conditions were similar when each seral stage was initiated. Because sites under investigation are often disturbed by human activity, their plant communities may be in the early seral stages, during which community composition changes rapidly.

Variable Growth Forms – The same plant species may exhibit a range of different growth forms depending on the conditions under which the plant grows. Species such as arrowhead (*Sagittaria latifolia*) have extremely variable leaves and different types of flowers on male and female plants. This can make identification challenging, and can complicate comparisons among sites.

Tolerance to Contaminants – Different plant species have different tolerances to metals and other contaminants. For example, the common cattail (*Typha latifolia*) is highly tolerant to salinity, low pH, and metals exposure, and healthy plants can be observed growing in contaminated wetlands where no other plants will grow. Lesions on several vascular plant species have been used as bioindicators of photochemical smog (National Research Council, 1986). For woody vegetation in wetlands, shallow-rooted species are generally believed to be more sensitive to contaminants than deep-rooted species, due to their greater exposure to waterborne contaminants (Adamus and Brandt, 1990). However, vegetation response to contaminants is highly contaminant- and species-specific, such that few generalizations can be made. For example, a study of the response of wetland species to an oil spill (Burk, 1977) reported post-spill absence of red maple (*Acer rubrum*), but no effect or increase in sugar maple (*Acer saccharinum*) and wild grape (*Vitis labrusca*), providing evidence that contaminant sensitivity can range significantly even within the same genus. The practitioner should conduct a preliminary literature review of the primary COPCs and regional species to determine whether bioindicator species are available. This review can be supported by additional toxicological information available through the U.S. EPA (2008) ECOTOX database.

Weeds – Introduced plant species are common and widespread in developed areas, and are often the most successful colonizers of disturbed sites. As a result, the plant communities on these sites may be very different from what would occur naturally. Table 2 lists weed species that are common in British Columbia (Cranston *et al.*, 2002). Although weed species should be characterized, their protection is a site-specific issue, and protection goals for weeds (if applicable) should be documented in the DERA problem formulation.

16.4 Specific Issues and Considerations for Vegetation Surveys

Disturbance – The practitioner should evaluate whether the site has been disturbed by human activities, and to what extent. As many risk assessments are conducted in urbanized or semi-urbanized areas, there are numerous ecological disruptions (rights-of-way, excavations, regrading, infilling, traffic) that would significantly affect the nature of the plant communities. Furthermore, the magnitude of the impact is often correlated with the contamination profile because both contamination and physical alteration are often proportional to the intensity of human access. Kent and Coker (2004) describe the spectrum of biotic controls on plant communities including chopping,

mowing, burning, manuring, grazing, treading, ploughing, cutting sods, and excavating. The investigator must consider not only the current status of human interference at a site, but also the historical influences. Human disturbances are linked to the successional stage of the community, and because multiple climax communities may develop in response to environmental perturbations, it is not possible to identify a single climax (*i.e.,* idealized regional plant community) against which to evaluate a site. In general, the greater the level of human disturbance, the more challenging it is to identify and discriminate contaminant influences.

Reference Conditions – The practitioner should determine whether there is a suitable reference site nearby, particularly in terms of being well matched in physical human influences, biophysical conditions, and other factors. What is the vegetation community one would expect to see on the site based on the community observed at the reference site? This question is generally answered by comparing the site and reference with respect to structural and functional attributes (physiognomic approach) and taxonomic analysis (floristic approach), and may consist of both quantitative and qualitative methods.

Causality – If impairment of plant growth, abundance, diversity or plant health is apparent, the investigator should assemble a site specific catalog of possible causative variables. Soil conditions, moisture, pest infestations, and disease are some possible plausible explanations for observed plant impairment. The practitioner should consider the strength of the empirical evidence for the likely influence of each possible causative variable, including contamination, and the specificity of the observed impairment (*e.g.,* are leaf deformities characteristic of a specific contaminant influence, or are they generic responses to a number of potential stresses?).

Pollution Tolerance – If an apparently healthy plant community is present on the site, the investigator should evaluate whether it is composed mainly of pollution tolerant species (Medina *et al.*, 2003). For example, ragweed (*Ambrosia artemisiifolia*) shows a high tolerance to lead and can accumulate mainly in its roots up to l ,600 mg/kg without lethal consequences (Brandes and Nitzsche, 2006). Alternatively, the investigator could document the presence, absence, or density of species known to demonstrate sensitivity to the COPCs. The investigator should conduct a literature review on the COPCs identified using ORNL (1994), EPA (2008), or other screening methodology, and determine whether there are species in the region that fall on either extreme of the sensitivity distribution; these may be useful indicators of potential effects to communities.

16.5 Suggested Monitoring Methods

The complexity of environmental and human influences on plant communities makes it difficult to specify a single approach to sampling. The literature review summarized in Section I-16.7 provides a number of useful references for the design and interpretation of vegetation studies. In this section, we provide a general framework for conducting plant surveys, emphasizing the floristic methods that are commonly applied to small to medium-sized sites.

General Plant Surveys

The Washington Department of Natural Resources (WDNR, 2006), the Alberta Native Plant Council (ANPC, 2000), and Klinkenberg (2007) have compiled guidelines for conducting floristic site surveys. Methods described in these references were integrated to provide the following guidance on a systematic procedure for conducting plant surveys that incorporates the needs of the ERA process:

1. Ideally, find a place from which you can view the entire site, or alternatively use an aerial photograph to examine the variability of the site vegetation. Obtain and review all available mapping resources for the study area, including topographic, biophysical, and/or species maps;

2. Note the major vegetation types present on the site, and differences in vegetation among different areas of the site. Do not focus on specific plant species at this preliminary stage, but instead look at the general plant community patterns.

3. Look for patterns of natural and anthropogenic alteration (including wildlife alterations) or biophysical controls (limiting factors) that should be controlled for in the study design.

4. Determine the habitat and ecosystems present in the study area, including consideration of off-site communities. Evaluate whether there are reference communities that are appropriate for comparison to the site, and/or whether there is a soil contamination gradient extending across similar habitat types;

5. Obtain and review a list of the species possibly present in the study area. Review the regional status for these species and highlight the rare species of the region;

6. Compare the list of regional rare species with available habitat types in the study area, and adjust the list to include only those species that could potentially occur in the habitat and ecosystems present in the study area;

7. Delineate the polygons encompassing the vegetation types on a site plan or on the aerial photograph. Identify representative 100 m^2 plots within major vegetation type polygons. The number of plots will depend on the contamination gradient and distribution of limiting factors for plant development. The location of plots should be indicated on the site plan or aerial photograph. The shape of the plot (*e.g.*, 10 m × 10 m, 20 m × 5 m) should be appropriate for the polygon shape.

8. Review the flowering periods and growing requirements for rare species or indicator species in order to determine desirable survey periods.

9. Consider seasonal coverage of the site in order to include flowering periods. Multiple sampling rounds may be required depending on the objectives of the study and the desired degree of certainty.

10. Mark and record the plot boundaries, using appropriate methods and/or technologies. Use semi-permanent markers if there will be monitoring over time as part of the assessment.

11. Identify the major plant species present in the plot. Plants inside the border are counted, and plants rooted outside the border, but with branches extending over the sides of the plot, should also be included in the sample.

12. Document the presence or absence of species identified to be potential bioindicators of chemical contamination.

13. Assign cover values for the main plant species in the plot, and estimate cover values as shown in Table 3.

14. Investigate candidate plant species for visual evidence of deformity or disease (*e.g.*, stunting, wilting, discoloration);

15. Obtain voucher documentation for each rare species found or for representative indicator species (if applicable)[3];

16. Consider collection of tissue samples for chemical analysis to support subsequent weight-of-evidence evaluations. Patterns where changes in plant community coincide with changes in plant tissue COPC concentrations are central to evaluating the hypothesis that COPC concentrations are the primary stressors influencing plant community structure. Also, any collection of tissue contaminant data should

[3] Klinkenberg (2007) guidance rare plant surveys suggest collecting vouichers only if at least 20 individuals are available, but cautions that even this may be damaging to some sensitive populations. Digital photography and leaf collection are being encouraged as alternatives for rare plants.

consider the plant components (*e.g.*, seeds, berries, or foliage) relevant to the wildlife food consumption pathway, if such a pathway is part of the conceptual model. These data may be used to parameterize a wildlife trophic transfer model (Appendix II-13) or replace generic bioaccumulation factors (Appendix II-1) with site-specific data.

17. Check identifications with experts using herbarium collections, and obtain verifications of identifications (if applicable).

Study Options

The generic study design presented above provides a general framework that should be applicable to most sites and situations. Due to the complexity of terrestrial plant ecology, the range of sites considered under DERA, and the large amount of literature on alternative sampling methodologies, there is a need to incorporate flexibility in this technical aspect of DERA. Section I-16.7 provides a number of useful references; consideration of these or other references may result in customizing a sampling program on a site-specific basis. Considerations in tailoring the study design include (Morris *et al.*, 1995):

- Sample Size – There is no objective way of determining an absolute minimum requirement; however, sampling of numerous small quadrats is generally preferred to sampling of a few large quadrats. The optimal size of the quadrat can be established by assessing a preliminary quadrat that progressively doubles in size: the resulting species-area curve describes a minimal quadrat size that provides optimal information about species diversity (Kent and Coker, 1994)

- Sampling Pattern – The approach to positioning of quadrats may be based on random sampling, systematic regular sampling, restricted random sampling, stratified sampling, or selective sampling. Each of these techniques has advantages and disadvantages that should be carefully considered in study design, and rationale presented in the reporting phase. At many sites, stratified or selective sampling is conducted in an effort to deliberately control for discrete vegetation boundaries or to sample along a gradient of environmental contamination. Where this is performed, the decisions and assumptions should be discussed. For example, a stratified sampling program may be preferred when a site contains multiple ecotypes; in such situations the practitioner should consider concentrating effort on the more uncommon ecotypes in order to maximize detection of rare plants.

- Species Abundance Measures – In addition to the semi-quantitative abundance methods described above, other methods include number of individuals (enumerated abundance or density), cover percentage (using line-intercept or point-intercept

method), combined cover and abundance (to avoid underestimating importance of small species with scattered individuals), and frequency (percentage of observations in the sample[s] that contain the species).

- Relevant Environmental Factors – Climatic, edaphic (soil-related), and hydrological factors should be considered in terms of their potential influence on vegetation characteristics, particularly in terms of interactions with contamination gradients. Human alterations of the environmental factors, either direct or indirect, may have substantial influence on the study design.

- Data Analysis Methods – The sampling design should be established to satisfy the data analysis needs, which in turn are governed by the measurement endpoints and testable hypotheses determined during problem formulation. For example, a study conducted to identify a threshold exposure value for phytotoxicity may require greater representation of the contamination gradient relative to a study that seeks to determine whether worst-case soil contamination leads to alteration of ecological condition (*i.e.*, reference comparison).

Rare Species

The requirement to evaluate rare species should be evaluated during the Problem Formulation stage of risk assessment. Rare plants can occur in a broad range of habitats, however, they are more frequently found in uncommon habitats and transition zones, such as limestone outcrops, seeps, ephemeral wetlands areas, *etc.* Protocols for rare vascular plant surveys are found in Klinkenberg (2007). An essential tool for assessing plant species at risk is the Conservation Data Centre BC Species and Ecosystem Explorer[4] that enables screening of species at risk by forest district and biogeoclimatic zone. The output can be pared down further by considering the specific habitat on the site and consulting local experts. The strategy for screening and identifying rare species is analogous to the procedure for wildlife discusses in Appendix I-18 (Sections I-18.3 and I.18.4).

WDNR (2006) has identified the following three points as the most important factors in rare plant surveys:

- The taxonomic ability of the surveyors;
- Surveying at the appropriate time of year; and,
- Full documentation of methods for determination.

[4] http://www.env.gov.bc.ca/atrisk/toolintro.html

Klinkenberg (2007) notes that the ability to recognize and identify rare flora is highly specialized, requiring knowledge of plant ecology and taxonomy obtained over many years. When faced with a requirement for rare species evaluations, the practitioner should consult a specialist and consult detailed guidance for rare plant enumerations.

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TABLE I-16-1 Common Plant Species in British Columbia Vegetation Zones, Compiled from Klackenberg (2007)

Vegetation Type	Subtype	Dominant Species
Coniferous Forest	Coastal forest – low to medium elevation	*Tsuga heterophylla* (Western hemlock) *Thuja plicata* (Western redcedar) *Pseudotsuga menziesii* (Douglas-fir) *Abies amabilis* (Amabilis fir) *Picea sitchensis* (Sitka spruce) *Arbutus menziesii* (Arbutus)
	Coastal forest – Subalpine	*Tsuga mertensiana* (Mountain hemlock) *Abies amabilis* (Amabilis fir) *Chamaecyparis nootkatensis* (Yellow-cedar)
	Dry forest, parkland and savanna of the southern Interior	*Pinus ponderosa* (Ponderosa pine) *Pseudotsuga menziesii* (Douglas-fir) *Larix occidentalis* (Western larch)
	Southern half of the Interior Plateau	*Pinus contorta* var. *latifolia* (Lodgepole pine) *Pseudotsuga menziesii* (Douglas-fir)
	Sorthern half of the Interior Plateau	*Picea glauca* (White spruce) *Picea engelmannii* x *glauca* (Hybrid white spruce) *Abies lasiocarpa* (Subalpine fir) *Pinus contorta* var. *latifolia* (Lodgepole pine)
	Columbia and Southern Rocky mountains	*Tsuga heterophylla* (Western hemlock) *Thuja plicata* (Western redcedar) *Pinus monticola* (Western white pine) *Pseudotsuga menziesii* (Douglas-fir) *Larix occidentalis* (Western larch) *Abies grandis* (Grand fir) *Picea engelmannii* (Engelmann spruce) *Picea engelmannii* x *glauca* (Hybrid white spruce) *Abies lasiocarpa* (Subalpine fir)
	Upper elevation southern interior British Columbia	*Picea engelmannii* (Engelmann spruce) *Abies lasiocarpa* (Subalpine fir) *Pinus contorta* var. *latifolia* (Lodgepole pine) *Pinus albicaulis* (Whitebark pine)
	Low / middle elevation of northern boreal	*Picea glauca* (White spruce) *Picea mariana* (Black spruce) *Pinus contorta* var. *latifolia* (Lodgepole pine)
	Northern subalpine forest	*Picea glauca* (White spruce) *Abies lasiocarpa* (Subalpine fir)
Deciduous Forest	Interior Plateau and boreal forest region	*Populus tremuloides* (Trembling aspen) *Populus balsamifera* (Black cottonwood) *Betula papyrifera* (Paper birch)
	Coastal forest	*Alnus rubra* (Red alder)
	Northern forest	*Populus balsamifera* ssp. *Balsamifera* (Balsam poplar)

Vegetation Type	Subtype	Dominant Species
	Southwestern British Columbia	*Acer macrophyllum* (Bigleaf maple) *Quercus garryana* (Garry oak)
Scrub	Dry southern Interior (shrub-steppe)	*Artemisia tridentate* (Big sagebrush) *Ericameria nauseosus*, *Chrysothamnus nauseosus* (Rabbit-brush) *Purshia tridentate* (Antelope-brush) Several grass species
	Northern forest and Great Plains	*Salix* spp. (Willows) *Ledum groenlandicum* (Labrador tea) *Chamaedaphne calyculata* (Leatherleaf) *Betula nana* (Scrub birch) *Picea mariana* (Stunted black spruce_
	Outer northern coast	*Pinus contorta* var. *contorta* (Stunted shore pine) *Chamaecyparis nootkatensis* (Yellow-cedar) *Thuja plicata* (Western redcedar) Hemlocks *Juniperus communis*) (Common juniper) *Ledum groenlandicum* (Labrador tea) *Gaultheria shallon* (Salal) *Kalmia microphylla* ssp. *occidentalis* (Bog-laurel) *Vaccinium* spp. (Blueberries and huckleberries) *Empetrum nigrum* (Crowberry)
	Subalpine and alpine	*Salix* spp. (Willows) *Betula nana* (Scrub birch) *Vaccinium* spp. (Blueberries and huckleberries) *Alnus viridis* ssp. *sinuata* (Sitka alder). *Cassiope* and *Phyllodoce* spp. (Mountain-heathers) *Empetrum nigrum*, *Dryas* spp. (Mountain-avens)
Grasslands	Southcentral and Southeastern British Columbia	*Pseudoroegneria spicata* (Bluebunch wheatgrass; formerly known as *Agropyron spicatum*) *Festuca altaica* s. *lat.* (Altai fescue; including *Festuca scabrella*, Rough fescue) *Festuca idahoensis* (Idaho fescue) *Poa sandbergii* s. *lat.* (Sandberg's bluegrass) *Koeleria macrantha* (Junegrass) *Bromus tectorum* (Cheatgrass) *Hesperostipa comata* (Needle-and-thread grass) *Achnatherum* (=*Stipa*) *richardsonii* (Spreading needlegrass) *Hesperostipa curtiseta* (Porcupinegrass; formerly known as *Stipa spartea* var. *curtiseta*) *Poa pratensis* (Kentucky bluegrass) *Artemisia tridentata*, *Artemisia frigida* (Pasture sage) *Ericameria* (=*Chrysothamnus*) *nauseosus* (Rubber rabbitbrush)

Vegetation Type	Subtype	Dominant Species
Grasslands (continued)	Southwestern British Columbia (dry vernal grasslands)	*Bromus* (Bromegrass) *Vulpia* (Fescue) *Aira* (Hairgrass) Many introduced species Spring-flowering forbs
	Northern two-thirds of British Columbia (low elevations)	*Elymus trachycaulus* (Slender wheatgrass) *Festuca altaica* (Altai fescue) *Calamagrostis purpurascens* (Purple reedgrass) *Achnatherum richardsonii* (Spreading needlegrass) *Achnatherum nelsonii* (Stiff needlegrass; formerly known as *Stipa columbiana*) *Leymus innovatus* (Fuzzy-spiked wildrye) *Poa glauca* (Glaucous bluegrass) *Artemisia frigida* and *A. campestris* (Northern wormwood)
	High elevation (dry areas)	*Festuca altaica* (now including former *F. scabrella*) *Festuca viridula* (Green fescue) *Festuca brachyphylla* (Alpine fescue) *Poa arctica* (Arctic bluegrass) *Hierochloe alpina* (Alpine sweetgrass) *Calamagrostis purpurascens* (Purple reedgrass) *Carex phaeocephala* (Dunhead sedge) *Carex spectabilis* (Showy sedge) *Carex microchaeta* (Small-awned sedge) *Carex nardina* (Spikenard sedge) *Carex albonigra* (Two-toned sedge) *Carex scirpoidea ssp. pseudoscirpoidea* (Single-spiked sedge) *Carex capitata* (Capitate sedge) *Kobresia myosuroides* (Bellard's kobresia)
Wetland	Freshwater marsh	*Carex aquatilis* (Water sedge) *Carex utriculata* (Beaked sedge) *Carex vesicaria* (Inflated sedge) *Carex nigricans* (Black alpine sedge) *Schoenoplectus acutus* & *Schoenoplectus tabernaemontani* (Great bulrush; formerly *Scirpus lacustris* s. *lat.*) *Trichophorum caespitosum* (Tufted clubrush) *Phalaris arundinacea* (Reed canarygrass) *Phragmites australis* (Common reed)
Wetland (continued)	Coastal saline marsh	*Carex lyngbyei* (Lyngbye's sedge) *Deschampsia cespitosa* (Tufted hairgrass)
	Alkaline marsh (dry southern interior)	*Distichlis spicata* var. *stricta* (Alkali saltgrass) *Muehlenbergia asperifolia* (Alkali muhly) *Hordeum jubatum* (Foxtail barley)

Vegetation Type	Subtype	Dominant Species
		Juncus balticus (Baltic rush) *Schoenoplectus* spp. (Bulrush) *Salicornia europaea* (European glasswort) *Suaeda depressa* (Seablite)
Broad-leaved herb	High elevations of the southern two-thirds of the Interior of British Columbia	*Senecio triangularis* (Arrow-leaved groundsel) *Veratrum viride* (False hellebore) *Valeriana sitchensis* (Sitka valerian) *Erigeron peregrinus* (Subalpine daisy) *Lupinus arcticus* (Arctic lupine)
	Mid-elevations (localized)	*Heracleum maximum* (Cow-parsnip) *Epilobium angustifolium* (Fireweed)
Bryoid	Harsh environments throughout British Columbia	*Sphagnum* bogs Lichens and mosses such as *Racomitrium*, *Polytrichum*, and *Dicranum* Alpine lichen tundra

TABLE I-16-2: Common Weed Species in British Columbia (from Cranston *et al.*, 2002)

Common Name	Latin Name
Annual bluegrass	*Poa annua*
Annual sowthistle	*Sonchus oleraceus*
Baby's-breath	*Gypsophila paniculata*
Barnyardgrass	*Echinochloa crusgalli*
Bladder campion	*Silene cucubalus*
Blueweed	*Echium vulgare*
Bog rush	*Juncus effusus*
Broad-leaved plantain	*Plantago major*
Bull thistle	*Cirsium vulgare*
Burdock	*Arctium* spp.
Canada thistle	*Cirsium arvense*
Chicory	*Cichorium intybus*
Cleavers	*Galium aparine*
Cluster tarweed	*Madia glomerata*
Common bugloss	*Anchusa officinalis*
Common chickweed	*Stellaria media*
Common mallow	*Malva neglecta*
Common tansy	*Tanacetum vulgare*
Corn spurry	*Spergula arvensis*
Creeping buttercup	*Ranunculus repens*
Crupina	*Crupina vulgaris*
Cudweed	*Gnaphalium uliginosum*
Curled dock	*Rumex crispus*
Dalmatian toadflax	*Linaria dalmatica*
Diffuse knapweed	*Centaurea diffusa*
Dodder	*Cuscuta* spp.
Field bindweed	*Convolvulus arvensis*
Field horsetail	*Equisetum arvense*
Field scabious	*Knautia arvensis*
Foxtail barley	*Hordeum jubatum*
Giant hogweed	*Heracleum mantegazzianum*
Gorse	*Ulex europaeus*

Common Name	Latin Name
Green foxtail	*Setaria viridis*
Groundsel	*Senecio vulgaris*
Hemp-nettle	*Galeopsis tetrahit*
Henbit	*Lamium amplexicaule*
Himalayan balsam	*Impatiens glandulifera*
Hoary alyssum	*Berteroa incana*
Hoary cress	*Cardaria spp.*
Hound's-tongue	*Cynoglossum officinale*
Japanese knotweed	*Polygonum cuspidatum*
Jointed goatgrass	*Aegilops cylindrica*
Kochia	*Kochia scoparia*
Lady's-thumb	*Polygonum persicaria*
Lamb's-quarters	*Chenopodium album*
Leafy spurge	*Euphorbia esula*
Marsh plume thistle	*Cirsium palustre*
Meadow knapweed	*Centaurea pratensis*
Mullein	*Verbascum thapsus*
Night-flowering catchfly	*Silene noctiflora*
Nightshade	*Solanum species*
Nodding beggar-ticks	*Bidens cernua*
Nodding thistle, a.k.a. Musk thistle	*Carduus nutans*
Orange hawkweed	*Hieracium aurantiacum*
Oxeye daisy	*Chrysanthemum leucanthemum*
Perennial pepperweed	*Lepidium latifolium*
Perennial sowthistle	*Sonchus arvensis*
Pineappleweed	*Matricaria matricariodes*
Plumeless thistle	*Carduus acanthoides*
Puncturevine	*Tribulus terrestris*
Purple loosestrife	*Lythrum salicaria*
Purple nutsedge	*Cyperus rotundus*
Quackgrass	*Agropyron repens*
Redroot pigweed	*Amaranthus retroflexus*
Rush skeletonweed	*Chondrilla juncea*
Russian knapweed	*Acroptilon repens*
Russian thistle	*Salsola kali*

Common Name	Latin Name
Scentless chamomile	*Matricaria maritima*
Scotch broom	*Cytisus scoparius*
Scotch thistle	*Onopordum acanthium*
Sheep sorrel	*Rumex acetosella*
Shepherd's-purse	*Capsella bursa-pastoris*
Showy milkweed	*Asclepias speciosa*
Spiny annual sow-thistle	*Sonchus asper*
Spotted knapweed	*Centaurea maculosa*
St. John's-wort	*Hypericum perforatum*
Stinkweed	*Thlapsi arvense*
Sulphur cinquefoil	*Potentilla recta*
Tansy ragwort	*Senecio jacobaea*
Tartary buckwheat	*Fagopyrum tataricum*
Velvetleaf	*Abutilon theophrasti*
Water hemlock	*Cicuta douglasii*
Western goat's-beard	*Tragopogon dubius*
White cockle	*Lychnis alba*
Wild buckwheat	*Polygonum convolvulus*
Wild chervil	*Anthriscus sylvestris*
Wild mustard	*Sinapsis arvensis*
Wild oats	*Avena fatua*
Witchgrass	*Panicum capillare*
Yellow nutsedge	*Cyperus esculentus*
Yellow starthistle	*Centaurea solstitialis*
Yellow toadflax	*Linaria vulgaris*

TABLE I-16-3: Relationship between Cover Class and Estimate of Total Percent Cover

Cover Class (CC)	Cover Percentage	Description
6	75-100%	Nearly completely covered.
5	50-75%	Large group, definitely >50% cover.
4	25-50%	Small group, with near 50% cover.
3	5-25%	Plant common in plot, with >5% cover.
2	1-5%	Plant is well established in plot; minimal coverage.
1	0-1%	Plant is rare; insignificant cover.

DIRECT MEASUREMENT TOOL #17

17.0 OTHER POPULATION AND COMMUNITY SURVEYS

What does this tool consist of? Population and community surveys consist of a wide range of environmental investigations and objectives, and may incorporate information on organism presence, reproduction, behaviour, and ecological interactions. From a DERA perspective, the most important output from population and community surveys relates to quantification of types and numbers of organisms (*e.g.*, inventory studies). Inventory studies typically have the objective of estimating population distribution (presence/absence), relative abundance and/or absolute abundance.

The DERA manual includes expanded modules on survey methods for some specific environments (*e.g.*, rocky intertidal communities, vascular plant communities), and it is expected that additional modules will be developed over time. This appendix chapter is intended to summarize issues common to the remaining ecosystem types for which expanded modules are not yet available.

Which ecosystem(s) would this tool typically be applied in? All ecosystems. Population and community surveys are conducted as supporting lines of evidence in conjunction with toxicity evaluations and screening to environmental benchmark values.

How frequently is this tool used in a DERA? This tool is common for DERAs for aquatic ecosystems (*i.e.*, deep aquatic; rivers & streams, shoreline), and particularly for assessments of invertebrate community composition (see Appendix I-14). The tool is more rarely applied in DERAs for terrestrial ecosystems (uplands wildlands; uplands human use). For example, meaningful assessments of wildlife populations in the context of chemical stressors are complex, resource-intensive, and often relate to the regional or watershed level rather than the spatial scale of most contaminated sites. Evaluation of fish, amphibian, soil invertebrate, and small mammal communities is conducted more frequently than bird or large mammal population assessment. These kinds of tools may have more common use as part of a "top-down" assessment, where risks to higher-level organisms are predicted (*e.g.*, based on food chain modeling) such that more detailed and long-term study/monitoring is required.

What are the benefits of using this tool in a DERA?

- Direct measurement of *in situ* communities has high ecological relevance, and is often closely linked to the assessment endpoint for the ERA.

- Community surveys complement other lines of evidence (*e.g.*, laboratory toxicity tests) because the community structure reflects the response of the ROPCs to the

environmentally-relevant COPC. Contaminant bioavailability, modifying factors, and compensatory mechanisms are accounted for in a population study.

- Data from screening-level qualitative surveys assist in the selection of appropriate ROPCs, and also for determining the relationships among ROPCs (*e.g.*, feeding preferences).

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- Community surveys should be considered in conjunction with other lines of evidence because natural variability can confound the interpretation of the survey data. For example:

  - *In situ* communities can be influenced by numerous abiotic habitat factors (*e.g.*, variations in sediment grain size, soil quality, *etc*) as well as landscape- or watershed-level influences (*e.g.*, habitat alteration from forest fire; logging, *etc)*. It is difficult to establish cause-effect relationships between the community-level measurement and the specific COPCs under investigation.

  - Seasonal influences must be considered (*e.g.*, seasonal patterns in food availability; site occupation; variation in sensitivity to COPCs due to life history stage).

  - Mobile and migratory species may also be exposed to stressors outside the contaminated site.

- Historical/baseline data for a given ecosystem are not always available. These data are important for establishing the bounds of natural variability both spatially and temporally.

- The secretive (cryptic) nature of some species makes them difficult to inventory by direct counts or quantification of signs (tracks, scat, *etc.*).

- Physical capture of individuals to provide data has a risk of causing impairment of the health of the affected individuals (*e.g.*, trapping, electrofishing, *etc.*). When sampling for absolute abundance, more intensive sampling/trapping is required, increasing these risks.

- Survey results are sensitive to survey timing, and include fluctuations related to the reproductive cycle, dispersal of juveniles, and general population movements. Certain small mammal populations follow multi-year cycles (*e.g.*, cyclic population irruptions for voles).

- Capture probabilities can be highly variable. White *et al.* (1982) identify three possible sources of variation in capture probabilities, including: time variation (variance over time due to weather effects or the amount of effort used to capture animals on any occasion); heterogeneity (variance among individuals due to innate factors such as age, sex, social status or the number of traps); and behavioural variation (variance depending on whether the individual had been previously captured).

- Considerable resources may be required to conduct surveys that provide robust data (*e.g.*, large enough sample size to detect statistically significant differences; multiple sampling events to address seasonality of the community).

Where can I find additional information about this tool?

The methods for population and community surveys are highly receptor- and site-specific. Study design for many wildlife population assessments (*e.g.*, large mammal surveys) requires the involvement of a specialized wildlife biologist with local knowledge and familiarity with the life-history of the organism(s) under evaluation. However, here are some generic protocols for sampling and inventory analysis of some wildlife groups that provide a starting point for these studies. For example, the Multiple Species Inventory and Monitoring (MSIM) protocol (Manley *et al.*, 2006; Manley and Van Horne, 2004) documents primary survey methods for obtaining basic presence/absence data and associated habitat condition data for several taxonomic groups, including terrestrial and aquatic birds, mammals, amphibians, reptiles, and plants. The Ministry of Environment (BC MELP, 1997, 1998a, 1998b) has also developed guidance for sampling of bacteria, zooplankton, phytoplankton, benthic fauna, fish, and small mammals.

A list of useful sampling and study design references is provided below:

- BC MELP (British Columbia Ministry of Environment, Lands and Parks, [renamed as BC Ministry of Environment]). 1990. Procedures for Environmental Monitoring in Range and Wildlife Habitat Monitoring. Habitat Monitoring Committee. Ministry of Environment, Lands and Parks and Ministry of Forests, Victoria, BC

- BC MELP. 1997. *Freshwater Biological Sampling Manual*. Published by the Resources Inventory Committee. Integrated Land Management Bureau. Available at: http://ilmbwww.gov.bc.ca/risc/pubs/aquatic/freshwaterbio/index.htm#TOC

- BC MELP. 1998a. *Inventory Methods for Small Mammals: Shrews, Voles, Mice & Rats*. Standards for Components of British Columbia's Biodiversity No. 31. Prepared by Ministry of Environment, Lands and Parks, Resources Inventory Branch for the

Terrestrial Ecosystems Task Force Resources Inventory Committee. Version 2.0. May 6, 1998. Available at: http://ilmbwww.gov.bc.ca/risc/pubs/tebiodiv/smallmammals/

- BC MELP. 1998b. Inventory Methods for Marten and Weasels: Standards for Components of British Columbia's Biodiversity. No. 24.

- Corn, P.S. and R.B. Bury. 1990. *Sampling Methods for Terrestrial Amphibians and Reptiles*. United States Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland OR. General Technical Report. PNW-GTR-256.

- Davis, D.E. 1982. *CRC Handbook of Census Methods for Terrestrial Vertebrates*. CRC Press, Inc., Boca Raton, FL.

- Environment Canada. 2002. *Metal Mining Guidance Document for Aquatic Environmental Effects Monitoring*. National Environmental Effects Monitoring (EEM) Office, Environment Canada, Ottawa, Ontario. Available at: http://www.ec.gc.ca/eem/english/MetalMining/Guidance/default.cfm

- Jones, C., W.J. McShae, M.J. Conroy, and T.H. Dunz. 1996. Capturing Mammals. In: D.E. Wilson, F.R. Cole, J.D. Nichols, R. Rudran, and M.S. Foster (Eds). *Measuring and Monitoring Biological Diversity: Standard Methods for Mammals*. Smithsonian Institution Press, Washington, DC. pp. 115-155.

- Manley, P.N. and B. Van Horne. 2004. *The Multiple Species Inventory and Monitoring Protocol: A Population, Community, and Biodiversity Monitoring Solution for National Forest System Lands*. International Monitoring Science and Technology Symposium Proceedings. September 2004. Available at: http://www.fs.fed.us/psw/programs/snrc/featured_topics/msim/documents/msts_msim_manley_and_vanhorne_2004.pdf

- Manley, P.N., B. Van Horne, J.K. Roth, W.J. Zielinski, M.M. McKenzie, T.J. Weller, F.W. Weckerly, and C. Vojta. 2006. *Multiple Species Inventory and Monitoring Technical Guide*. General Technical Report WO-73. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 204 pp. Available at: http://www.fs.fed.us/rm/pubs_other/wo_gtr073.pdf

- Pearson, D.E. and L.F. Ruggiero. 2003. Transect versus Grid Trapping Arrangements for Sampling Small-Mammal Communities. *Wildlife Society Bulletin* 31(2):454-459.

- Sullivan, T.P. 1997. *Sampling Methodology for Small Mammals*. University of BC Forestry-Wildlife Integrated Management. 31 pp.

- Suter, G.W. 1996. *Risk Characterization for Ecological Risk Assessment of Contaminated Sites*. Prepared by Lockheed Martin Energy Systems, Inc., Oakridge, TN for U.S. Department of Energy. ES/ER/TM-200.

- White, G.C., D.R. Anderson, K.P. Burnham, and D.L. Otis. 1982. *Capture-Recapture and Removal Methods for Sampling Closed Populations*. Los Alamos, NM: Los Alamos National Laboratory. 235 pp.

DIRECT MEASUREMENT TOOL #18

18.0 DEVELOPMENT OF SITE-SPECIFIC WILDLIFE SPECIES LISTS

18.1 Overview

What does this tool consist of? This tool consists of enumeration and screening methods used to develop a list of wildlife species relevant to a contaminated site assessment. Although detailed field surveys can be used to document the presence, absence, or density of species (*e.g.*, scat surveys, live trapping, scent-post surveys, burrow searches, wildlife call/response surveys, *etc.*), these surveys require highly specialized expertise and are typically applied only for advanced risk assessments. In practice, most risk assessments apply simpler procedures based on screening of literature-derived information combined with a site-specific habitat assessment by a professional biologist. This appendix describes a systematic procedure for identifying candidate species, and enumerates the wildlife species found in British Columbia in tabular format. The tables provide a starting point for the site-specific identification of species.

Generating species lists can laborious, time-consuming, and prone to inaccuracies. In this appendix, standard lists of the bird, mammal, reptile, amphibian, and fish species that could occur at a site in British Columbia were generated (using the methodologies as outlined in Section I-18.3) and are provided herein. Section I-18-4 describes methods that the risk assessment practioner can then use to screen these lists to generate an annotated site-specific species list.

Which ecosystem(s) would this tool typically be applied in? Upland Terrestrial (Wildlands and Human Use), Rivers and Streams.

How frequently is this tool used in a DERA? Common. Development of receptor lists is often conducted at the screening level; in DERA the level of detail increases and may entail analysis to species level.

What are the benefits of using this tool in a DERA? In the context of an ecological risk assessment, an accurate list of wildlife species that could potentially occur on a site is useful because:

- It allows identification of the potential wildlife receptors present on a site, and may be used in the identification of receptors of potential concern (ROPCs) during problem formulation.

- It avoids identification of the species that could occur in a broad geographic area (*e.g.*, a Forest District) but that are unlikely to be relevant to the site of interest.

By taking into account their regional or site-specific distribution and habitat preferences, a species inventory can be customized to maximize site-relevance. For example, most seabird species listed as occurring in Coastal BC are very unlikely to be found at terrestrial sites. Other broad habitat features can be used to efficiently screen species inventories.

- Many wildlife species listed as potentially occurring in a given geographic area are accidental or casual visitors, and are known from only one or a few local records. These species are very unlikely to be encountered on a project site, and are unlikely to be suitable for selection as ROPCs, except in the case of listed species.

- Identification of the possibility of listed species at the planning stage provides an opportunity for specific consideration of these species in the risk assessment.

What are the common pitfalls or issues that should be considered when using this tool in a DERA?

- The common and Latin names of wildlife species change over time, and it is important that the most current nomenclature be used. For example, in 2006, the American Ornithologists' Union (AOU) recently split one common BC species, the "blue grouse" (*Dendragapus obscurus*) into the "dusky grouse" (*Dendragapus obscurus*) in the Rocky Mountains and the "sooty grouse" (*Dendragapus fuliginosus*) in the Pacific Coast Ranges.

- Regional species lists are commonly available but are limited in utility because they cover wide areas encompassing multiple habitat types, not all of which may be relevant to the site. Furthermore, such lists do not take into consideration site usage and habitat factors that make the site more (or less) suitable for certain species. Customization of regional lists is discussed in detail in Section I-18.4.

18.2 Linkage to SLRA

MOE (2008) has prepared a protocol for screening level risk assessment (SLRA) that includes consideration of receptor identification and selection. In SLRA, a habitat assessment must be conducted (Appendix B in MOE, 2008). The main difference between SLRA and DERA with respect to receptor identification is that the latter may require greater taxonomic detail and/or rigour, potentially including identifications to species level. MOE (2008) includes *Form B-2* that is a checklist for the identification of feeding guilds, but it does not require enumeration of species (except COSEWIC-listed, red-listed, or blue-listed species).

There are several similarities between the SLRA and DERA approaches to receptor identification, and the approaches are compatible (*i.e.*, a species enumeration could be conducted in DERA following completion of a Form B-2 in SLRA). Both require the practitioner to check for local expertise. In the SLRA framework, a site visit and interviews with local residents are required. In DERA, it is also recommended that a site visit be conducted; however, the site visit should be supported by the check for local inventories and/or local expertise. Another similarity is that both SLRA and DERA require identification of any COSEWIC-listed, red-listed, or blue-listed species that may be present in the vicinity of the site. Such is required because the listed species are considered individually in the risk assessment and have different decision rules for determining acceptable effects.

18.3 Provincial Species Inventories

Integrated and harmonized provincial species lists were compiled and are presented in Tables I-18-1 through Table I-18-5. These broad lists represent the starting point for species evaluations and enumerations in British Columbia. Provincial species lists for BC were generated from the following sources:

- **Birds** – A base can be generated from the Royal BC Museum's "Birds of British Columbia" series (Campbell *et al.*, 1992a, 1992b, 1997, 2001). Information contained in the Campbell *et al*. books was used to identify rare, casual or accidental bird species. The base list was compared with a species list generated by the BC Ministry of Environment's (MOE's) Conservation Data Centre (CDC) "Species and Ecosystems Explorer" web portal[5] to verify that the nomenclature was current, and the nomenclature was further verified against the AOU's on-line checklist[6].

- **Mammals** – A base list of mammals of BC was derived from the 2007 list developed by David Nagorsen, which is posted on the E-Fauna BC website[7]. The base list was compared with a species list generated by the "Species and Ecosystems Explorer" web portal to verify that the nomenclature is current.

- **Reptiles and Amphibians –** A base list was derived from the "Checklist of the Amphibians and Reptiles of British Columbia (last updated April 2007)", edited by Brent Matsuda and posted on the E-Fauna BC website. The base list was compared with a species list generated by the "Species and Ecosystems Explorer" web portal to verify that the nomenclature is current.

[5] http://www.env.gov.bc.ca/atrisk/toolintro.html
[6] http://www.aou.org/checklist/index.php3
[7] http://www.geog.ubc.ca/biodiversity/efauna/SpeciesChecklists.html

- **Fish –** This list was derived from J.D. McPhail's 2007 "Annotated Checklist of the Indigenous Species of Freshwater Fish of British Columbia (2007)", which is posted on the E-Fauna BC website. This list was supplemented with the MOE's checklist of introduced species[8].

Updates to these lists may be required over time (*i.e.*, approximately every five years) to accommodate future changes to species status or nomenclature.

18.4 Refinement of Provincial Species Lists

An ERA practitioner should filter the provincial species lists described in Section I-18.3 to remove irrelevant species, using the following steps:

1) *Consult Local Species Inventories* – Determine whether there is a local species list available for the region. For example, bird species checklists have been developed for many localities in BC, including Vancouver, the Pacific Rim National Park, the Tumbler Ridge area, the George C. Reifel Migratory Bird Sanctuary in Delta, and the Richmond Nature Park. Some lists are available online at the E-Fauna BC web site[9], but it can be a challenge to obtain copies of older lists, which may be out of print. The practitioner should bear in mind that the species nomenclature in regional lists may differ from that which is currently in use.

2) *Check for Local Expertise* – As appropriate, consult with a local naturalist or biologist who is willing to share the knowledge of the local fauna with you. This can often add valuable local insight (*e.g.*, about which species are found on your site, and which species are common or rare).

3) *Consult Range Maps* – If it is not possible to find or obtain a regional checklist, the alternative is to consult the range maps found in the Campbell *et al*. bird references, the most recent BC Museum handbooks for reptiles, amphibians, and mammals (Matsuda *et al*., 2006; Nagorsen and Brigham, 1993; Nagorsen, 1996, 2005; Shackleton, 1999), or the BC Resource Information Standards Committee standards for fish and wildlife[10]. McPhail and Carveth (1993) provides lists of, and keys to, fish species present in the various large watersheds of the province[11]. Partial checklists of fish species present in BC streams and lakes are accessible through the MOE's "Fishwizard" database[12].

[8] http://www.env.gov.bc.ca/wld/fishhabitats/introduced.html
[9] http://efauna.bc.ca/
[10] http://ilmbwww.gov.bc.ca/risc/pubs/index.html
[11] http://ilmbwww.gov.bc.ca/risc/pubs/aquatic/freshfish/assets/fresh.pdf
[12] http://www.fishwizard.com/

4) *Short-List Species* – Use either the checklists or range maps to short-list species which occur in the region of the project site.

5) *Determine the Applicable Forest District* – A map showing the Forest District of boundaries is accessible via the "Map" button on the Conservation Data Centre (CDC)'s "Species and Ecosystems Explorer" web interface[13].

6) *Identify Threatened and Endangered Species* – Use the CDC Species and Ecosystems Explorer to generate a list of the rare, threatened, or endangered wildlife species found in the applicable Forest District. The CDC's provincial "blue list" includes any "element (*i.e.*, ecological community, indigenous species, or subspecies) that is of "Special Concern" (formerly "Vulnerable") in BC. Elements are considered to be of Special Concern if their characteristics make them particularly sensitive to human activities or natural events. Blue-listed elements are considered to be at risk, but are not "Extirpated", "Endangered" or "Threatened". The CDC's "red list" includes any element that is "Extirpated", "Endangered", or "Threatened" in BC. Extirpated elements no longer exist in the wild in BC, but occur elsewhere. Endangered elements are facing imminent extirpation or extinction throughout their range. Threatened elements are likely to become endangered if limiting factors are not addressed. Additional guidance for identifying red and blue-listed species and their geographic range is available online[14].

   Under the federal *Species at Risk Act* (SARA), the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) has the responsibility to evaluate the status of wildlife in Canada, and designate "species at risk" in one of the five following categories: special concern, threatened, endangered, extirpated, or extinct. "Special Concern" species are those that may become threatened or endangered because of a combination of biological characteristics and identified threats if no action is taken to prevent this. "Threatened" species are those likely to become endangered if nothing is done to reverse the factors leading to their extirpation or extinction. "Endangered" species face imminent extirpation or extinction. The SARA-listed species are listed in SARA Schedule 1[15], and the CDC Species and Inventory Explorer provides species' SARA listings in addition to provincial listings. Additional guidance for identifying COSEWIC species and their geographic range is available online[16].

[13] http://srmapps.gov.bc.ca/apps/eswp/
[14] http://www.env.gov.bc.ca/cdc/
[15] http://www.sararegistry.gc.ca/species/schedules_e.cfm?id=1
[16] http://www.cosewic.gc.ca/eng/sct5/index_e.cfm

7) For ease of comparison with your species list, make sure the list is sorted taxonomically. Annotate information regarding provincial (CDC) and federal (SARA) listing of elements onto your list.

8) Annotate your species list with any confirmed sightings of species in or near the project site, or local abundance (*e.g.*, from site visits, local species lists, information from local naturalists, publications, *etc*.).

9) Have the finished species list peer-reviewed by a wildlife biologist before incorporating it into your report.

18.5 Application

Once the practitioner has completed assembly and filtering of the site-specific wildlife species list, the final step is to highlight the species of greatest concern in the context of the risk assessment. This includes species that are considered at risk, species that have a close association with the site (*e.g.*, rodents which have made burrows into contaminated soil), or species that are consumed by people (*e.g.*, deer) and species that occupy the main feeding niches and taxa. The process for identifying ROPCs is described in the Problem Formulation section of this document; accordingly, the wildlife identifications described in this section should be considered along with other factors (societal values, sensitivity, representativeness) in determining the final list of ROPCs.

18.6 References

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TABLE I-18-1: British Columbia Species Inventory for Birds

Order	Family	Common Name	Latin Name	BC Code	Status
Anseriformes	Anatidae	Fulvous whistling-duck	*Dendrocygna bicolor*	B-FWDU	Accidental
		Greater white-fronted goose	*Anser albifrons*	B-GWFG	Native
		Emperor goose	*Chen canagica*	B-EMGO	Very rare
		Snow goose	*Chen caerulescens*	B-SNGO	Native
		Ross's goose	*Chen rossii*	B-ROGO	Very rare
		Brant	*Branta bernicla*	B-BRAN	Native
		Cackling goose	*Branta hutchinsii*	B-CKGO	Native
		Canada goose	*Branta canadensis*	B-CAGO	Native
		Mute swan	*Cygnus olor*	B-MUSW	Introduced
		Trumpeter swan	*Cygnus buccinator*	B-TRUS	Native
		Tundra swan	*Cygnus columbianus*	B-TUSW	Native
		Whooper swan	*Cygnus cygnus*	B-WHSW	Accidental
		Wood duck	*Aix sponsa*	B-WODU	Native
		Northern pintail	*Anas acuta*	B-NOPI	Native
		American wigeon	*Anas americana*	B-AMWI	Native
		Northern shoveler	*Anas clypeata*	B-NOSL	Native
		Green-winged teal	*Anas crecca*	B-GWTE	Native
		Cinnamon teal	*Anas cyanoptera*	B-CITE	Native
		Blue-winged teal	*Anas discors*	B-BWTE	Native
		Falcated duck	*Anas falcata*	B-FADU	Casual

Order	Family	Common Name	Latin Name	BC Code	Status
		Baikal teal	*Anas formosa*	B-BATE	Accidental
		Eurasian wigeon	*Anas penelope*	B-EUWI	Native
		Mallard	*Anas platyrhynchos*	B-MALL	Native
		Garganey	*Anas querquedula*	B-GARG	Accidental
		American black duck	*Anas rubripes*	B-ABDU	Introduced
		Gadwall	*Anas strepera*	B-GADW	Native
		Lesser scaup	*Aythya affinis*	B-LESC	Native
		Redhead	*Aythya americana*	B-REDH	Native
		Ring-necked duck	*Aythya collaris*	B-RNDU	Native
		Tufted duck	*Aythya fuligula*	B-TUDU	Native
		Greater scaup	*Aythya marila*	B-GRSC	Native
		Canvasback	*Aythya valisineria*	B-CANV	Native
		Steller's eider	*Polysticta stelleri*	B-STEI	Casual
		Spectacled eider	*Somateria fischeri*	B-SPEI	Accidental
		Common eider	*Somateria mollissima*	B-COEI	Casual
		King eider	*Somateria spectabilis*	B-KIEI	Very rare
		Harlequin duck	*Histrionicus histrionicus*	B-HADU	Native
		White-winged scoter	*Melanitta fusca*	B-WWSC	Native
		Surf scoter	*Melanitta perspicillata*	B-SUSC	Native
		Black scoter	*Melanitta nigra*	B-BLSC	Native
		Long-tailed duck (Oldsquaw)	*Clangula hyemalis*	B-OLDS	Native
		Bufflehead	*Bucephala albeola*	B-BUFF	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Common goldeneye	*Bucephala clangula*	B-COGO	Native
		Barrow's goldeneye	*Bucephala islandica*	B-BAGO	Native
		Smew	*Mergellus albellus*	B-SMEW	Casual
		Hooded merganser	*Lophodytes cucullatus*	B-HOME	Native
		Red-breasted Merganser	*Mergus serrator*	B-RBME	Native
		Common Merganser	*Mergus merganser*	B-COME	Native
		Ruddy duck	*Oxyura jamaicensis*	B-RUDU	Native
Galliformes	Cracidae	Chukar	*Alectoris chukar*	B-CHUK	Introduced
		Gray partridge	*Perdix perdix*	B-GRPA	Introduced
		Ring-necked pheasant	*Phasianus colchicus*	B-RNPH	Introduced
		Ruffed grouse	*Bonasa umbellus*	B-RUGR	Native
		Greater Sage-Grouse	*Centrocercus urophasianus*	B-GSGR	Extirpated
		Spruce grouse	*Falcipennis canadensis*	B-SPGR	Native
		Willow ptarmigan	*Lagopus lagopus*	B-WIPT	Native
		White-tailed ptarmigan	*Lagopus leucura*	B-WTPT	Native
		Rock ptarmigan	*Lagopus muta*	B-ROPT	Native
		Dusky Grouse (Blue grouse)	*Dendragapus obscurus*	B-DUGR	Native
		Sooty Grouse (Blue grouse)	*Dendragapus fuliginosus*	B-SOGR	Native
		Sharp-tailed grouse	*Tympanuchus phasianellus*	B-STGR	Native
		Wild turkey	*Meleagris gallopavo*	B-WITU	Introduced
	Odontophoridae	Mountain quail	*Oreortyx pictus*	B-MOQU	Introduced

Order	Family	Common Name	Latin Name	BC Code	Status
		California quail	*Callipepla californica*	B-CAQU	Introduced
		Northern bobwhite	*Colinus virginianus*	B-NOBO	Introduced
Gaviiformes	Gaviidae	Common loon	*Gavia immer*	B-COLO	Native
		Pacific loon	*Gavia pacifica*	B-PALO	Native
		Red-throated loon	*Gavia stellata*	B-RTLO	Native
		Yellow-billed loon	*Gavia adamsii*	B-YBLO	Native
Podicipediformes	Podicipedidae	Pied-billed grebe	*Podilymbus podiceps*	B-PBGR	Native
		Horned grebe	*Podiceps auritus*	B-HOGR	Native
		Red-necked grebe	*Podiceps grisegena*	B-RNGR	Native
		Eared grebe	*Podiceps nigricollis*	B-EAGR	Native
		Western grebe	*Aechmophorus occidentalis*	B-WEGR	Native
		Clark's grebe	*Aechmophorus clarkii*	B-CLGR	Native
Procellariiformes	Diomedeidae	Short-tailed albatross	*Phoebastria albatrus*	B-STAL	Accidental
		Laysan albatross	*Phoebastria immutabilis*	B-LAAL	Very rare
		Black-footed albatross	*Phoebastria nigripes*	B-BFAL	Native
	Procellariidae	Northern fulmar	*Fulmarus glacialis*	B-NOFU	Native
		Mottled petrel	*Pterodroma inexpectata*	B-MOPE	Casual
		Murphy's petrel	*Pterodroma ultima*	B-MUPE	Casual
		Buller's shearwater	*Puffinus bulleri*	B-BLSH	Native
		Flesh-footed shearwater	*Puffinus carneipes*	B-FFSH	Native
		Pink-footed shearwater	*Puffinus creatopus*	B-PFSH	Native
		Sooty shearwater	*Puffinus griseus*	B-SOSH	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Black-vented shearwater	*Puffinus opisthomelas*	B-BVSH	Very rare
		Manx shearwater	*Puffinus puffinus*	B-MASH	Casual
		Short-tailed shearwater	*Puffinus tenuirostris*	B-STSH	Native
	Hydrobatidae	Fork-tailed storm-petrel	*Oceanodroma furcata*	B-FTSP	Native
		Leach's storm-petrel	*Oceanodroma leucorhoa*	B-LSPE	Native
Pelecaniformes	Phaethontidae	Red-tailed tropicbird	*Phaethon rubricauda*	B-RTTR	Accidental
	Pelecanidae	American white pelican	*Pelecanus erythrorhynchos*	B-AWPE	Native
		Brown pelican	*Pelecanus occidentalis*	B-BRPE	Very rare
	Phalacrocoracidae	Double-crested cormorant	*Phalacrocorax auritus*	B-DCCO	Native
		Pelagic cormorant	*Phalacrocorax pelagicus*	B-PECO	Native
		Brandt's cormorant	*Phalacrocorax penicillatus*	B-BRCO	Native
		Red-faced cormorant	*Phalacrocorax urile*	B-RFCO	Accidental
	Fregatidae	Magnificent frigatebird	*Fregata magnificens*	B-MAFR	Accidental
Ciconiiformes	Ardeidae	American bittern	*Botaurus lentiginosus*	B-AMBI	Native
		Least bittern	*Ixobrychus exilis*	B-LEBI	Casual
		Great egret	*Ardea alba*	B-GREG	Very rare
		Great blue heron	*Ardea herodias*	B-GBHE	
		Little blue heron	*Egretta caerulea*	B-LBHE	Accidental
		Snowy egret	*Egretta thula*	B-SNEG	Casual
		Cattle egret	*Bubulcus ibis*	B-CAEG	Native
		Green heron	*Butorides virescens*	B-GRHE	Native
		Black-crowned night-heron	*Nycticorax nycticorax*	B-BCNH	Native

Order	Family	Common Name	Latin Name	BC Code	Status
	Threskiornithidae	White-faced ibis	*Plegadis chihi*	B-WFIB	Casual
	Ciconiidae	Wood stork	*Mycteria americana*	B-WOST	Accidental
	Cathartidae	Turkey vulture	*Cathartes aura*	B-TUVU	Native
		Black vulture	*Coragyps atratus*	B-BLVU	Casual
Falconiformes	Accipitridae	Osprey	*Pandion haliaetus*	B-OSPR	Native
		White-tailed kite	*Elanus leucurus*	B-WTKI	Casual
		Bald eagle	*Haliaeetus leucocephalus*	B-BAEA	Native
		Northern harrier	*Circus cyaneus*	B-NOHA	Native
		Sharp-shinned hawk	*Accipiter striatus*	B-SSHA	Native
		Cooper's hawk	*Accipiter cooperii*	B-COHA	Native
		Northern goshawk	*Accipiter gentilis*	B-NOGO	Native
		Broad-winged hawk	*Buteo platypterus*	B-BWHA	Rare
		Swainson's hawk	*Buteo swainsoni*	B-SWHA	Native
		Red-tailed hawk	*Buteo jamaicensis*	B-RTHA	Native
		Ferruginous hawk	*Buteo regalis*	B-FEHA	Very rare
		Rough-legged hawk	*Buteo lagopus*	B-RLHA	Native
		Golden eagle	*Aquila chrysaetos*	B-GOEA	Native
	Falconidae	Eurasian kestrel	*Falco tinnunculus*	B-EUKE	Accidental
		American kestrel	*Falco sparverius*	B-AMKE	Native
		Merlin	*Falco columbarius*	B-MERL	Native
		Gyrfalcon	*Falco rusticolus*	B-GYRF	Native
		Peregrine falcon	*Falco peregrinus*	B-PEFA	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Prairie falcon	*Falco mexicanus*	B-PRFA	Native
		Crested caracara	*Caracara cheriway*	B-CRCA	Accidental
Gruiformes	Rallidae	Yellow rail	*Coturnicops noveboracensis*	B-YERA	Native
		Virginia rail	*Rallus limicola*	B-VIRA	Native
		Sora	*Porzana carolina*	B-SORA	Native
		Common moorhen	*Gallinula chloropus*	B-COMO	Accidental
	Gruidae	Sandhill crane	*Grus canadensis*	B-SACR	Native
		Whooping crane	*Grus americana*	B-WHCR	Casual
Charadriiformes	Charadriidae	Black-bellied plover	*Pluvialis squatarola*	B-BBPL	Native
		American golden-plover	*Pluvialis dominica*	B-AGPL	Native
		Pacific golden plover	*Pluvialis fulva*	B-PGPL	Native
		Mongolian plover	*Pluvialis mongolus*	B-MGPL	Accidental
		Snowy plover	*Charadrius alexandrinus*	B-SNPL	Very rare
		Semipalmated plover	*Charadrius semipalmatus*	B-SEPL	Native
		Mountain plover	*Charadrius montanus*	B-MOPL	Accidental
		Killdeer	*Charadrius vociferus*	B-KILL	Native
		Black oystercatcher	*Haematopus bachmani*	B-BLOY	Native
		Black-necked stilt	*Himantopus mexicanus*	B-BNST	Very rare
		American avocet	*Recurvirostra americana*	B-AMAV	Native
	Scolopacidae	Terek sandpiper	*Xenus cinereus*	B-TESA	Accidental
		Spotted sandpiper	*Actitis macularius*	B-SPSA	Native
		Solitary sandpiper	*Tringa solitaria*	B-SOSA	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Wandering tattler	*Tringa incana (Heteroscelus incanus)*	B-WATA	Native
		Spotted redshank	*Tringa erythropus*	B-SPRE	Accidental
		Greater yellowlegs	*Tringa melanoleuca*	B-GRYE	Native
		Lesser yellowlegs	*Tringa flavipes*	B-LEYE	Native
		Willet	*Tringa semipalmata*	B-WILL	Very rare
		Upland sandpiper	*Bartramia longicauda*	B-UPSA	Native
		Whimbrel	*Numenius phaeopus*	B-WHIM	Native
		Bristle-thighed curlew	*Numenius tahitiensis*	B-BTCU	Casual
		Far Eastern curlew	*Numenius madagascariensis*	B-FECU	Accidental
		Long-billed curlew	*Numenius americanus*	B-LBCU	Native
		Hudsonian godwit	*Limosa haemastica*	B-HUGO	Native
		Bar-tailed godwit	*Limosa lapponica*	B-BTGO	Very rare
		Marbled godwit	*Limosa fedoa*	B-MAGO	Native
		Ruddy turnstone	*Arenaria interpres*	B-RUTU	Native
		Black turnstone	*Arenaria melanocephala*	B-BLTU	Native
		Surfbird	*Aphriza virgata*	B-SURF	Native
		Red knot	*Calidris canutus*	B-REKN	Native
		Sanderling	*Calidris alba*	B-SAND	Native
		Great knot	*Calidris tenuirostris*	B-GRKN	Casual
		Semipalmated sandpiper	*Calidris pusilla*	B-SESA	Native
		Western sandpiper	*Calidris mauri*	B-WESA	Native
		Red-necked stint	*Calidris ruficollis*	B-RNST	Casual

Order	Family	Common Name	Latin Name	BC Code	Status
		Little stint	*Calidris minuta*	B-LIST	Accidental
		Temminck's stint	*Calidris temminckii*	B-TEST	Accidental
		Long-toed stint	*Calidris subminuta*		Native
		Least sandpiper	*Calidris minutilla*	B-LESA	Native
		White-rumped sandpiper	*Calidris fuscicollis*		Native
		Baird's sandpiper	*Calidris bairdii*	B-BASA	Native
		Pectoral sandpiper	*Calidris melanotos*	B-PESA	Native
		Sharp-tailed sandpiper	*Calidris acuminata*	B-SHSA	Native
		Rock sandpiper	*Calidris ptilocnemis*	B-ROSA	Native
		Dunlin	*Calidris alpina*	B-DUNL	Native
		Curlew sandpiper	*Calidris ferruginea*	B-CUSA	Very rare
		Stilt sandpiper	*Micropalama himantopus*	B-STSA	Native
		Spoon-billed Sandpiper	*Eurynorhynchus pygmeus*	B-SBSA	Accidental
		Buff-breasted sandpiper	*Tryngites subruficollis*	B-BBSA	Native
		Ruff	*Philomachus pugnax*	B-RUFF	Casual
		Short-billed dowitcher	*Limnodromus griseus*	B-SBDO	Native
		Long-billed dowitcher	*Limnodromus scolopaceus*	B-LBDO	Native
		Wilson's Snipe	*Gallinago gallinago delicata*	B-WISN	Native
		Wilson's phalarope	*Phalaropus tricolor*	B-WIPH	Native
		Red-necked phalarope	*Phalaropus lobatus*	B-RNPL	Native
		Red phalarope	*Phalaropus fulicarius*	B-REPH	Native
	Laridae	Franklin's gull	*Larus pipixcan*	B-FRGU	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Laughing gull	*Larus atricilla*	B-LAGU	Accidental
		Little gull	*Larus minutus*	B-LIGU	Very rare
		Black-headed gull	*Larus ridibundus*	B-CBHG	Very rare
		Bonaparte's gull	*Larus philadelphia*	B-BOGU	Native
		Heermann's gull	*Larus heermanni*	B-HEEG	Native
		Black-tailed gull	*Larus crassirostris*	B-BTGU	Accidental
		Mew gull	*Larus canus*	B-MEGU	Native
		Ring-billed gull	*Larus delawarensis*	B-RBGU	Native
		California gull	*Larus californicus*	B-CAGU	Native
		Herring gull	*Larus argentatus*	B-HEGU	Native
		Thayer's gull	*Larus thayeri*	B-THGU	Native
		Iceland gull	*Larus glaucoides*	B-ICGU	Accidental
		Lesser black-backed gull	*Larus fuscus*	B-LBBG	Accidental
		Slaty-backed gull	*Larus schistisagus*	B-SBGU	Casual
		Western gull	*Larus occidentalis*	B-WEGU	Native
		Glaucous-winged gull	*Larus glaucescens*	B-GWGU	Native
		Glaucous gull	*Larus hyperboreus*	B-GLGU	Native
		Great black-backed gull	*Larus marinus*	B-GBBG	Accidental
		Sabine's gull	*Xema sabini*	B-SAGU	Native
		Black-legged kittiwake	*Rissa tridactyla*	B-BLKI	Native
		Ross's gull	*Rhodostethia rosea*	B-ROGU	Accidental
		Ivory gull	*Pagophila eburnea*	B-IVGU	Casual

Order	Family	Common Name	Latin Name	BC Code	Status
		Aleutian tern	*Onychoprion aleuticus*	B-ALTE	Casual
		Caspian tern	*Hydroprogne caspia*	B-CATE	Native
		Black tern	*Chlidonias niger*	B-BLTE	Native
		Common tern	*Sterna hirundo*	B-COTE	Native
		Arctic tern	*Sterna paradisaea*	B-ARTE	Native
		Forster's tern	*Sterna forsteri*	B-FOTE	Native
		Least tern	*Sterna antillarum*	B-LETE	Accidental
		Elegant tern	*Thalasseus elegans*	B-ELTE	Casual
	Stercorariidae	South polar skua	*Stercorarius maccormicki*	B-SPSK	Native
		Pomarine jaeger	*Stercorarius pomarinus*	B-POJA	Native
		Parasitic jaeger	*Stercorarius parasiticus*	B-PAJA	Native
		Long-tailed jaeger	*Stercorarius longicaudus*	B-LTJA	Native
	Alcidae	Common murre	*Uria aalge*	B-COMU	Native
		Thick-billed murre	*Uria lomvia*	B-TBMU	Very rare
		Pigeon guillemot	*Cepphus columba*	B-PIGU	Native
		Marbled murrelet	*Brachyramphus marmoratus*	B-MAMU	Native
		Kittlitz's murrelet	*Brachyramphus brevirostris*	B-KIMU	Accidental
		Xantus's murrelet	*Synthliboramphus hypoleucus*	B-XAMU	Accidental
		Ancient murrelet	*Synthliboramphus antiquus*	B-ANMU	Native
		Cassin's auklet	*Ptychoramphus aleuticus*	B-CAAU	Native
		Parakeet auklet	*Aethia psittacula*	B-PAAU	Casual
		Crested auklet	*Aethia cristatella*	B-CRAU	Accidental

Order	Family	Common Name	Latin Name	BC Code	Status
		Rhinoceros auklet	*Cerorhinca monocerata*	B-RHAU	Native
		Horned puffin	*Fratercula corniculata*	B-HOPU	Native
		Tufted puffin	*Fratercula cirrhata*	B-TUPU	Native
Columbiformes	Columbidae	Rock Pigeon	*Columba livia*	B-RODO	Introduced
		Passenger pigeon	*Ectopistes migratorius*	B-PAPI	Extinct
		Band-tailed pigeon	*Patagioenas fasciata*	B-BTPI	Native
		Oriental turtle-dove	*Streptopelia orientalis*	B-OTDO	Accidental
		White-winged dove	*Zenaida asiatica*	B-WWDO	Accidental
		Mourning dove	*Zenaida macroura*	B-MODO	Native
Cuculiformes	Cuculidae	Yellow-billed cuckoo	*Coccyzus americanus*	B-YBCU	Extirpated
		Black-billed cuckoo	*Coccyzus erythropthalmus*	B-BBCU	Very rare
Strigiformes	Tytonidae	Barn owl	*Tyto alba*	B-BNOW	Native
	Strigidae	Flammulated owl	*Otus flammeolus*	B-FLOW	Native
		Western screech-owl	*Megascops kennicottii*	B-WSOW	Native
		Great horned owl	*Bubo virginianus*	B-GHOW	Native
		Snowy owl	*Bubo scandiacus*	B-SNOW	Native
		Northern hawk owl	*Surnia ulula*	B-NHOW	Native
		Northern pygmy-owl	*Glaucidium gnoma*	B-NPOW	Native
		Burrowing owl	*Athene cinicularia*	B-BUOW	Very rare
		Spotted owl	*Strix occidentalis*	B-SPOW	Very rare
		Barred owl	*Strix varia*	B-BDOW	Native
		Great gray owl	*Strix nebulosa*	B-GGOW	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Long-eared owl	*Asio otus*	B-LEOW	Native
		Short-eared owl	*Asio flammeus*	B-SEOW	Native
		Boreal owl	*Aegolius funereus*	B-BOOW	Native
		Northern saw-whet owl	*Aegolius acadicus*	B-NSWO	Native
Caprimulgiformes	Caprimulgidae	Common nighthawk	*Chordeiles minor*	B-CONI	Native
		Common poorwill	*Phalaenoptilus nuttallii*	B-COPO	Native
Apodiformes	Apodidae	Black swift	*Cypseloides niger*	B-BLSW	Native
		Vaux's swift	*Chaetura vauxi*	B-VASW	Native
		White-throated swift	*Aeronautes saxatalis*	B-WTSW	Native
		Ruby-throated hummingbird	*Archilochus colubris*	B-RTHU	Casual
		Black-chinned hummingbird	*Archilochus alexandri*	B-BCHU	Very rare
		Anna's hummingbird	*Calypte anna*	B-ANHU	Native
		Costa's hummingbird	*Calypte costae*	B-COHU	Casual
		Calliope hummingbird	*Stellula calliope*	B-CAHU	Native
		Rufous hummingbird	*Selasphorus rufus*	B-RUHU	Native
		Xantus's hummingbird	*Hylocharus xantusii*	B-XAHU	Accidental
		Broad-tailed hummingbird	*Selasphorus platycercus*	B-BTHU	Accidental
Coraciiformes	Alcedinidae	Belted kingfisher	*Ceryle alcyon*	B-BEKI	Native
Piciformes	Picidae	Lewis's woodpecker	*Melanerpes lewis*	B-LEWO	Native
		Red-headed woodpecker	*Melanerpes erythrocephalus*	B-RHWO	Accidental
		Acorn woodpecker	*Melanerpes formicivorus*	B-ACWO	Accidental

Order	Family	Common Name	Latin Name	BC Code	Status
		Williamson's sapsucker	*Sphyrapicus thyroideus*	B-WISA	Native
		Yellow-bellied sapsucker	*Sphyrapicus varius*	B-YBSA	Native
		Red-naped sapsucker	*Sphyrapicus nuchalis*	B-RNSA	Native
		Red-breasted sapsucker	*Sphyrapicus ruber*	B-RBSA	Native
		Downy woodpecker	*Picoides pubescens*	B-DOWO	Native
		Hairy woodpecker	*Picoides villosus*	B-HAWO	Native
		White-headed woodpecker	*Picoides albolarvatus*	B-WHWO	Very rare
		American three-toed woodpecker	*Picoides dorsalis*	B-ATTW	Native
		Black-backed woodpecker	*Picoides arcticus*	B-BBWO	Native
		Northern flicker	*Colaptes auratus*	B-NOFL	Native
		Pileated woodpecker	*Dryocopus pileatus*	B-PIWO	Native
Passeriformes	Tyrannidae	Olive-sided flycatcher	*Contopus cooperi*	B-OSFL	Native
		Western wood-pewee	*Contopus sordidulus*	B-WWPE	Native
		Yellow-bellied flycatcher	*Empidonax flaviventris*	B-YBFL	Native
		Acadian flycatcher	*Empidonax virescens*	B-ACFL	Accidental
		Alder flycatcher	*Empidonax alnorum*	B-ALFL	Native
		Willow flycatcher	*Empidonax traillii*	B-WIFL	Native
		Least flycatcher	*Empidonax minimus*	B-LEFL	Native
		Hammond's flycatcher	*Empidonax hammondii*	B-HAFL	Native
		Gray flycatcher	*Empidonax wrightii*	B-GRFL	Native
		Dusky flycatcher	*Empidonax oberholseri*	B-DUFL	Native
		Pacific-slope flycatcher	*Empidonax difficilis*	B-PSFL	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Cordilleran flycatcher	*Empidonax occidentalis*	B-COFL	Native
		Black phoebe	*Sayornis nigricans*	B-BLPH	Casual
		Eastern phoebe	*Sayornis phoebe*	B-EAPH	Native
		Say's phoebe	*Sayornis saya*	B-SAPH	Native
		Ash-throated flycatcher	*Myiarchus cinerascens*	B-ATFL	Very rare
		Brown-crested flycatcher	*Myiarchus tyrannulus*		Unconfirmed
		Great crested flycatcher	*Myiarchus crinitus*	B-GCFL	Accidental
		Tropical kingbird	*Tyrannus melancholicus*	B-TRKI	Very rare
		Thick-billed kingbird	*Tyrannus crassirostris*	B-TBKI	Accidental
		Western kingbird	*Tyrannus verticalis*	B-WEKI	Native
		Eastern kingbird	*Tyrannus tyrannus*	B-EAKI	Native
		Gray kingbird	*Tyrannus dominicensis*	B-GRKI	Accidental
		Scissor-tailed flycatcher	*Tyrannus forficatus*	B-STFL	Very rare
	Laniidae	Loggerhead shrike	*Lanius ludovicianus*	B-LOSH	Very rare
		Northern shrike	*Lanius excubitor*	B-NOSH	Native
	Vireonidae	Cassin's vireo	*Vireo cassinii*	B-CAVI	Native
		Blue-headed vireo	*Vireo solitarius*	B-BHVI	Native
		Hutton's vireo	*Vireo huttoni*	B-HUVI	Native
		Warbling vireo	*Vireo gilvus*	B-WAVI	Native
		Philadelphia vireo	*Vireo philadelphicus*	B-PHVI	Native
		Red-eyed vireo	*Vireo olivaceus*	B-REVI	Native
	Corvidae	Gray jay	*Perisoreus canadensis*	B-GRJA	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Steller's jay	*Cyanocitta stelleri*	B-STJA	Native
		Blue jay	*Cyanocitta cristata*	B-BLJA	Native
		Western scrub-jay	*Aphelocoma californica*	B-CLNU	Accidental
		Clark's nutcracker	*Nucifraga columbiana*	B-CLNU	Native
		Black-billed magpie	*Pica hudsonia*	B-BBMA	Native
		American crow	*Corvus brachyrhynchos*	B-AMCR	Native
		Northwestern crow	*Corvus caurinus*	B-NOCR	Native
		Common raven	*Corvus corax*	B-CORA	Native
	Alaudidae	Sky lark	*Alauda arvensis*	B-SKLA	Introduced
		Horned lark	*Eremophila alpestris*	B-HOLA	Native
	Hirundinidae	Purple martin	*Progne subis*	B-PUMA	Native
		Tree swallow	*Tachycineta bicolor*	B-TRSW	Native
		Violet-green swallow	*Tachycineta thalassina*	B-VGSW	Native
		Northern rough-winged swallow	*Stelgidopteryx serripennis*	B-NRWS	Native
		Bank swallow	*Riparia riparia*	B-BKSW	Native
		Cliff swallow	*Petrochelidon pyrrhonota*	B-CLSW	Native
		Barn swallow	*Hirundo rustica*	B-BASW	Native
	Paridae	Black-capped chickadee	*Poecile atricapillus*	B-BCCH	Native
		Mountain chickadee	*Poecile gambeli*	B-MOCH	Native
		Chestnut-backed chickadee	*Poecile rufescens*	B-CBCH	Native
		Boreal Chickadee	*Poecile hudsonica*	B-BOCH	Native

Order	Family	Common Name	Latin Name	BC Code	Status
	Aegithalidae	Bushtit	*Psaltriparus minimus*	B-BUSH	Native
	Sittidae	Red-breasted nuthatch	*Sitta canadensis*	B-RBNU	Native
		White-breasted nuthatch	*Sitta carolinensis*	B-WBNU	Native
		Pygmy nuthatch	*Sitta pygmaea*	B-PYNU	Native
	Certhiidae	Brown creeper	*Certhia americana*	B-BRCR	Native
	Troglodytidae	Rock wren	*Salpinctes obsoletus*	B-ROWR	Native
		Canyon wren	*Catherpes mexicanus*	B-CAWR	Native
		Bewick's wren	*Thryomanes bewickii*	B-BEWR	Native
		House wren	*Troglodytes aedon*	B-HOWR	Native
		Winter wren	*Troglodytes troglodytes*	B-WIWR	Native
		Marsh wren	*Cistothorus palustris*	B-MAWR	Native
		American dipper	*Cinclus mexicanus*	B-AMDI	Native
	Regulidae	Golden-crowned kinglet	*Regulus satrapa*	B-GCKI	Native
		Ruby-crowned kinglet	*Regulus calendula*	B-RCKI	Native
	Sylviidae	Blue-gray gnatcatcher	*Polioptila caerulea*	B-BGGN	Casual
	Turdidae	Northern wheatear	*Oenanthe oenanthe*	B-NOWH	Accidental
		Western bluebird	*Sialia mexicana*	B-WEBL	Native
		Mountain bluebird	*Sialia currucoides*	B-MOBL	Native
		Townsend's solitaire	*Myadestes townsendi*	B-TOSO	Native
		Veery	*Catharus fuscescens*	B-VEER	Native
		Gray-cheeked thrush	*Catharus minimus*	B-GCTH	Native
		Swainson's thrush	*Catharus ustulatus*	B-SWTH	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Hermit thrush	*Catharus guttatus*	B-HETH	Native
		Eyebrowed thrush	*Turdus obscurus*		Unconfirmed
		Dusky thrush	*Turdus naumanni*	B-DUTH	Casual
		American robin	*Turdus migratorius*	B-AMRO	Native
		Varied thrush	*Ixoreus naevius*	B-VATH	Native
		Siberian accentor	*Prunella montanella*	B-SIAC	Accidental
	Mimidae	Gray catbird	*Dumetella carolinensis*	B-GRCA	Native
		Northern mockingbird	*Mimus polyglottos*	B-NOMO	Native
		Sage thrasher	*Oreoscoptes montanus*	B-SATH	Native
		Brown thrasher	*Toxostoma rufum*	B-BRTH	Casual
	Sturnidae	European starling	*Sturnus vulgaris*	B-EUST	Introduced
		Crested mynah	*Acridotheres cristellatus*	B-CRMY	Extirpated
	Motacillidae	Eastern yellow wagtail	*Motacilla tschutchensis*	B-YEWG	Casual
		Black-backed wagtail	Motacilla alba (lugens)	B-BBWA	Accidental
		Sprague's pipit	Anthus spragueii	B-SPPI	Casual
		Red-throated pipit	*Anthus cervinus*	B-RTPI	Casual
		American pipit	*Anthus rubescens*	B-AMPI	Native
	Bombycillidae	Bohemian waxwing	*Bombycilla garrulus*	B-BOWA	Native
		Cedar waxwing	*Bombycilla cedrorum*	B-CEWA	Native
	Parulidae	Tennessee warbler	*Vermivora peregrina*	B-TEWA	Native
		Orange-crowned warbler	*Vermivora celata*	B-OCWA	Native
		Nashville warbler	*Vermivora ruficapilla*	B-NAWA	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Northern parula	*Parula americana*	B-NOPA	Casual
		Yellow warbler	*Dendroica petechia*	B-YEWA	Native
		Chestnut-sided warbler	*Dendroica pensylvanica*	B-CSWA	Very rare
		Magnolia warbler	*Dendroica magnolia*	B-MGNW	Native
		Cape May warbler	*Dendroica tigrina*	B-CMWA	Native
		Black-throated blue warbler	*Dendroica caerulescens*	B-BTBW	Casual
		Yellow-rumped warbler	*Dendroica coronata*	B-YRWA	Native
		Black-throated gray warbler	*Dendroica nigrescens*	B-BTGW	Native
		Hermit warbler	*Dendroica occidentalis*	B-HEWA	Casual
		Black-throated green warbler	*Dendroica virens*	B-BTNW	Native
		Townsend's warbler	*Dendroica townsendi*	B-TOWA	Native
		Blackburnian warbler	*Dendroica fusca*	B-BBWA	Accidental
		Yellow-throated warbler	*Dendoica dominica*	B-YTWA	Accidental
		Prairie warbler	*Dendroica discolor*	B-PRWA	Casual
		Palm warbler	*Dendroica palmarum*	B-PAWA	Native
		Bay-breasted warbler	*Dendroica castanea*	B-BAYW	Native
		Blackpoll warbler	*Dendroica striata*	B-BKPW	Native
		Black-and-white warbler	*Mniotilta varia*	B-BAWW	Native
		American redstart	*Setophaga ruticilla*	B-AMRE	Native
		Ovenbird	*Seiurus aurocapillus*	B-OVEN	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Northern waterthrush	*Seiurus noveboracensis*	B-NOWA	Native
		Connecticut warbler	*Oporornis agilis*	B-COWA	Native
		Mourning warbler	*Oporornis philadelphia*	B-MOWA	Native
		MacGillivray's Warbler	*Oporornis tolmiei*	B-MACW	Native
		Common yellowthroat	*Geothlypis trichas*	B-COYE	Native
		Hooded warbler	*Wilsonia citrina*	B-HOWA	Casual
		Wilson's warbler	*Wilsonia pusilla*	B-WIWA	Native
		Canada warbler	*Wilsonia canadensis*	B-CAWA	Native
		Painted redstart	*Myioborus pictus*	B-PARE	Accidental
		Yellow-breasted chat	*Icteria virens*	B-YBCH	Native
	Thraupidae	Scarlet tanager	*Piranga olivacea*	B-SCTA	Accidental
		Western tanager	*Piranga ludoviciana*	B-WETA	Native
	Emberizidae	Green-tailed towhee	*Pipilo chlorurus*	B-GTTA	Casual
		Spotted towhee	*Pipilo maculatus*	B-SPTO	Native
		American tree sparrow	*Spizella arborea*	B-ATSP	Native
		Chipping sparrow	*Spizella passerina*	B-CHSP	Native
		Clay-colored sparrow	*Spizella pallida*	B-CCSP	Native
		Brewer's sparrow	*Spizella breweri*	B-BRSP	Native
		Vesper sparrow	*Pooecetes gramineus*	B-VESP	Native
		Lark sparrow	*Chondestes grammacus*	B-LASP	Native
		Black-throated sparrow	*Amphispiza bilineata*	B-BTSP	Very rare
		Sage sparrow	*Amphispiza belli*	B-SASP	Casual

Order	Family	Common Name	Latin Name	BC Code	Status
		Lark bunting	*Calamospiza melanocorys*	B-LKBU	Very rare
		Savannah sparrow	*Passerculus sandwichensis*	B-SAVS	Native
		Grasshopper sparrow	*Ammodramus savannarum*	B-GRSP	Native
		Baird's sparrow	*Ammodramus bairdii*	B-BASP	Casual
		Le Conte's sparrow	*Ammodramus leconteii*	B-LCSP	Native
		Nelson's sharp-tailed sparrow	*Ammodramus nelsoni*	B-NSTS	Native
		Fox sparrow	*Passerella iliaca*	B-FOSP	Native
		Song sparrow	*Melospiza melodia*	B-SOSP	Native
		Lincoln's sparrow	*Melospiza lincolnii*	B-LISP	Native
		Swamp sparrow	*Melospiza georgiana*	B-SWSP	Native
		White-throated sparrow	*Zonotrichia albicollis*	B-WTSP	Native
		Harris's sparrow	*Zonotrichia querula*	B-HASP	Native
		White-crowned sparrow	*Zonotrichia leucophrys*	B-WCSP	Native
		Golden-crowned sparrow	*Zonotrichia atricapilla*	B-GCSP	Native
		Dark-eyed junco	*Junco hyemalis*	B-DEJU	Native
		McCown's longspur	*Calcarius mccownii*	B-MCLO	Casual
		Lapland longspur	*Calcarius lapponicus*	B-LALO	Native
		Smith's longspur	*Calcarius pictus*	B-SMLO	Native
		Chestnut-collared longspur	*Calcarius ornatus*	B-CCLO	Casual
		Rustic bunting	*Emberiza rustica*	B-RUBU	Accidental
		McKay's bunting	*Plectrophenax hyperborus*	B-MCBU	Accidental
		Snow bunting	*Plectrophenax nivalis*	B-SNBU	Native

Order	Family	Common Name	Latin Name	BC Code	Status
	Cardinalidae	Rose-breasted grosbeak	*Pheucticus ludovicianus*	B-RBGR	Native
		Black-headed grosbeak	*Pheucticus melanocephalus*	B-BHGR	Native
		Blue grosbeak	*Guiraca caerulea*	B-BLGB	Casual
		Lazuli bunting	*Passerina amoena*	B-LZBU	Native
		Indigo bunting	*Passerina cyanea*	B-INBU	Very rare
		Dickcissel	*Spiza americana*	B-DICK	Casual
	Icteridae	Bobolink	*Dolichonyx oryzivorus*	B-BOBO	Native
		Red-winged blackbird	*Agelaius phoeniceus*	B-RWBL	Native
		Western meadowlark	*Sturnella neglecta*	B-WEME	Native
		Yellow-headed blackbird	*Xanthocephalus xanthocephalus*	B-YHBL	Native
		Rusty blackbird	*Euphagus carolinus*	B-RUBL	Native
		Brewer's blackbird	*Euphagus cyanocephalus*	B-BRBL	Native
		Common grackle	*Quiscalus quiscula*	B-COGR	Native
		Great-tailed grackle	*Quiscalus mexicanus*	B-GTGR	Accidental
		Brown-headed cowbird	*Molothrus ater*	B-BHCO	Native
		Bullock's oriole	*Icterus bullockii*	B-BUOR	Native
		Baltimore oriole	*Icterus galbula*	B-BAOR	Native
		Orchard oriole	*Icterus spurius*	B-OROR	Accidental
		Hooded oriole	*Icterus cucullatus*	B-HOOR	Casual
	Fringillidae	Brambling	*Fringilla montifringilla*	B-BRAM	Native
		Gray-crowned rosy-finch	*Leucosticte tephrocotis*	B-GCRF	Native
		Pine grosbeak	*Pinicola enucleator*	B-PIGR	Native

Order	Family	Common Name	Latin Name	BC Code	Status
		Purple finch	*Carpodacus purpureus*	B-PUFI	Native
		Cassin's finch	*Carpodacus cassinii*	B-CAFI	Native
		House finch	*Carpodacus mexicanus*	B-HOFI	Native
		Red crossbill	*Loxia curvirostra*	B-RECR	Native
		White-winged crossbill	*Loxia leucoptera*	B-WWCR	Native
		Common redpoll	*Carduelis flammea*	B-CORE	Native
		Hoary redpoll	*Carduelis hornemanni*	B-HORE	Native
		Pine siskin	*Carduelis pinus*	B-PISI	Native
		Lesser goldfinch	*Carduelis psaltria*	B-LEGO	Casual
		American goldfinch	*Carduelis tristis*	B-AMGO	Native
		Evening grosbeak	*Coccothraustes vespertinus*	B-EVGR	Native
	Passeridae	House sparrow	*Passer domesticus*	B-HOSP	Introduced
		Eurasian tree sparrow	*Passer montanus*	N/A	Unconfirmed

TABLE I-16-2: British Columbia Species Inventory for Mammals

Order	Family	Common Name	Latin Name	BC Code	Status
Soricomorpha	Soricidae	Black-backed shrew, Arctic shrew	*Sorex arcticus*	M-SOAR	Native
	Soricidae	Pacific water shrew, Marsh shrew	*Sorex bendirii*	M-SOBE	Native
	Soricidae	Common shrew, Cinereus shrew	*Sorex cinereus*	M-SOCI	Native
	Soricidae	American pygmy shrew	*Sorex hoyi*	M-SOHO	Native
	Soricidae	Merriam's shrew	*Sorex merriami*	M-SOME	Native
	Soricidae	Dusky shrew	*Sorex monticolus*	M-SOMO	Native
	Soricidae	American water shrew, common water shrew	*Sorex palustris*	M-SOPA	Native
	Soricidae	Preble's shrew	*Sorex preblei*	M-SOPR	Native
	Soricidae	Trowbridge's shrew	*Sorex trowbridgii*	M-SOTR	Native
	Soricidae	Tundra shrew	*Sorex tundrensis*	M-SOTU	Native
	Soricidae	Vagrant shrew	*Sorex vagrans*	M-SOVA	Native
	Talpidae	American Shrew-Mole	*Neurotrichus gibbsii*	M-NEGI	Native
	Talpidae	Coast mole	*Scapanus orarius*	M-SCOR	Native
	Talpidae	Townsend's mole	*Scapanus townsendii*	M-SCTO	Native
Chiroptera	Vespertilionidae	Pallid bat	*Antrozous pallidus*	M-ANPA	Native
	Vespertilionidae	Townsend's big-eared bat	*Corynorhinus townsendii*	M-COTO	Native
	Vespertilionidae	Big brown bat	*Eptesicus fuscus*	M-EPFU	Native
	Vespertilionidae	Spotted bat	*Euderma maculatum*	M-EUMA	Native
	Vespertilionidae	Silver-haired bat	*Lasionycteris noctivagans*	M-LANO	Native

Order	Family	Common Name	Latin Name	BC Code	Status
	Vespertilionidae	Western red bat	*Lasiurus blossevillii*	M-LABL	Native
	Vespertilionidae	Hoary bat	*Lasiurus cinereus*	M-LACI	Native
	Vespertilionidae	California myotis	*Myotis californicus*	M-MYCA	Native
	Vespertilionidae	Western small-footed myotis	*Myotis ciliolabrum*	M-MYCI	Native
	Vespertilionidae	Long-eared myotis, western long-eared myotis	*Myotis evotis*	M-MYEV	Native
	Vespertilionidae	Keen's myotis, keen's long-eared myotis	*Myotis keenii*	M-MYKE	Native
	Vespertilionidae	Little brown myotis	*Myotis lucifugus*	M-MYLU	Native
	Vespertilionidae	Northern myotis, northern long-eared myotis	*Myotis septentrionalis*	M-MYSE	Native
	Vespertilionidae	Fringed myotis	*Myotis thysanodes*	M-MYTH	Native
	Vespertilionidae	Long-legged myotis	*Myotis volans*	M-MYVO	Native
	Vespertilionidae	Yuma myotis	*Myotis yumanensis*	M-MYYU	Native
Carnivora	Canidae	Coyote	*Canis latrans*	M-CALA	Native
	Canidae	Grey wolf	*Canis lupus*	M-CALU	Native
	Canidae	Red fox	*Vulpes vulpes*	M-VUVU	Native
	Felidae	Domestic cat, Feral cat	*Felis catus*		Introduced
	Felidae	Lynx, Canadian lynx	*Lynx canadensis*	M-LYCA	Native
	Felidae	Bobcat	*Lynx rufus*	M-LYRU	Native
	Felidae	Cougar	*Puma concolor*	M-PUCO	Native
	Mephitidae	Striped skunk	*Mephitis mephitis*	M-MEME	Native
	Mephitidae	Western spotted skunk	*Spilogale gracilis*	M-SPGR	Native
	Mustelidae	Sea otter	*Enhydra lutris*	M-ENLU	Native
	Mustelidae	Wolverine	*Gulo gulo*	M-GUGU	Native

Order	Family	Common Name	Latin Name	BC Code	Status
	Mustelidae	North American river otter	*Lontra canadensis*	M-LOCA	Native
	Mustelidae	American marten	*Martes americana*	M-MAAM	Native
	Mustelidae	Fisher	*Martes pennanti*	M-MAPE	Native
	Mustelidae	Ermine, short-tailed weasel	*Mustela erminea*	M-MUER	Native
	Mustelidae	Long-tailed weasel	*Mustela frenata*	M-MUFR	Native
	Mustelidae	Least weasel	*Mustela nivalis*	M-MUNI	Native
	Mustelidae	American mink	*Neovison vison*	M-NEVI	Native
	Mustelidae	American badger	*Taxidea taxus*	M-TATA	Native
	Otariidae	Northern fur seal	*Callorhinus ursinus*	M-CAUR	Native
	Otariidae	Northern sea lion, Steller sea lion	*Eumetopias jubatus*	M-EUJU	Native
	Otariidae	California sea lion	*Zalophus californianus*	M-ZACA	Native
	Phocidae	Northern elephant seal	*Mirounga angustirostris*	M-MIAN	Native
	Phocidae	Harbour seal	*Phoca vitulina*	M-PHVI	Native
	Procyonidae	Raccoon	*Procyon lotor*	M-PRLO	Native
	Ursidae	American black bear	*Ursus americanus*	M-URAM	Native
	Ursidae	Grizzly bear, brown bear	*Ursus arctos*	M-URAR	Native
	Balaenidae	North Pacific right whale	*Eubalaena japonica*	M-EUJA	Rare
Cetacea	Balaenopteridae	Common minke whale	*Balaenoptera acutorostrata*	M-BAAC	Native
	Balaenopteridae	Sei whale	*Balaenoptera borealis*	M-BABO	Native
	Balaenopteridae	Blue whale	*Balaenoptera musculus*	M-BAMU	Rare
	Balaenopteridae	Fin whale	*Balaenoptera physalus*	M-BAPH	Native

Order	Family	Common Name	Latin Name	BC Code	Status
	Balaenopteridae	Humpback whale	*Megaptera novaeangliae*	M-MENO	Native
	Delphinidae	Long-beaked common dolphin	*Delphinus capensis*		Very rare
	Delphinidae	Short-beaked common dolphin	*Delphinus delphis*		Very rare
	Delphinidae	Short-finned pilot whale	*Globicephala macrorhynchus*	M-GLMA	Native
	Delphinidae	Risso's dolphin	*Grampus griseus*	M-GRGR	Rare
	Delphinidae	Pacific white-sided dolphin	*Lagenorhynchus obliquidens*	M-SAOB	Native
	Delphinidae	Northern right whale dolphin	*Lissodelphis borealis*	M-LIBO	Rare
	Delphinidae	Killer whale	*Orcinus orca*	M-OROR	Native
	Delphinidae	False killer whale	*Pseudorca crassidens*		Very rare
	Delphinidae	Striped dolphin	*Stenella coeruleoalba*	M-STCO	Rare
	Eschrichtiidae	Grey whale	*Eschrichtius robustus*	M-ESRO	Native
	Phocoenidae	Harbour porpoise	*Phocoena phocoena*	M-PHPH	Native
	Phocoenidae	Dall's porpoise	*Phocoenoides dalli*	M-PHDA	Native
	Physeteridae	Pygmy sperm whale	*Kogia breviceps*	M-KOBR	Rare
	Physeteridae	Dwarf sperm whale	*Kogia sima*	M-KOSI	Rare
	Physeteridae	Sperm whale	*Physeter macrocephalus*	M-PHMA	Native
	Ziphiidae	Baird's beaked whale, northern Pacific bottlenosed whale	*Berardius bairdii*		Very rare
	Ziphiidae	Hubb's beaked whale, arch-beaked whale	*Mesoplodon carlhubbsi*		Very rare
	Ziphiidae	Stejneger's beaked whale, Bering sea beaked	*Mesoplodon stejnegeri*		Very rare

Order	Family	Common Name	Latin Name	BC Code	Status
		whale			
	Ziphiidae	Cuvier's beaked whale, goose-beaked whale	*Ziphius cavirostris*	M-ZICA	Rare
Didelphimorphia	Didelphidae	North American opossum, Virginia opossum	*Didelphis virginiana*		Introduced
Artiodactyla	Bovidae	Wood Bison	*Bos bison athabascae*	M-BOBI-AT	Native
	Bovidae	Plains Bison	*Bos bison bison*	M-BOBI-BI	Native
	Bovidae	Mountain goat	*Oreamnos americanus*	M-ORAM	Native
	Bovidae	Bighorn sheep	*Ovis canadensis*	M-OVCA	Native
	Bovidae	Thinhorn sheep (Dall sheep, Stone sheep)	*Ovis dalli*	M-OVDA	Native
	Cervidae	Moose	*Alces alces americanus*	M-ALAM	Native
	Cervidae	Elk, Wapiti	*Cervus canadensis*	M-CECA	Native
	Cervidae	Red deer	*Cervus elaphus*		Introduced
	Cervidae	Fallow deer	*Dama dama*		Introduced
	Cervidae	Mule deer, Black-tailed deer	*Odocoileus hemionus*	M-ODHE	Native
	Cervidae	White-tailed deer	*Odocoileus virginianus*	M-ODVI	Native
	Cervidae	Caribou	*Rangifer tarandus*	M-RATA	Native
Perissodactyla	Equidae	Horse	*Equus caballus*		Introduced
Rodentia	Aplodontiidae	Mountain beaver	*Aplodontia rufa*	M-APRU	Native
	Myocastoridae	Coypu, nutria	*Myocastor coypus*		Introduced
	Sciuridae	Northern flying squirrel	*Glaucomys sabrinus*	M-GLSA	Native
	Sciuridae	Hoary marmot	*Marmota caligata*	M-MACA	Native
	Sciuridae	Yellow-bellied marmot	*Marmota flaviventris*	M-MAFL	Native

Order	Family	Common Name	Latin Name	BC Code	Status
	Sciuridae	Woodchuck	*Marmota monax*	M-MAMO	Native
	Sciuridae	Vancouver Island marmot	*Marmota vancouverensis*	M-MAVA	Native
	Sciuridae	Yellow-pine chipmunk	*Neotamias amoenus*	M-TAAM	Native
	Sciuridae	Least chipmunk	*Neotamias minimus*	M-TAMI	Native
	Sciuridae	Red-tailed chipmunk	*Neotamias ruficaudus*	M-TARU	Native
	Sciuridae	Townsend's chipmunk	*Neotamias townsendii*	M-TATO	Native
	Sciuridae	Eastern grey squirrel	*Sciurus carolinensis*		Introduced
	Sciuridae	Eastern fox squirrel	*Sciurus niger*		Introduced
	Sciuridae	Columbian ground squirrel	*Spermophilus columbianus*	M-SPCO	Native
	Sciuridae	Golden-mantled ground squirrel	*Spermophilus lateralis*	M-SPLA	Native
	Sciuridae	Arctic ground squirrel	*Spermophilus parryii*	M-SPPA	Native
	Sciuridae	Cascade mantled ground squirrel	*Spermophilus saturatus*	M-SPSA	Native
	Sciuridae	Douglas' squirrel	*Tamiasciurus douglasii*	M-TADO	Native
	Sciuridae	Red squirrel	*Tamiasciurus hudsonicus`*	M-TAHU	Native
	Castoridae	American beaver	*Castor canadensis*	M-CACA	Native
	Geomyidae	Northern pocket gopher	*Thomomys talpoides*	M-THTA	Native
	Heteromyidae	Great Basin pocket mouse	*Perognathus parvus*	M-PEPA	Native
	Dipodidae	Meadow jumping mouse	*Zapus hudsonius*	M-ZAHU	Native
	Dipodidae	Western jumping mouse	*Zapus princeps*	M-ZAPR	Native
	Dipodidae	Pacific jumping mouse	*Zapus trinotatus*	M-ZATR	Native

Order	Family	Common Name	Latin Name	BC Code	Status
	Cricetidae	Nearctic brown lemming	*Lemmus trimucronatus*	M-LETR	Native
	Cricetidae	Long-tailed vole	*Microtus longicaudus*	M-MILO	Native
	Cricetidae	Montane vole	*Microtus montanus*	M-MIMO	Native
	Cricetidae	Tundra vole, root vole	*Microtus oeconomus*	M-MIOE	Native
	Cricetidae	Creeping vole	*Microtus oregoni*	M-MIOR	Native
	Cricetidae	Meadow vole	*Microtus pennsylvanicus*	M-MIPE	Native
	Cricetidae	North American water vole	*Microtus richardsoni*	M-MIRI	Native
	Cricetidae	Townsend's vole	*Microtus townsendii*	M-MITO	Native
	Cricetidae	Southern red-backed vole	*Myodes gapperi*	M-MYGA	Native
	Cricetidae	Northern red-backed vole	*Myodes rutilus*	M-MYRU	Native
	Cricetidae	Bushy-tailed woodrat	*Neotoma cinerea*	M-NECI	Native
	Cricetidae	Common muskrat	*Ondatra zibethicus*	M-ONZI	Native
	Cricetidae	Keen's mouse, Northwestern deer mouse	*Peromyscus keeni*	M-PEKE	Native
	Cricetidae	Deer mouse, North American deer mouse	*Peromyscus maniculatus*	M-PEMA	Native
	Cricetidae	Western Heather Vole	*Phenacomys intermedius*	M-PHIN	Native
	Cricetidae	Western harvest mouse	*Reithrodontomys megalotis*	M-REME	Native
	Cricetidae	Northern bog lemming	*Synaptomys borealis*	M-SYBO	Native
	Erethizontidae	North American porcupine	*Erethizon dorsatum*	M-ERDO	Native
	Muridae	House mouse	*Mus musculus*		Introduced
	Muridae	Norway rat, brown rat, ship rat	*Rattus norvegicus*		Introduced

Order	Family	Common Name	Latin Name	BC Code	Status
	Muridae	Black rat, roof rat	*Rattus rattus*		Introduced
Lagomorpha	Leporidae	Snowshoe hare	*Lepus americanus*	M-LEAM	Native
	Leporidae	White-tailed jackrabbit	*Lepus townsendii*	M-LETO	Native
	Leporidae	European rabbit	*Oryctolagus cuniculus*		Introduced
	Leporidae	Eastern cottontail	*Sylvilagus floridanus*		Introduced
	Leporidae	Nuttall's cottontail, mountain cottontail	*Sylvilagus nuttallii*	M-SYNU	Native
	Ochotonidae	Collared pika	*Ochotona collaris*	M-OCCO	Native
	Ochotonidae	American pika, common pika	*Ochotona princeps*	M-OCPR	Native

TABLE I-16-3: British Columbia Species Inventory for Reptiles

Group	Common Name	Latin Name	BC Code	Status
Freshwater Turtles	Western pond turtle	*Clemmys (Actinemys) marmorata*	R-EMMA	Extirpated
	Western painted turtle	*Chrysemys picta*	R-CHPI	Native
	Asiatic (Chinese) turtle	*Chinemys reevesi*		Introduced
	Pond slider	*Trachemys scripta*		Introduced
	Snapping turtle	*Chelydra serpentina*		Introduced
Marine Turtles	Green turtle	*Chelonia mydas*		Accidental
	Leatherback sea turtle	*Dermochelys coriacea*	R-DECO	Native
	Olive Ridley turtle	*Lepidochelys olivacea*		Possible
Lizards	Pigmy short-horned lizard	*Phrynosoma douglasii*	R-PHDO	Extirpated
	Common wall lizard	*Podarcis muralis*		Introduced
	Western skink	*Eumeces skiltonianus*	R-EUSK	Native
	Northern alligator lizard	*Elgaria coerulea*	R-ELCO	Native
Snakes	Rubber boa	*Charina bottae*	R-CHBO	Native
	Sharp-tailed snake	*Contia tenuis*	R-COTE	Native
	Night snake	*Hypsiglena torquata*	R-HYTO	Native
	Racer	*Coluber constrictor*	R-COCO	Native
	Gopher Snake, *catenifer* subspecies	*Pituophis catenifer catenifer*	R-PICA-CA	Native
	Gopher Snake, *deserticola* subspecies	*Pituophis catenifer deserticola*	R-PICA-DE	Native
	Common garter snake	*Thamnophis sirtalis*	R-THSI	Native
	Northwestern garter snake	*Thamnophis ordinoides*	R-THOR	Native
	Western terrestrial garter snake	*Thamnophis elegans*	R-THEL	Native
	Western rattlesnake	*Crotalus oreganus*	R-CROR	Native

TABLE I-16-4: British Columbia Species Inventory for Amphibians

Order	Common Name	Latin Name	BC Code	Status
Caudata (Salamanders)	Long-toed salamander	*Abystoma macrodactylum*	A-AMMA	Native
	Northwestern salamander	*Ambystoma gracile*	A-AMGR	Native
	Tiger salamander	*Ambystoma tigrinum*	A-AMTI	Native
	Coastal giant salamander	*Dicamptodon tenebrosus*	A-DITE	Native
	Rough-skinned newt	*Taricha granulosa*	A-TAGR	Native
	Ensatina	*Ensatina eschscholtzii*	A-ENES	Native
	Coeur d'Alene salamander	*Plethodon idahoensis*	A-PLID	Native
	Western red-backed Salamander	*Plethodon vehiculum*	A-PLVE	Native
	Wandering salamander	*Aneides vagrans*	A-ANVA	Native
Anura (Frogs and Toads)	Coast tailed frog	*Ascaphus truei*	A-ASTR	Native
	Rocky Mountain tailed frog	*Ascaphus montanus*	A-ASMO	Native
	Great Basin spadefoot toad	*Spea intermontana*	A-SPIN	Native
	Western toad	*Bufo boreas*	A-BUBO	Native
	Pacific treefrog (Pacific chorus frog)	*Hyla (Pseudacris) regilla*	A-PSRE	Native
	Boreal chorus frog	*Pseudoacris maculata*	A-PSMA	Native
	Red-legged frog	*Rana aurora*	A-RAAU	Native
	Columbia spotted frog	*Rana luteiventris*	A-RALU	Native
	Oregon spotted frog	*Rana pretiosa*	A-RAPR	Native
	Wood frog	*Rana sylvatica*	A-RASY	Native
	Northern leopard frog	*Rana pipiens*	A-RAPI	Native
	Green frog	*Rana (Lithobates) clamitans*		Introduced
	Bullfrog	*Rana catesbiana (= Lithobates catesbianus)*		Introduced
	Cascades frog	*Rana cascadae*		Possible

TABLE I-16-5: British Columbia Species Inventory for Freshwater Fish

Group	Common Name	Latin Name	Code	Comment/Status
Lampreys (Pteromyzontidae)	River lamprey	*Lampetra ayresii*	F-LAAY	The relationship between this species and the western brook lamprey (*L. richadsoni*) is unclear. They may be anadromous and non-migratory forms of the same species (not unlike the anadromous and freshwater-resident forms of threespine sticklebacks). If the river lamprey is a separate species, its spawning sites are unknown.
	Cowichan Lake lamprey	*Lampetra macrostoma*	F-LAMA	This species occurs only in Cowichan and Mesachie lakes. Its validity is questionable. It is clearly a non-migratory form derived from the anadromous Pacific lamprey. The major distinctions between this species and the Pacific lamprey are freshwater residence and freshwater feeding. Other Pacific lamprey populations are known to feed in fresh water but have not been studied.
	Western brook lamprey	*Lampetra richardsoni*	F-LARC	This species may be a non-migratory form of the river lamprey (*L. ayresii*). If they are the same species, the brook lamprey has taxonomic priority.
	Pacific lamprey	*Lampetra tridentata*	F-LATR	The relationships of non-migratory, freshwater populations of this species need clarification. Additionally, the Upper Fraser River population(s) needs to be examined. Also, there is strong evidence that the Pacific lamprey does not belong in the genus *Lampetra*.
Sturgeons (Acipenseridae)	Green sturgeon	*Acipenser medirostris*	F-ACME	The green sturgeon is an occasional visitor to BC marine and estuarine waters. There is no evidence that this species breeds in either the Columbia or Fraser river systems even though it is seasonally common in the Columbia estuary. The source of both the Columbia and our green sturgeons appears to be the Klamath River.
	White sturgeon	*Acipenser transmontanus*	F-ACRT	The white sturgeon is in serious decline in the Nechako, Upper Columbia, and Kootenay rivers. The Kootenay population is genetically distinctive. The apparent head shape differences between the Fraser River populations above and below the Fraser Canyon need to be quantified and clarified.
Herrings (Clupeidae)	American shad	*Alosa sapidissima*		Introduced into MOE Region 2
Salmonids (Salmonidae)	Cisco	*Coregonus artedi*	F-COAR	Although the cisco is widespread across the northern Great Plains, only one population is known from BC (Maxhamish Lake). This lake, and its fishes, has been studied and the cisco population appears to be healthy. There is some debate about the relationship of this species to Arctic cisco.

Group	Common Name	Latin Name	Code	Comment/Status
	Arctic cisco	*Coregonus autumnalis*	F-COAU	Normally, the Arctic cisco is an anadromous species. About 30 years ago a spawning run was intercepted in the lower Liard River and, apparently, spawned below the Liard Canyon. It is not known if this is an annual event. There is little work done in this area and nothing is known about their biology, or status, in BC.
	Lake whitefish	*Coregonus clupeaformis*	F-COCL	The lake whitefish is native to central and northern BC. The taxonomy of this complex of species is still confused. We may have one or, perhaps, two species in BC. In addition, there is a genetically divergent form that apparently survived the last glaciation in the Nahanni glacial refuge.
	Broad whitefish	*Coregonus nasus*	F-CONA	The only known population of broad whitefish in BC is in Teslin Lake. Nothing is known about its biology or interactions with the lake whitefish. The only published reports on this species in North America are from Alaska and the Northwest Territories. Most of the reports refer to anadromous populations and the Teslin fish represent an isolated freshwater resident population. Two whitefish species (broad whitefish and inconnu) in Teslin Lake harbours are isolated lacustrine populations that warrant more study.
	Least cisco	*Coregonus sardinella*	F-COSR	The least cisco occurs in the BC portion of the Yukon drainage system. Elsewhere in its range, this species sometimes occurs as two sympatric trophic forms. There is a hint of such a situation in one unnamed lake in northern BC. The lake is remote and there were too few fish in the original samples to be certain of their population structure. Nothing is published on the ecology or behaviour of BC populations of least ciscoes.
	Pygmy whitefish	*Prosopium coulterii*	F-PRCO	The pygmy whitefish is usually characterized as a glacial relict. Typically it occurs in deep oligitrophic lakes and most BC populations fit this description; however, two BC populations occur in eutrophic lakes and grow to a giant size (for pygmies). A third BC population occurs in a shallow (max. depth 25 m) mesotrophic lake with a depauperate fauna (originally only three fish species). In this lake, pygmy whitefish are very abundant and show some unusual behaviour (*e.g.*, schooling, nocturnal use of the littoral zone). It is an enigmatic species with little known about its biology.
	Round whitefish	*Prosopium cylindraceum*	F-PRCY	This species survived glaciation in the Mississippi and Bering refugia. All the published information on the biology of round whitefish is on the eastern North American form. Our populations are confined to the upper Yukon, Taku, and upper Liard systems. They are derived from Beringia and undoubtedly are genetically distinct from the eastern form. We know little about their biology but Dease Lake is the only place in the world where all three of the widespread species of *Prosopium* co-exist.

Group	Common Name	Latin Name	Code	Comment/Status
	Mountain whitefish	*Prosopium williamsoni*	F-PRWI	The biology of riverine mountain whitefish is reasonably well known. They have a complex life cycle and make major spawning, over-wintering, and summer feeding migrations. In contrast, the biology of lacustrine populations is poorly known. We do know that there are major differences in body form between riverine and lacustrine populations and, that in Kootenay Lake there are spatially and temporally separate spawning runs. This suggests that in large lakes there may be multiple demes. Additionally, in many interior rivers there are two riverine forms — a normal form and a longnose "pinocchio" form. The two forms differ in foraging behaviour, morphology and there is some evidence of genetic differences.
	Inconnu	*Stenodus leucichthys*	F-STLU	The distribution of inconnu in BC is disjunct — there is a freshwater resident population in Teslin Lake and a migratory population in the Liard River system. The Liard population is biologically complex: some individuals appear to be part of the general upper Mackenzie River population and probably is not anadromous but other individuals are known to migrate to the Mackenzie Delta, and other tagged individuals have been taken in the Beaufort Sea. There is some evidence that some of the Liard population breeds in BC; however, no fry have been collected in BC although one juvenile was collected in the Ft. Nelson River. In contrast, the Teslin Lake population is thought to breed in the lake and appears to be isolated from the migratory Yukon populations. As far as is known, the Teslin population is the only lacustrine population in the Yukon system. In itself, this makes it unusual. Additionally, all the Yukon populations appear to be separated from the Mackenzie populations by a substantial distributional gap along the north coast of Alaska. This suggests that the two BC groups of *Stenodus* may be genetically different.
	Coastal cutthroat trout	*Oncorhynchus clarkii clarkii*	F-ONCL-CL	In BC, the coastal cutthroat occurs as three major life history forms — sea-run populations, freshwater-resident populations (lacustrine and fluvial) and headwater stream populations. Within each of these groups there are complex arrays of life history variants. For example, although sea-run cutthroats typically return to freshwater to overwinter, in the Bella Coola system there was once a run of exceptionally large cutthroat that may have spent a year or more in the sea. This run now appears to be extinct. The complexity and inter- and intra-population variation in coastal cutthroat life histories presents a challenge to conservation managers. It will be difficult to maintain biodiversity in this subspecies. Unlike most of the hybridization between rainbows and cutthroats this population may be a natural hybrid swarm that has existed since before European colonization.

Group	Common Name	Latin Name	Code	Comment/Status
	Westslope cutthroat trout	*Oncorhynchus clarkii lewisi*	F-ONCL-LE	The native range of the westslope cutthroat is the southeast part of the province (especially the Kootenay drainage system). Generally, its life history is not as complex as the coastal subspecies. There are, however, "dwarf" headwater populations and also some in high mountain lakes. At one time, some of these small-bodied populations were recognized as separate subspecies but there is no good evidence that they warrant taxonomic recognition. The biggest threat to the westslope cutthroat is the introduction of rainbow trout into areas where westslope cutthroat are native. Such introductions inevitably lead to massive hybridization and the loss of the "pure" westslope genome. The process has gone so far in the northwestern U.S. that pure westslope cutthroats are almost gone. In BC, there are enough isolated populations above barriers that "pure" populations probably will survive; however, further introductions of rainbows into areas that are occupied by westslope cutthroat would be ecological vandalism. The few headwater populations in some Eagle River tributaries are the only native populations of westslope cutthroat in the Fraser River system.
	Rainbow trout	*Oncorhynchus mykiss*	F-ONMY	The rainbow trout is the most common and popular trout in BC. In the past, biodiversity in this species has been compromised by fish culture operations but in recent years the emphasis has shifted to protecting the remaining wild stocks. In BC, rainbow trout occur both as freshwater-resident and anadromous (steelhead) populations. Some authors recognize two subspecies of *O. mykiss* and both appear to occur in British Columbia — the coastal rainbow trout, *O. mykiss irideus* and, in the interior, the Columbia redband trout, *O. mykiss gairdneri*. However, subspecies should represent monophyletic clusters of populations. Consequently, although the coastal and interior forms of rainbow trout generally are treated as two distinct lineages, subspecific names are rarely used. In BC, recent molecular studies support the notion of two rainbow trout clades. Although the geography of these lineages roughly supports a coastal-interior dichotomy, there is extensive overlap in their distributions. Thus, while many BC sites include both lineages, the coastal clade is dominant on the coast and the interior clade is dominant in southern Interior populations. Because the molecular markers characteristic of the two clades occur in both coastal and interior populations, the boundaries between the purported subspecies are fuzzy and confound attempts to assign formal subspecific names to the two clades. Typically, the considerable life history variation in this species (*e.g.*, differences in run-timing, body size, and

Group	Common Name	Latin Name	Code	Comment/Status
				foraging behaviour) is present in both clades. Nonetheless, some life history types are relatively rare (*e.g.*, large, piscivorous forms) and special efforts should be made to preserve them. Many of the anadromous (steelhead) populations on the south coast are in jeopardy but managers are well aware of the problems.
	Golden trout	*Oncorhynchus mykiss aguabonita*		Introduced into MOE Regions 2, 4 and 8. Continued presence unknown.
	Pink salmon	*Oncorhynchus gorbuscha*	F-ONGO	There are still unsolved problems involving the relationships and distribution of the odd and even year broodlines in this species. Because virtually all pink salmon mature at two years, the generations spawning on odd and even years are genetically isolated from one another and often differ in life history and genetic characteristics. Usually one broodline is dominant (*i.e.*, there is a strong run one year followed by a much smaller run the next year). At the southern end of their North American distribution (including southern British Columbia) odd year runs are dominant but, in BC, north of the Fraser River system, many rivers support relatively strong runs on both odd and even years. From the Queen Charlotte Islands north into Alaska, even year runs are dominant. Presumably, odd and even year runs have evolved independently in different areas but the reasons for the broad geographic pattern in run-dominance is still a mystery. The pattern of mitochondrial variation in northern pink salmon indicates multiple Pleistocene divergences followed by a relatively recent (postglacial) expansion from different sources and, perhaps, different colonization routes for the odd and even year broodlines.
	Chum salmon	*Oncorhynchus keta*	F-ONKE	The chum salmon is still abundant along the BC coast. There are some interesting life history variants in this species — temporally separated runs to the same small streams are common and some populations spawn intertidally. Of special interest are two northern runs. Most chum runs spawn within 100 km of the sea but there is a run of chum salmon in the Yukon system that reaches Teslin Lake (>2,000 km from the sea). Unlike southern chums, these Yukon fish are bright when they enter freshwater and in good condition when they reach their spawning grounds. The other northern run of potential interest

Group	Common Name	Latin Name	Code	Comment/Status
				is in the Liard River (Mackenzie system). Chum salmon have reached the lower Liard River in BC; however, it is not clear that there is a self-sustaining run in the Liard. Interestingly, this chum run was reported the same year as the Arctic cisco run into BC.
	Coho salmon	*Oncorhynchus kisutch*	F-ONKI	The coho salmon is genetically heterogeneous and locally adapted populations are common in this species. Much of this local adaptation is associated with small populations in small streams. This biodiversity is threatened by hatchery operations (genetic swamping) and the practice of basing management decisions on a few, large populations.
	Sockeye salmon	*Oncorhynchus nerka*	F-ONNE	The sockeye salmon is also a genetically heterogeneous species and locally adapted populations are common. In the past, kokanee were often referred to as a subspecies, *Oncorhynchus nerka kenneryli*. We now know that most natural kokanee populations evolved from different populations of anadromous sockeye. Because shared common ancestors is a crucial component in defining any taxon, and the kokanee life-history form is clearly polyphyletic, it is inappropriate to assign the same subspecific name to all kokanee populations. This does not mean that kokanee are simply small sockeye. Some kokanee populations spawn sympatrically (*i.e.*, in the same stream and at the same time) as anadromous sockeye but still retain a suite of inherited morphological, physiological, and behavioural differences from sockeye. Populations where kokanee and sockeye are sympatric for part of their life history are scientifically important and some should be protected. Also, there are lakes (*e.g.*, Okanagan Lake) where two, or more populations, spawn in different habitats (*i.e.*, beach and stream spawners). These situations are of scientific interest. There is also a mysterious population of deep-bodied kokanee in Seton Lake. Apparently, they spawn late in the year and at great depth.

Group	Common Name	Latin Name	Code	Comment/Status
	Chinook salmon	*Oncorhynchus tshawytscha*	F-ONTS	This is another genetically heterogeneous species and locally adapted populations are common. Apparently, much of the variation within the species is derived from the presence of two behavioural forms of chinooks — a "stream type" and an "ocean type". Stream type chinooks have a relatively long period of freshwater residence (one or more years), at sea they make major offshore migrations, and they return to their natal rivers in the spring or summer. In contrast, ocean type chinook usually migrate to sea within about three months of emergence; they spend most of their ocean life in inshore waters, and they return to their natal streams in the fall.
	Brown trout	*Salmo trutta*		Introduced; stocked in MOE Regions 1 and 8
	Atlantic salmon	*Salmo salar*		Introduced; aquacultures escapees in MOE Regions 1 and 2
	Bull trout	*Salvelinus confluentus*	F-SACO	The relationship between bull trout and Dolly Varden has a long and complex history. In BC, the bull trout is primarily an interior species; however, it reaches the coast wherever large rivers cut through the Coast Mountains. Again, there are a number of life history types — stream-residents, large bodied fluvial and adfluvial populations, and even a few anadromous (or perhaps, more properly, amphidromous) populations. These populations that migrate to estuaries appear to be unique to southern British Columbia but probably at one time also occurred in the Puget Sound region of Washington State. Where they come together, bull trout and Dolly Varden commonly hybridize; however, even in the face of persistent hybridization (and back-crossing) they maintain themselves as distinct ecological and genetic entities.
	Dolly Varden	*Salvelinus malma*	F-SAMA	The relationship between Dolly Varden and bull trout has a long and complex history. In BC, the Dolly Varden is a coastal species. It occurs in most rivers and streams along the length of the coast but is more common on the north coast than on the south coast. There are at least three life history forms — stream-resident, adfluvial and lacustrine, and sea-run populations. On Vancouver Island, sea-run Dolly Varden appear to end at about Campbell River. Dolly Varden have crossed the Coast Mountains in at least three areas — the middle Fraser, upper Peace, and upper Liard systems. There are no bull trout on Vancouver Island and the Dolly Varden in some of the island's large lakes have adopted a bull trout-like life history as deep-water piscivores. Where they come together, Dolly Varden and bull trout commonly hybridize; however, even in the face of persistent hybridization (and back-crossing) they maintain themselves as distinct ecological and genetic entities.

Group	Common Name	Latin Name	Code	Comment/Status
	Lake trout	*Salvelinus namaycush*	F-SANA	There is evidence that, during the Pleistocene, lake trout survived (and diverged) in at least five separate refugia. The BC populations are derived from two refugia: those in the Yukon, Chilkat, Taku, and Stikine systems, and, perhaps, in the upper Liard are derived from the Bering Refugium, whereas those in the Skeena, Fraser, Peace and lower Liard systems are derived from eastern sources. In addition, lake trout from eastern North America have been introduced into BC. Again, from a biodiversity perspective, it is important to distinguish between indigenous and introduced populations. Any populations south of Shuswap Lake are probably introduced.
	Brook trout	*Salvelinus fontinalis*		Introduced; stocked in all MOE regions
	Arctic grayling	*Thymallus arcticus*	F-THAR	Arctic grayling occur in both North America and Siberia. As the name implies, in North America this coldwater species ranges in a broad band from the west coast of Hudson Bay to Alaska. They are absent from the Arctic Archipelago but, historically, isolated populations occurred in upper Michigan and in the upper Missouri system in Montana. Arctic grayling appear to be especially vulnerable to over-fishing and habitat changes. At one time they were the most abundant recreational fish in the upper Peace system; however, since the formation of the Williston Reservoir they have dramatically declined in this region. After the fact, we have learned more about their life history and, especially, about the importance of large, valley-bottom rivers as over-wintering sites. Elsewhere in northern BC, wherever road access allows anglers into grayling waters, grayling appear to be in decline. Current management practices are designed to stop these declines but it is too early to know whether they are working.
Smelts (Osmeridae)	Surf smelt	*Hypomesus pretiosus*		The surf smelt is a marine or estuarine species; however, in the lower Fraser River it commonly occurs upstream as far as Queen's Reach, and it has been collected as far upstream as the upper end of Pitt Lake. Also, young surf smelts were taken in a tow-net sets in Queen's Reach. Probably, these young were entrained in tidal water moving upstream. Nonetheless, there is a possibility that this species spawns in the tidal portions of the Fraser River.
	Arctic smelt	*Osmerus dentex*		There are unconfirmed reports of Arctic smelt in some north-coast estuaries and there is a marine record from Barclay Sound (based on a desiccated juvenile specimen). In Alaska, it has not been recorded south of the Alaska Peninsula. Consequently, it is unlikely that this species occurs in the fresh waters of BC.

Group	Common Name	Latin Name	Code	Comment/Status
	Longfin smelt	*Spirinchus thaleichthys*	F-SPTH	There are records of longfin smelt from the central and north coasts but the only known BC spawning run is in the lower Fraser River. It is not clear how far upstream they migrate but young-of-the-year longfin smelts have been taken near Chilliwack (Island 22). Apparently, there are two spawning runs into the river — one in late August and September, and one in November. The data suggest that the runs are discrete and the fish differ in body size and spawning area. Additionally, there are pygmy, neotenic smelts in both Pitt and Harrison lakes. The relationships between these freshwater resident smelts and the anadromous populations are unknown; however, in Pitt Lake they are seasonally sympatric. The relationships between neotenic and anadromous longfin smelts are currently under study.
	Eulachon	*Thaleichthys pacificus*	F-THPA	Spawning runs of eulachon are known from most major rivers along the BC coast. This fish is particularly significant to the coastal first nations and appears to be in serious decline in the Fraser River.
Mooneyes (Hiodontidae)	Goldeye	*Hiodon alosoides*	F-HIAL	The status of this species in BC needs to be clarified. Only small numbers of goldeye are taken in BC and they may be occasional wanderers from downstream populations in Alberta. Juveniles are present in the Fort Nelson River, however, and there may be a breeding population in the BC portion of the Liard system.
Pikes (Esocidae)	Northern pike	*Esox lucius*	F-ESLU	Pike are abundant in suitable habitats in the northeastern part of the province (Mackenzie River system) and in the upper Yukon system. Their geographic distribution in the province suggests our populations were derived from two sources: the Beringian and Great Plains refugia. Genetically, they probably are slightly divergent.
Minnows Cyprinidae)	Chiselmouth	*Acrocheilus aleutaceus*	F-ACAL	The fragmented distribution of chiselmouth in BC suggests that the species was more widely distributed in the past. The Kettle River population is isolated above Cascade Falls and may be slightly divergent. The hybrids (including backcrosses) between chiselmouth and pikeminnows in Missezula Lake provide a potentially interesting problem in trophic ecology - how do hybrids between a periphytonscraper and a predator make a living?
	Lake chub	*Couesius plumbeus*	F-COPL	This is the most cold-adapted minnow in North America. The southern edge of the species' range closely corresponds to the southern margin of glaciation. Most of the BC populations in the south are now extinct. It is not clear why they disappeared, although many of the small lakes in this area were "rehabilitated" in the 1950s and 1960s.

Group	Common Name	Latin Name	Code	Comment/Status
	Brassy minnow	*Hybognathus hankinsoni*	F-HYHA	The biology of this small fish is not well understood. It is especially abundant in the lower Fraser Valley and headwater ponds and small lakes in the Prince George area. In between it is exceedingly rare. In the lower Fraser Valley it appears to migrate to and from the main river; however, its movements are largely uncharted. Seasonally, it turns up in large numbers at some sites and then disappears. The populations in the Esker Provincial Park appear to be extinct. Probably victims of an exotic species (brook trout) introduced to create a recreational fishery.
	Northern pearl dace	*Margariscus margarita*	F-MAMA	The northern pearl dace only occurs in the northeastern portion of the province. Here, its distribution is spotty. It is widespread in boggy habitats in eastern North America and across the northern Great Plains. Curiously, all of the largest known specimens of this species are from the extreme northwestern margin of its range (BC). In some BC lakes pearl dace are involved in a three-way hybrid swarm with finescale and northern redbelly dace. Given the propensity of hybrids between the latter two species to give rise to all female, diploid and triploid clones, the three-way cross might produce some interesting offspring. There is evidence that the northern pearl dace is species distinct from the southern pearl dace. If so, the name of the local species should be revised to *Margariscus nachtriebi*.
	Peamouth	*Mylocheilus caurinus*	F-MYCA	The peamouth is the only primary freshwater fish on Vancouver Island and the Sechelt Peninsula. Its presence on both the west and east coasts of Vancouver Island is a minor biogeographic puzzle that probably could be solved with a microsatellite study. On the mainland, peamouth normally occur with a suite of minnows and suckers with which it has coevolved. These fish are absent on Vancouver Island.
	Emerald shiner	*Notropis atherinoides*	F-NOAT	The emerald shiner was collected once in BC (in a small tributary to the Fort Nelson River). It is not clear if this species breeds in BC or even if there is a BC population; however, in other provinces it is characterized as a large river fish.
	Spottail Shiner	*Notropis hudsonius*	F-NOHU	There is one indigenous population of this species in BC (Maxhamish Lake) but it has been introduced (from Alberta) as a forage fish into Charlie Lake. From Charlie Lake, the spottail minnow has spread into other Peace River tributaries. The Maxhamish population is native, but the other populations are not.

Group	Common Name	Latin Name	Code	Comment/Status
	Northern redbelly dace	*Phoxinus eos*	F-PHEO	Pure populations of the northern redbelly dace are rare in BC. In the few places where this species is known to occur, it usually hybridizes with the finescale dace. Elsewhere (including adjacent lakes in Alberta), this hybrid combination produces diploid and triploid all-female "species".
	Finescale dace	*Phoxinus neogaeus*	F-PHNE	The finescale dace is widely distributed in the northeastern part of the province. Its biology is not well known and the products of its hybridization with the northern redbelly dace are of considerable scientific interest.
	Flathead chub	*Platygobio gracilis*	F-PLGR	The flathead chub is probably the least studied — but abundant — species of freshwater fish in North America. Again, it is a fish of our large, turbid, northern rivers. Its reproductive biology, life history, and habitat use are largely unknown.
	Northern pikeminnow	*Ptychocheilus oregonensis*	F-PTOR	The northern pikeminnow is something of an anomaly — a large, predaceous minnow. Its biology has been reasonably well studied but always with the aim of "controlling" its numbers. The "dwarf" pikeminnows in some lakes on the Bonaparte Plateau may be unique to BC, but are probably stunted introduced populations.
	Longnose dace	*Rhinichthys cataractae*	F-RHCA	This widespread species is remarkably uniform across North America, except in British Columbia. In BC, there are three forms of longnose dace — the typical Great Plains form in northeastern BC, the typical Columbia-Fraser form in the rest of the province, and the Nooksack dace in the extreme southwestern region. The first two forms differ substantially in their reproductive biology and, genetically, they are quite (>4%) divergent. They may be different species. Similarly, the mitochondrial DNA of the Nooksack dace is >2% divergent from that of the Columbia-Fraser longnose dace. Although the Nooksack dace is abundant in western Washington State, it is seriously threatened by urban development in BC
	Leopard dace	*Rhinichthys falcatus*	F-RHFA	The leopard dace is a Columbia system endemic. It is abundant in gravel deposition reaches along the Fraser River. Curiously, with the exception of the lower Similkameen River, it is not common in the Columbia system. Its rarity in most of the Columbia system may be natural or may be a result of human intervention (dams). Although its general ecology is modestly well known, its reproductive biology is unknown. It is one of the species involved in the evolution, through an ancient hybridization event, of the Umatilla dace. The genetic relationships between leopard and Umatilla dace need more study — based on mitochondrial analyses, some populations group with Umatilla dace rather than with their own species. This may reflect past hybridization.

Group	Common Name	Latin Name	Code	Comment/Status
	Speckled dace	*Rhinichthys osculus*	F-RHOS	The Kettle River is the only place in Canada where the speckled dace occurs; however, it is widespread in the western U.S. Within the Kettle River system, it is widely distributed and abundant. Nonetheless, its biology in BC is poorly understood. Since it is on the COSEWIC list, an effort should be made to document its life history and quantify its habitat use. To determine if the BC population is actually unique, a molecular study of its relationships with other middle Columbia drainage populations is needed.
	Umatilla dace	*Rhinichthys umatilla*	F-RHUM	The Umatilla dace has only recently been recognized as a separate species. It is endemic to the Columbia River system and is thought to be the product of an early Pleistocene hybridization involving leopard and speckled dace. Very little is known about its habitat requirements. There are two forms of Umatilla dace in BC — one in the Columbia and Slocan rivers and the other in the Similkameen River. There are subtle differences in their morphology and some evidence of molecular differentiation. The relationship between the two forms and their habitat requirements needs study.
	Redside shiner	*Richardsonius balteatus*	F-RIBA	The redside shiner is the most common minnow in BC. There are consistent differences in body shape between riverine and lacustrine populations. How these differences affect habitat use is unknown.
	Carp	*Cyprinus carpio*		Introduced in MOE Regions 1 ,2, 3, 4 and 8
	Goldfish	*Carassius auratus*		Introduced in MOE Regions 1, 2, 3, 4 and 8
	Tench	*Tinca tinca*		Introduced in MOE Regions 4 and 8
	Fathead minnow	*Pimephales pomelas*		Introduced into MOE Regions 2 and 7
Suckers (Catastomidae)	Longnose sucker	*Catostomus catostomus*	F-CACT	This is the most widely distributed sucker in British Columbia. There are "dwarf" populations scattered around the province. The ecological factors associated with these populations of small suckers are unknown but in at least one lake (now rehabilitated) small-bodied and large-bodied longnose suckers once coexisted. In BC, one small-bodied form - the Salish sucker - is restricted to the lower Fraser Valley. Genetically and morphologically, it is slightly, but consistently, different from other northwestern longnose suckers.
	Bridgelip sucker	*Catostomus columbianus*	F-CACO	The bridgelip sucker is another Columbia system endemic. Ecologically, it bridges the trophic gap between the largescale sucker and the mountain sucker — it is less dependent on periphyton than the mountain sucker but is not as clearly a general benthivore as the largescale sucker. Interestingly, much of its anatomy is also intermediate between these species. Little is known about its reproductive biology or the details of its habitat use.

Group	Common Name	Latin Name	Code	Comment/Status
	White sucker	*Catostomus commersonii*	F-CACM	The white sucker is an eastern North American species that postglacially colonized the upper Fraser and Skeena drainage systems. Where the largescale and white sucker co-exist, they often hybridize. In eastern North America, the biology of this species is well studied and, presumably, its biology is similar in BC
	Largescale sucker	*Catostomus macrocheilus*	F-CAMA	The largescale sucker probably is the most common sucker in the southern half of our province. It grows to a large size and appears to be morphologically and ecologically uniform over most of its BC range. In the summer, the population in Eagle Lake (Chilcotin) is reported to forage at the surface on emerging chironomids.
	Mountain sucker	*Catostomus platyrhynchus*	F-CAPL	The mountain sucker is the most specialized sucker in BC. It is a periphyton scraper and has a chisel-like lower jaw. Like the chiselmouth, its BC distribution is scattered - it occurs in the Fraser River between Hope and Chilliwack, the North Thompson near Heffley, the Similkameen River near Keremeos, and perhaps the Salmo River near its junction with the Pend d'Oreille River. Little is known about its biology in BC, and BC mountain suckers are about 5% divergent (mtDNA) from those on the Great Plains (Alberta and Saskatchewan). The two published accounts of mountain sucker life history are both from east of the Continental Divide and probably refer to a different species.
Catfishes (Ictaluridae)	Black bullhead	*Ameiurus melas*		Introduced into MOE Regions 4 and 8
	Brown bullhead	*Ameiurus nebulosus*		Introduced into MOE Regions 1, 2 and 4
Cods (Gadidae)	Burbot	*Lota lota*	F-LOLO	In North America, burbot survived glaciation in multiple refugia and different morphological forms of this species now occur in different regions. At least two subspecies have been recognized — one in Siberia, Alaska, parts of the Yukon, and in northern BC, and the other on the Great Plains and eastern North America. Regardless of whether or not forms derived from different refugia warrant subspecific recognition, BC probably was colonized from both refugia. Consequently, genetically, our northern and southern burbot populations probably are different. Whether these differences translate into life history or habitat differences is unknown. The northern populations appear to be healthy but some of our southern populations are in trouble. For example, the once thriving burbot population in Kootenay Lake is almost gone and other Columbia system populations are also be in decline.

Group	Common Name	Latin Name	Code	Comment/Status
Sticklebacks	Brook stickleback	*Culaea inconstans*	F-CUIN	In BC, the brook stickleback is found in the northeastern part of the province. It is extremely abundant in muskeg areas and occupies a number of major drainages (although they are all part of the Mackenzie River system).
	Threespine stickleback	*Gasterosteus aculeatus*	F-GAAC	The threespine stickleback is notorious for the complexity of its morphological, ecological, and behavioural forms. These forms often are sympatric or parapatric and, in many cases, they act like good biological species (*i.e.*, they are reproductively isolated and use different trophic and spatial resources). To further confuse matters, these forms tend to evolve repeatedly. They are of great scientific interest and a challenge for biodiversity managers. A rule of thumb for prioritizing the different forms of *Gasterosteus* for protection is to examine their geographic distributions. For example, the anadromous stream-resident dichotomy is widespread in Europe, Asia, and both coasts of North America. This suggests that the conditions that produce this dichotomy are widespread. Consequently, although local examples may be lost the dichotomy is unlikely to go extinct. In contrast, the benthic-limnetic dichotomy only occurs in BC (although it has been searched for elsewhere). This suggests that the conditions that lead to this dichotomy are rare, local, and unique. Thus, this dichotomy has a higher biodiversity value than the anadromous stream-resident dichotomy and warrants more rigorous protection than the other dichotomy.
	Ninespine stickleback	*Pungitius pungitius*	F-PUPU	The ninespine stickleback may not breed in BC. Only four specimens have been documented. Three came from the Petitot River just west of the Alberta border and one came from the Fort Nelson River just downstream from old Fort Nelson. The Petitot River specimens probably drifted downstream from Bistcho Lake in Alberta. Although no breeding fish were taken in the Petitot, the region close to the Alberta border has only been collected once. The Fort Nelson fish is more puzzling. It was taken hundreds of kilometers from any known self-sustaining population. If it was a stray from the nearest known source (Bistcho Lake) it had to swim down the Petitot River to the Liard River and then upstream against the current to the Fort Nelson area,.a formidable journey against a strong current. A simpler explanation is that there is some unknown, but nearby, source. If so, there maybe a self-sustaining BC population.
Trout-perches (Percopsidae)	Trout-perch	*Percopsis omiscomaycus*	F-PEOM	The trout-perch is an archaic small fish. In BC, it only occurs east of the Continental Divide. Although it has not been studied extensively in our province, it appears to be doing well.

Group	Common Name	Latin Name	Code	Comment/Status
Sunfishes (Centrarchidae)	Largemouth bass	*Micropterus salmoides*		Introduced into MOE Regions 2,4 and 8
	Smallmouth bass	*Micropterus dolomieui*		Introduced; stocked in MOE Regions 1, 4 and 8
	Black crappie	*Pomoxis nigromaculatus*		Introduced into MOE Regions 2 and 8
	Pumpkinseed	*Lepomis gibbosus*		Introduced into MOE Regions 1, 2, 4 and 8
Perches	Yellow perch	*Perca flavescens*	F-PEFL	Most BC populations of this species are introduced; however, it is possible that the Swan Lake population near Tupper in the Peace Region is native. It is indigenous to Peace River drainages in adjacent Alberta and was first recorded from Swan Lake in BC in the 1950s.
	Walleye	*Sander vitreus*	F-SAVI	The walleye is indigenous to the northeast corner of BC but has been introduced into the Columbia River system.
Sculpins (Cottidae)	Spinynose Sculpin	*Asemichthys taylori*	F-ASTA	Marine species may enter fresh water near river mouths
	Coastrange sculpin	*Cottus aleuticus*	F-COAL	As its name implies, the coastrange sculpin is a coastal species. In southern BC, it rarely penetrates more than 150 km inland. Reproductive adults in minor coastal drainages migrate downstream and spawn just above estuaries. The larvae over-winter in brackish water before moving upstream. On Vancouver Island and, presumably on the mainland, the larvae of populations associated with large lakes are swept down into the lakes and live limnetically for an unknown length of time before migrating back into streams. The "dwarf" adults in Cultus Lake are probably derived from this life history. It is not known if the lake and stream populations in Cultus Lake represent separate gene pools or if some limnetic larvae stay behind in the lake. Morphology, however, suggests that the lake form is neotenic. Although this species does not occur above the Fraser Canyon, there is a disjunct population associated with Anderson and Seton Lakes. On the central coast (*e.g.*, Skeena system), coastrange sculpins penetrate over 500 km inland (Morrison Lake).

Group	Common Name	Latin Name	Code	Comment/Status
	Prickly sculpin	*Cottus asper*	F-COAS	There are two morphological forms of the prickly sculpin in BC, and a number of life history types. There is a coastal and an interior form that, morphologically, are slightly different. Additionally, preliminary mtDNA data suggest a modest divergence between the two forms. Presumably the coastal form dispersed into BC through the sea — many coastal populations are catadromous — while the interior entered from the unglaciated portions of the Columbia system (a pattern common to other species; *e.g.*, chinook salmon, rainbow trout). In catadromous populations, adults migrate downstream and spawn in estuaries. The larvae spend at least a year in the estuary before migrating upstream. The biology of the non-seagoing interior form has not been studied extensively and there may be significant life history difference between the two forms.
	Slimy sculpin	*Cottus cognatus*	F-COCO	The slimy sculpin is the most widely distributed sculpin in BC. Distributional and morphological evidence suggests that the province was postglacially colonized from three sources — the Columbia River, the northern Great Plains, and Beringia. Except in the southern part of the province, the populations are generally healthy. With one exception (a geographically isolated population in central Idaho), the southern populations of this species closely coincide with the maximum extent of glaciation. This raises the possibility that they did not survive glaciation in the Columbia system but postglacially entered Columbia drainages from the north. The slimy sculpin is a coldwater species and most populations in southern BC are found in glacial streams or cool headwaters. Nevertheless, there are populations above barriers (where they are the only sculpin) that occupy relatively warm streams (*e.g.*, the Kettle River). It is common to find slimy sculpins, by themselves, above barriers in the BC portion of the Columbia system. They tolerate warmer conditions if they are the only sculpin but appear to be excluded where warm-water sculpins occur (*e.g.*, Columbia, torrent, and Rocky Mountain sculpins).
	Shorthead sculpin	*Cottus confusus*	F-COCN	In BC, the shorthead sculpin occurs in Columbia River tributaries below Bonnington Falls and in the three km of the Kettle River below Cascade Falls. Their life history has not been studied extensively in BC (early BC reports refer to the Rocky Mountain sculpin in the Flathead River as the shorthead sculpin). Most of the existing BC populations appear to be strong and in no immediate danger; however, Blueberry Creek near Castlegar is being encroached upon by development. Additionally, Brilliant Dam divides the Slocan populations from those below the dam.

Group	Common Name	Latin Name	Code	Comment/Status
	Columbia sculpin	*Cottus hubbsi*	F-COBA-HU	The Columbia sculpin is endemic to the Columbia River system. In BC, it occurs in the Columbia River and tributaries (*e.g.*, the Slocan, Kootenay (below Bonnington Falls), Kettle (below Cascade Falls), and Similkameen rivers. In the Castlegar-Trail area it is rare in small tributaries but common in the mainstem Columbia. The tributary streams in this area are dominated by shorthead sculpins in their lower reaches. In contrast, Columbia sculpins are abundant in small tributaries to the Similkameen (below Similkameen Falls) and Tulameen rivers. These habitat shifts may reflect interspecific interactions (there are no shorthead sculpins in the Similkameen system). The only known lacustrine populations of this species occurred in a series of small lakes in the Allison Creek drainage (Similkameen system). Unfortunately, the lakes were rehabilitated before any data was collected on these fish. Some sculpins appear to have modified life histories in lakes (see comments under torrent sculpin). There are lakes in the Otter Creek drainage (also Similkameen system) where Columbia sculpins are abundant in streams above and below the lakes. There is another sculpin in the upper Otter Creek area that may be the Columbia sculpin but, morphologically, some individuals fit the description of the Malheur mottled sculpin, *Cottus bendirei*. These upper Otter Creek individuals also differ in their mitochondrial sequence from the Columbia sculpin.
	Torrent sculpin	*Cottus rhotheus*	F-CORH	The torrent sculpin occurs throughout the Columbia River system and in the North Thompson River. Normally, this species is heavily prickled but there are two populations in BC that lack prickles — one in Pass (Norns) Creek and one in Beaver Creek. In both cases these populations are isolated above barriers. There are no other sculpins at these sites. As their name implies torrent sculpins are usually associated with fast water; however, it is not clear if this is by choice or a result of interactions with other sculpins. In areas where they coexist with shorthead sculpins (*e.g.*, the Little Slocan River), they appear to shift into quiet water. There are also lacustrine populations of torrent sculpin. (<10 mm) found in open water off the bottom. They are transparent and remain in the plankton until they reach about 15 mm (newly emerged stream dwellers are typically 10-12 mm). Presumably, this lacustrine population metamorphoses at about 15-20 mm and settles to the bottom.
	Spoonhead sculpin	*Cottus ricei*	F-CORI	In BC, the spoonhead sculpin only occurs in the northeastern part of the province. In eastern North America it is described as a lacustrine species that often occurs in deep water. In BC, it is another species associated with large, turbid, northern rivers. Little is known about its biology in these habitats.

Group	Common Name	Latin Name	Code	Comment/Status
	Rocky mountain sculpin	*Cottus sp.*		This is the sculpin found in the lower Flathead River. It has been variously called the shorthead sculpin (*C. confusus*), the mottled sculpin (*C. bairdi*), and another sculpin (*C. punctulatus*), but it is none of the above. Recent molecular studies show that it is a yet undescribed species. The same species occurs in southwestern Alberta and the upper Missouri system in Montana. Its common name is the Rocky Mountain sculpin. This COSEWIC listed species is threatened by extensive coal mine development in southeastern BC.

APPENDIX II

MODELLING TOOLS

TABLE OF CONTENTS

MODELLING TOOL #1

1.0 LITERATURE-BASED BIOACCUMULATION/BIOCONCENTRATION FACTORS AND UPTAKE MODELS

What does this tool consist of? Internal concentrations of chemicals in organisms can be related to concentrations in their ambient environment. For most chemicals at relatively low ambient concentrations (*i.e.*, typically encountered in the environment), the ratio of internal to ambient concentrations ($C_{internal}/C_{ambient}$) is assumed to be independent of ambient concentration (*i.e.*, constant ratio). In these cases, chemical accumulation is expressed as a bioconcentration factor (BCF) or bioaccumulation factor (BAF). BCFs and BAFs are simply ratios of the organism tissue concentration to the concentration in the water (for aquatic organisms) or soil (for terrestrial organisms). BAFs are often lipid-normalized for organic contaminants.

- The BCF is intended to reflect the tendency of a chemical to accumulate in a species via passive diffusion, according to equilibrium partitioning. BCFs are measured in a laboratory, under conditions of water exposure only (*i.e.*, no dietary exposure).

- The BAF is intended to reflect the tendency of a chemical to accumulate in a species via all routes, including passive diffusion from the environment and uptake from the diet. BAFs may be measured in the lab, but are more commonly measured in the field.

- The BSAF (biota-sediment accumulation factor) is a closely-related approach applied to sediment-associated contaminants. In a BSAF, the contaminant concentration is typically normalized to organic carbon content in sediment and lipid content in organisms (see Modeling Tool #8).

For some inorganic chemicals, the ratio $C_{internal}/C_{ambient}$ has been observed to vary with $C_{ambient}$. The form of this relationship, often called an "uptake model", is described in documents such as Sample *et al.* (1999); see Appendix II-13. These uptake models are used to predict contaminant concentrations in various media evaluated within an ecological risk assessment.

The tendency of a chemical to bioconcentrate, bioaccumulate, and/or biomagnify depends many factors, including:

- Physical and chemical properties of the chemical, including solubility in water and in lipid, molecular weight, degradation (transformation in the environment);

- Physical and chemical properties of the exposure medium, including temperature, dissolved and particulate organic carbon, water hardness, suspended solids in water, organic carbon content of soil, particle size distribution of soil, pH, redox potential, *etc.*; and,

- The nature of the organism, including its ability to metabolize or excrete the chemical, lipid composition, size, and dietary factors (feeding rate, diet composition, and dietary assimilation efficiency).

Which ecosystem(s) would this tool typically be applied in? All ecosystems.

How frequently is this tool used in a DERA? Common in both aquatic and terrestrial ecosystems.

What are the benefits of using this tool in a DERA?

- Literature-based BCFs/BAFs and uptake models make it possible to estimate the tissue COPC concentrations for an organism using data on COPC concentrations in environmental media, without collection of site-specific tissue data.

- Literature-based BCFs/BAFs are relatively inexpensive to obtain, requiring literature review rather than application on site-specific bioaccumulation tests. As such, they are often applied during screening phases using conservative (upper-bound) estimates.

- For substances without BCF/BAF data, it is sometimes possibly to extrapolate from similar compounds based on consideration of physical properties of the substances (USEPA, 1999).

- For BSAFs, there is a theoretical basis (equilibrium partitioning) for a restricted range of values for contaminants that partition preferentially to carbon or lipid pools. A theoretical value of 1.7 for BSAFs has been estimated based on partitioning of nonionic organic compounds between sediment carbon and tissue lipids (ASTM, 1997; Lee, 1992). The mechanistic basis for BSAF for certain classes of hydrophobic organic contaminants (*e.g.*, PCBs, pesticides) increases the confidence in the BSAFs.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- The greatest weakness of literature-based BAFs and BCFs is that they do not incorporate site-specific factors that govern the environmental fate of the contaminants of interest (*i.e.*, differences in both the physicochemical environment as well as the interaction of organisms with their environment). Accordingly, there is a wide range of BAFs/BCFs and high uncertainty associated with their application.

- Different organisms respond differently to both essential and non-essential elements. For example decapods usually regulate internal concentrations of iron, copper, and zinc, but accumulate concentrations of lead and cadmium. Conversely, amphipods and barnacles are net accumulators of all of the above elements, to varying degrees. Differences in homeostatic regulation, assimilation efficiency of contaminants, and feeding mechanisms all contribute significant variability to literature-based bioaccumulation factors.

- BCFs may underestimate exposure for COPCs primarily absorbed from the diet (*e.g.*, chemicals with low solubility in water). The BCF may greatly underestimate exposure to biomagnifying COPCs. BAFs are always preferred to BCFs unless the purpose of the study is to evaluate uptake via respiration only.

- Laboratory-derived BAFs are usually based on maximum chemical bioavailability (*i.e.*, low dissolved organic carbon, highly digestible food, *etc*.), and may therefore overestimate exposure to COPCs under field conditions that reduce bioavailability. Field-derived BAFs may have the same limitation if there are differences in bioavailability between the system in which the BAF was measured and the system in which the DERA is being conducted (*i.e.*, may over- or underestimate exposure, depending on which system has higher bioavailability).

- Laboratory-derived BCFs/BAFs are not always measured over a long enough period for the animal to approach steady state, and may therefore underestimate the degree of bioaccumulation that will occur under real-world conditions.

- BCFs/BAFs are not available for all species. They should only be extrapolated between species that are very similar with respect to bioaccumulation, especially with respect to their ability to metabolize the chemical and their trophic position in the food web.

Where can I go for additional information about this tool?

- ASTM (American Society for Testing and Materials). 1997. Standard guide for determination of bioaccumulation of sediment-associated contaminants by benthic invertebrates. El688-97a. In: *ASTM Annual Book of Standards*, Vol. 11.05, American Society for Testing and Materials, Philadelphia, PA, pp. 1072-1121.

- Bechtel Jacobs Company. 1998. *Empirical Models for the Uptake of Inorganic Chemicals from Soil by Plants*. U. S. Department of Energy, Oak Ridge, TN. Available at: http://www.esd.ornl.gov/programs/ecorisk/documents/bjcor-133.pdf

- Lee, II. 1992. Models, muddles and mud. In: *Sediment Toxicity Assessment*, Ed. G.A. Burton, Lewis Publishers, pp. 267-293.

- Parkerton, T.F. 1993. *Estimating Toxicokinetic Parameters for Modeling the Bioaccumulation of Non-ionic Organic Chemicals in Aquatic Organisms*. Ph.D. Dissertation. Submitted to the Graduate School, New Brunswick, Rutgers, State University of New Jersey. Graduate program in Environmental Science. May 1993.

- Sample, B.E., G. W. Suter II, J. J. Beauchamp, and R. A. Efroymson. 1999. Literature-derived bioaccumulation models for earthworms: development and validation. *Environ. Toxicol. Chem.* 18:2110-2120.

- Torres K.C. and Johnson M.L. 2001. Testing of metal bioaccumulation models with measured body burdens in mice. *Environ. Toxicol. Chem.* 20:2627-2638.

- Torres K.C. and M.L. Johnson. 2001. Bioaccumulation of metals in plants, arthropods, and mice at a seasonal wetland. *Environ. Toxicol. Chem.* 20:2617-2626.

- US EPA (United States Environmental Protection Agency). 1999. *Screening level ecological risk assessment protocol for hazardous waste combustion facilities*. EPA530-D-99-001. US Environmental Protection Agency, Washington, DC.

MODELLING TOOL #2

2.0 SITE-SPECIFIC BIOACCUMULATION FACTORS OR UPTAKE MODELS

What does this tool consist of? Co-occurring samples of soil and soil invertebrates are collected from the site and analyzed for the contaminants of potential concern. Other combinations of environmental media can also be sampled (*e.g.*, soil and plant tissue; sediment and benthic invertebrates).

- Bioaccumulation factors (BAFs) are determined for each pair of samples, and a summary of the range of BAFs calculated (*e.g.*, mean, 95% upper confidence limit of the mean; 90th percentile or maximum)

- Uptake models can be developed using regression analyses to fit an appropriate form of statistical model to the available co-occurring soil and tissue data (*e.g.*, linear, exponential, or power model). Multivariate analyses can be used to improve the predictive ability of these uptake models if data are also available for soil parameters that influence contaminant bioavailability (*e.g.*, soil pH; sediment AVS concentrations; organic carbon concentration).

The summary BAFs or uptake models are then used to predict tissue concentrations across the remainder of the site based on the available soil or sediment chemistry data.

Which ecosystem(s) would this tool typically be applied in? Uplands (including Wildlands) for soil measurements; Deep Aquatic or Rivers and Streams for sediment measurements.

How frequently is this tool used in a DERA? Common for upland terrestrial environments, and occasional for aquatic environments.

What are the benefits of using this tool in a DERA? Construction of a site-specific BAF or uptake model is a compromise between (a) the use of literature-based BAFs and uptake models (see Modelling Tool #1) and (b) collection of substantial numbers of tissue samples from across the site:

- Literature-based BAFs and uptake models are developed from a limited set of experimental conditions; their application represents a substantial source of uncertainty. This uncertainty can be as much as several orders of magnitude in terms of both over-predicting and under-predicting tissue concentrations. Developing a site-specific BAF or uptake model substantially reduces this uncertainty.

- For large sites, developing a site-specific BAF or uptake model is advantageous in that it reduces the sampling effort (and costs) that would be required to provide sufficient spatial coverage for tissue samples.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- The sampling used to develop the site-specific BAF or uptake model needs to reflect the full range of contaminant concentrations across the site. Bioaccumulation of many contaminants is dependent on concentration—a sampling program that focuses on worst-case areas is therefore not necessarily a conservative approach. Uptake models are frequently superior to BAFs because they facilitate consideration of the soil (or sediment) concentration in the resulting tissue predictions.

- Practitioners must consider measurement ancillary parameters (*e.g.*, soil type, particle size distribution, organic carbon content, *etc.*) that influence contaminant bioavailability in soil to facilitate development of multivariate uptake models. Risk assessors should be familiar with these factors for the applicable combinations of receptors and contaminants-of-concern.

- An inherent assumption in the collection of co-occurring samples is that the tissue items collected are also relatively immobile and in direct contact with the environmental media of interest. This assumption is not strictly true: earthworms and benthic invertebrates have some mobility and may be transported by drift from the area of exposure; root systems may extend over a considerable area.

- This approach is less suitable for highly mobile taxa that are not in close contact with the local exposure medium. Values based on weighted average exposure conditions can be derived, but contain additional uncertainty associated with the exposure assumptions.

- The uncertainty in the site-specific BAF or uptake model is strongly influenced by sample size. Determination of a minimum site-specific sample size should consider contaminant distribution, heterogeneity, seasonal effects, and size of the area of interest. A minimum sample size of 10 is recommended unless it can be demonstrated that a smaller sample size is appropriate. Note that minimum sample sizes of greater than 10 may be necessary depending on the factors discussed above.

- The practitioner must consider the confounding effect of soil particles in the tissue analyses. In most instances, the objective is to predict the bioaccumulation of contaminants within the tissue of the organism, and therefore, organisms should be well-rinsed (and blotted dry) as well as depurated (following approved protocols) to reduce the influence of this confounding factor.

Where can I go for additional information about this tool? Examples of one or more aspects of the issues discussed above can be found in the following peer-reviewed scientific literature:

- Efroymson, R.A., B.E. Sample, and G.W. Suter II. 2001. Uptake of inorganic chemicals for soil by plant leaves: regressions of field data. *Environ. Toxicol. Chem*. 20:2561-2571.

- Hunter, B.A., M.S. Johnson, and D.J. Thompson. 1987. Ecotoxicology of copper and cadmium in a contaminated grassland ecosystem. II. Invertebrates. *J. Appl. Ecol.* 24:587-599.

- Torres, K.C. and M.L. Johnson. 2001. Bioaccumulation of metals in plants, arthropods, and mice at a seasonal wetland. *Environ. Toxicol. Chem*. 20:2617-2626.

- Torres, K.C. and M.L. Johnson. 2001. Testing of metal bioaccumulation models with measured body burdens in mice. *Environ. Toxicol. Chem.* 20:2627-2638.

MODELLING TOOL #3

3.0 BIOMAGNIFICATION OR TROPHIC TRANSFER FACTORS

What does this tool consist of? The biomagnification factor (BMF) or trophic transfer factor (TTF) is the ratio of chemical concentration between a species and its diet. BMF usually refers to organic chemicals (usually lipid-normalized concentrations) whereas the TTF usually refers to metals. The food web magnification factor (FWMF) or trophic magnification factor (TMF; especially in Europe) is an expression of the average BMF across several trophic levels. All of these terms reflect the tendency of a substance to biomagnify (*i.e.*, increase in concentration at higher trophic levels).

Measured BMFs for many substances in many types of organisms are available in the literature. BMFs for metals are usually near or below 1.0 because most metals and metalloids do not biomagnify (mercury and selenium are notable exceptions). BMFs for organic substances range from well below 1.0 (*e.g.*, in poorly-absorbed or rapidly-metabolized chemicals) to values on the order of 10 for mid-level receptors (fish, invertebrates) to 100 or more for higher trophic-level organisms (birds, mammals).

Which ecosystem(s) would this tool typically be applied in? All ecosystems.

How frequently is this tool used in a DERA? Infrequent.

What are the benefits of using this tool in a DERA? Biomagnification factors can be used to assess the exposure of receptors to COPCs present in their diets. BMFs are often used in food web models to simulate exposure throughout the food web.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- BMFs are taxon- (and sometimes species-) specific. It may be possible to generalize among similar species within a higher taxon (*e.g.*, compile several BMFs for a chemical in fish, and use these to estimate the BMF for another fish species), but these cannot be used to estimate the BMF of an unrelated taxon (*e.g.*, extrapolate from fish to a bird species). Distinct taxa have different gut absorption efficiencies, experience different degrees of gastrointestinal magnification, and have very different capacities to metabolize and excrete various chemicals. These differences produce very large differences among species in BMFs.

- The same taxon may occupy different levels in the food web at different locations, depending on the availability of prey items and competitive pressures. See Direct Measurement Tool #12 for methods used to determine site-specific food webs.

- BMFs are chemical-specific. It is not recommended that a measured BMF for one chemical be used to estimate the BMF for another chemical. BMFs are highly sensitive to the metabolizability of the chemical, and this is difficult to predict from chemical structure.

- Accurately estimating BMFs from models requires information on metabolic potential, which is not often available.

Where can I go for additional information about this tool?

- Kelly, BC, McLachlan, M.S. and Gobas, F.A.P.C. 2004. Intestinal absorption and biomagnification of organic contaminants in fish, wildlife and humans. *Environ. Toxicol. Chem.* 23:2324-2336.

- Gobas, F.A.P.C. and J.B. Wilcockson 1999. Mechanism of biomagnification in fish under laboratory and field conditions. *Environ. Sci. Technol.* 33:133-141.

- Campbell L.M., A.T. Fisk, X. Wang, G. Köck and D.C.G .Muir. 2005. Evidence for biomagnification of rubidium in freshwater and marine food webs. *Can. J. Fish. Aquat. Sci.* 62:1161-1167.

MODELLING TOOL #4

4.0 MASS-BALANCE BIOACCUMULATION MODELS

What does this tool consist of? Mass-balance bioaccumulation models are mechanistic models used to estimate the bioaccumulation of chemicals in organisms. The basic form of the model is an individual-based chemical mass balance, balancing the sum of inputs (dietary uptake, respiratory absorption) against the sum of outputs (fecal egestion, respiratory elimination, metabolic transformation, growth dilution). These models are typically used to estimate steady-state concentrations (*i.e.*, by assuming that internal chemical concentrations are not changing over time), but the approach can also be used in a time-dependent formulation. Taxon-specific mass-balance bioaccumulation models have been developed for many species and higher taxa (*e.g.*, fish), and recently some general models have been developed that can be used for most wildlife species.

Which ecosystem(s) would this tool typically be applied in? All ecosystems.

How frequently is this tool used in a DERA? Occasional. Frequency of application increases for large, complex sites or highly detailed ERAs. Mass-balance models are applied more in aquatic environments than terrestrial environments, although extension of the bioaccumulation framework to birds and mammals has received increased attention in recent years.

What are the benefits of using this tool in a DERA? Mechanistic bioaccumulation models can be used to estimate internal concentrations of COPCs for any receptor. If it is not possible to directly measure internal concentrations (*e.g.*, the species is protected, inaccessible, or impractical to sample), mechanistic models may be the best way to obtain estimates of exposure. These models are adaptable in terms of site-specificity, depending on how much local information is available. These models have been validated in a wide variety of environments, and typically can predict internal concentrations with relatively good precision (often within a factor of 2-3 for average tissue concentrations).

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- Substantial information may be required to parameterize this model. The predictive ability of the model improves with increased information on site-specific chemical, physical, and biological parameters and processes.

- As with all mechanistic models, some direct chemical measurements are necessary to validate the model. In some cases, available chemistry data must be split into calibration and validation data sets. The benefits and limitations of direct measurements must be traded off against the model uncertainties.

- Many mechanistic models require some element of model calibration, which entails modification of parameters within plausible ranges identified via scientific assessment. The degree of predictive ability of the model is inversely related to the degree of calibration required to achieve acceptable model fit.

- Mechanistic bioaccumulation models are most reliable in conditions for which steady-state conditions apply and for which site fidelity of organisms is greatest. Models are more difficult to apply to situations where the exposure conditions vary substantially over time. For example, receptors that migrate extensively among contaminated and uncontaminated areas, or environments with rapidly changing contamination profiles (pulsed discharges, chemical spills), introduce challenges for mathematical modeling.

Where can I go for additional information about this tool?

- Arnot, J.A. and F.A. P. C. Gobas. 2004. A food web bioaccumulation model for organic chemicals in aquatic ecosystems. *Environ. Toxicol. Chem.* 23:2343-2355.

- Connolly, J.P. 1991. Application of a food chain model to polychlorinated biphenyl contamination of the lobster and winter flounder food chains in New Bedford Harbor. *Environ. Sci. Technol.* 25:760-770.

- Connolly, J.P. and C.J. Pedersen. 1988. A thermodynamic-based evaluation of organic-chemical accumulation in aquatic organisms. *Environ. Sci. Technol.* 22:99.

- Connolly, J.P., T.F. Parkerton, J.D. Quadrini, S.T. Taylor, and A.J. Thuman. 1992. *Development and Application of a Model of PCBs in the Green Bay, Lake Michigan Walleye and Brown Trout and Their Food Webs*. Report for Large Lakes Research Station, U.S. Environmental Protection Agency, Grosse Ile, MI 48138, Cooperative Agreement CR-815396.

- deBruyn, A.M.H. and F.A.P.C. Gobas. 2006. A bioenergetic biomagnification model for the Animal Kingdom. *Environ. Sci. Technol.* 40(5):1581-1587.

- Glaser, D. and J.P. Connolly, J. P. 2002. A model of p,p '-DDE and total PCB bioaccumulation in birds from the Southern California Bight. *Continental Shelf Res.* 22 1079.

- Gobas, F.A.P.C. 1993. A Model for Predicting the Bioaccumulation of Hydrophobic Organic Chemicals in Aquatic Food-Webs: Application to Lake Ontario. *Ecol. Modelling* 69:1-17.

- Imhoff, J. C., J.S. Clough, R.A. Park, and A. Stoddard. 2004. *Evaluation of Chemical Bioaccumulation Models of Aquatic Ecosystems: Final Report*. Prepared for U.S. EPA ORD National Exposure Research Laboratory, Ecosystems Research Division, Athens, GA. Available at : http://hspf.com/pdf/FinalReport218.pdf

- Kelly, BC and F.A.P.C. Gobas. 2003. An Arctic terrestrial food-chain bioaccumulation model for persistent organic pollutants. *Environ. Sci. Technol.* 37:2966-2974.

- QEA (Quantitative Environmental Analysis, LLC). 1999. *PCBs in the Upper Hudson River, Volume 2, A Model of PCB Fate, Transport, and Bioaccumulation. Section 5 – Bioaccumulation Model*. Prepared for: General Electric, Albany, NY. May 1999. Amended July 1999.

- QEA (Quantitative Environmental Analysis, LLC). 2001. *A Model of PCB Bioaccumulation in the Lower Fox River and Green Bay: GBFood*. Prepared for ThermoRetec, St. Paul, MN. June 2001.

- Thomann, R.V. 1989. Bioaccumulation model of organic chemical distribution in aquatic food chains. *Environ. Sci. Technol.* 23:699-707.

- Thomann, R.V., J.P. Connolly, and T.F. Parkerton. 1992. An equilibrium model of organic chemical accumulation in aquatic food webs with sediment interaction. *Environ. Toxicol. Chem.* 11:615-629.

- Weston Solutions Inc. 2004. *Model Calibration: Modeling Study of PCB Contamination in the Housatonic River. Appendix C - Bioaccumulation Model Calibration*. Prepared for U.S. Army Corps of Engineers (Concord, Massachusetts) and U.S. Environmental Protection Agency (Boston, Massachusetts). December 2004. Available at:

  http://www.epa.gov/ne/ge/thesite/restofriver/reports/model_calibration/217058_cal_vol4_appC.pdf

MODELLING TOOL #5

5.0 FUGACITY FATE AND TRANSPORT MODELS

What does this tool consist of? Fugacity-based fate models are used to predict chemical concentrations in abiotic media (water, air, soil, *etc.*) and in biota in a specified environment. Fugacity is directly proportional to chemical concentration, but is normalized to the sorptive capacity of a particular medium. Fugacity is effectively a measure of the tendency of a chemical to migrate between media.

Which ecosystem(s) would this tool typically be applied in? Fugacity models can be used in any type of real or hypothetical ecosystem at any scale. Many fugacity models have been developed for individual bodies of water or watersheds, but the approach has also been applied at regional, continental and global scales. The fugacity approach has also been used in detailed models of bioaccumulation and the distribution of chemicals within an organism.

How frequently is this tool used in a DERA? Rare.

What are the benefits of using this tool in a DERA? Fugacity models can be used to predict chemical concentrations in any abiotic medium or type of organism. When direct measurements of some concentrations are available, these may be used to validate the model. Well-defined methods exist to estimate the necessary parameters (*e.g.*, sorptive capacities). There are four levels of complexity in fugacity modeling, so it is possible to construct a very simple model (with few parameters) when this is appropriate, and to increase the level of complexity as necessary.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- Fugacity models are most appropriate for neutral organic substances because it is straightforward to estimate the sorptive capacities of environmental media and biota for these chemicals. Some fugacity models have been developed for charged organic substances, but the fugacity approach is difficult to apply to inorganics.

- As with any model, the output is only reliable if the model is well-constructed and the parameters are accurate. The simpler fugacity models make many simplifying assumptions, and may not accurately reflect reality. The more complex fugacity models require a large number of parameters to describe the ecosystem, so there is greater potential for compounding errors and uncertainty.

- Most existing fugacity models come with default parameter sets, but these are not appropriate for all ecosystems. It is essential to evaluate all parameter choices with respect to the particular system being assessed.

Where can I go for additional information about this tool?

- Mackay D. 2001. *Multimedia Environmental Models: The Fugacity Approach - Second Edition*. Lewis Publishers, Boca Raton, Florida.

- Woodfine D.G., M. MacLeod, D. Mackay and J.R. Brimacombe. 2001. Development of continental scale multimedia contaminant fate models: integrating GIS. *Environ. Sci. & Pollut. Res.* 8:164-172.

- Kelly, BC and F.A.P.C. Gobas. 2003. An arctic terrestrial food-chain bioaccumulation model for persistent organic pollutants. *Environ. Sci. Technol.* 37: 2966-2974.

A detailed introduction to fugacity-based multimedia fate models and a wide selection of downloadable models is available from the Canadian Environmental Modelling Centre at Trent University:

- http://www.trentu.ca/cemc/CEMC200102.pdf

- http://www.trentu.ca/cemc/new.html

MODELLING TOOL #6

6.0 PHYSIOLOGICALLY-BASED PHARMACOKINETIC MODELS

What does this tool consist of? Physiologically-based pharmacokinetic (PBPK) models mechanistically predict the uptake and distribution of substances within an individual organism. PBPK models represent various parts of the body as interconnected "compartments", usually specifying at least three such compartments (*e.g.*, blood, rapidly-perfused tissues and slowly-perfused tissues) and often specifying many more than three (*e.g.*, a compartment for each major organ). Transfer among compartments is usually considered to be via blood, and is therefore a function of tissue-blood partition coefficients, the volume of the tissue, and the flux of blood through the tissue. Mathematically, PBPK models use differential equations to describe the chemical concentration in each compartment as a function of the concentrations in other compartments.

Which ecosystem(s) would this tool typically be applied in? PBPK models are usually applied to mammals, and could be used in any ecosystem in which mammals are a receptor of concern.

How frequently is this tool used in a DERA? Rare.

What are the benefits of using this tool in a DERA?

- PBPK models can provide detailed information on exposure of receptors to COPCs via all routes simultaneously (ingestion, inhalation/gill exchange/transdermal absorption).

- PBPK models can predict total uptake rates, internal whole-body concentrations, or concentrations in specific target organs, and can therefore be used with dose-response relationships (ecological effects profiles) based on any of these measures of exposure.

- PBPK models can be used to extrapolate internal exposures among species by adjusting the physiological parameters to account for interspecies differences.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- PBPK models have been described as data-hungry, resource intensive, complex, time consuming, compound-specific and difficult to validate.

- PBPK models require detailed information on the physiology of the receptor and the physical-chemical properties of the chemical. Therefore, they are typically only constructed for very well-known species, such as humans and experimental mammals.

Where can I go for additional information about this tool?

- Cahill, T., Cousins, I., and Mackay D. 2003. Development and application of a generalized physiologically based pharmacokinetic model for multiple environmental contaminants. *Environ. Toxicol. Chem.* 22: 26-34.

- Clark, L.H., Setzer, R.W. and Barton, H.A. (2004) Framework for evaluation of physiologically-based pharmacokinetic models for use in safety or risk assessment. *Risk Anal.* 24:1697-1718.

- Wintermyer, M., A. Skaidas, A. Roy, Y.C. Yang, P. Georgapoulos, J. Burger, and K. Cooper. 2005. The development of a physiologically-based pharmacokinetic model using the distribution of 2,3,7,8-tetrachlorodibenzo-p-dioxin in the tissues of the eastern oyster (*Crassostrea virginica*). *Marine Environmental Research* 60(2):133-152.

MODELLING TOOL #7

7.0 ACID VOLATILE SULPHIDES AND SIMULTANEOUSLY EXTRACTABLE METALS

What does this tool consist of? The bulk concentrations of metals in sediments are poor predictors of their bioavailability to aquatic organisms. A comparison of the molar concentrations of acid volatile sulphide (AVS) and simultaneously extractable metals (SEM) has been found to be a useful predictive tool for assessing the potential for divalent metals (*e.g.*, cadmium, lead, zinc) to cause toxicity in sediments. If the ratio of SEM:AVS is less than 1.0 or SEM minus AVS is less than zero, then toxicity is not expected. If the ratio of SEM to AVS is greater than 1 or the difference is greater than 0, then benthic organisms may or may not be exposed to toxicity.

Which ecosystem(s) would this tool typically be applied in? SEM:AVS relationships can applied in aquatic systems but is generally most relevant for anaerobic sediments where sulphides can accumulate (*i.e.*, this tool is not very useful in highly oxidized environments).

How frequently is this tool used in a DERA? This tool is commonly used in a DERA of freshwater and marine sediments.

What are the benefits of using this tool in a DERA? SEM:AVS data provide information regarding the potential bioavailability of selected divalent metals and can therefore help assess the potential for effects if bulk sediment chemistry results exceed published sediment quality guidelines.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- The ratio of SEM to AVS is not reliable at low AVS concentrations.

- AVS may be lost from a sample prior to analysis if handled improperly (*e.g.*, the sample is not placed in a container immediately and without headspace), thereby resulting in an overestimation of potential for divalent metals to be bioavailable.

- A ratio of SEM to AVS of greater than one does not necessarily mean that the divalent metals present will cause toxicity as many additional factors control binding of metals to sediments (*e.g.*, particulate organic carbon and iron and manganese oxyhydroxides).

- Trivalent iron (Fe^{3+}) has been observed to oxidize acid-insoluble copper sulphide complexes and therefore increase SEM_{Cu} during the AVS-SEM extraction procedure, without a corresponding increase in AVS. Therefore, an artifact of the analysis may be an overestimation of the potential for copper to become bioavailable (*i.e.*, an artificially elevated SEM:AVS).

Where can I go for additional information about this tool? Examples of one or more aspects of the issues discussed above can be found in the following peer-reviewed scientific literature:

- Allen, H.A., G. Fu, and B. Deng. 1993. Analysis of acid-volatile sulfide (AVS) and simultaneously extracted metals (SEM) for the estimation of potential toxicity in aquatic sediments. *Environ. Toxicol. Chem.* 12:1441-1453.

- Carlson, A.R., G.L. Phipps and V.R. Mattson. 1991. The role of acid-volatile sulfide in determining cadmium bioavailability and toxicity in freshwater sediments. *Environ. Toxicol. Chem.* 10:1309-1319.

- Chapman, P.M., F. Wang, C. Janssen, G. Persoone, and H.E. Allen. 1998. Ecotoxicology of metals in aquatic sediments: binding and release, bioavailability, risk assessment, and remediation. *Can. J. Fish. Aquat. Sci.* 55: 2221-2243.

- DiToro, D.M., J.D. Mahony, D.J. Hansen, K.J. Scott, A.R. Carlson, and G.T. Ankley. 1992. Acid volatile sulfide predicts the acute toxicity of cadmium and nickel in sediments. *Environ. Sci. Technol.* 26:96-101.

- DiToro, D.M., J.D. Mahony, D.J. Hansen, K.J. Scott, M.B. Hicks, S.M. Mayr, and M.S. Redmond. 1990. Toxicity of cadmium in sediments: the role of acid volatile sulfide. *Environ. Toxicol. Chem.* 9:1487-1502.

- Simpson, S.L., S.C. Apte, and G.E. Batley. 1998. Effect of short-term resuspension events on trace metal speciation in polluted anoxic sediments. *Environ. Sci. Technol.* 32:620-625.

MODELLING TOOL #8

8.0 ORGANIC CARBON AND LIPID NORMALIZATION

What does this tool consist of? Biota-sediment or biota-soil accumulation factors (BSAFs) for hydrophobic chemicals are most easily predicted and interpreted when the chemical concentrations in sediment/soil and biota are normalized to the organic carbon (OC) content of the sediment/soil and the lipid content of the organism:

$$C_{S,OC} = C_S / \phi_{OC} \text{ and } C_{Biota,L} = C_{Biota} / \phi_L$$

where C_S and C_{Biota} are the chemical concentrations in sediment/soil and biota (any units, as long as they are consistent), and Φ_S and Φ_L are the OC and lipid fractions (unitless) in sediment/soil and biota, respectively. When concentrations are normalized in this way, the BSAF is theoretically (assuming equilibrium partitioning) equal to the relative sorptive capacities of lipid and OC (usually estimated to be ~ 1.7), multiplied by the ratio of biota lipid to sediment/soil OC fractions (Φ_L / Φ_S).

Which ecosystem(s) would this tool typically be applied in? OC and lipid normalization can be applied in any system in which contaminant concentrations in organisms might be predicted from concentrations in soil or sediment

How frequently is this tool used in a DERA? Common.

What are the benefits of using this tool in a DERA? OC and lipid normalization provides a simple method to estimate the exposure of soil- or sediment-associated receptors from measured or estimated concentrations of COPCs in soil or sediment.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- Predicting BSAFs from OC- and lipid-normalized concentrations assumes that the organism and the soil or sediment are at or near chemical equilibrium, and that all of the chemical in sediment/soil is bioavailable. Empirical studies suggest that this is often not true. The true BSAF may be higher than predicted if the chemical is biomagnified, or lower than predicted if the chemical is rapidly metabolized or if the chemical in sediment/soil has low bioavailability.

- Predicting BSAFs from OC- and lipid-normalized concentrations is only appropriate for neutral (nonionic) organic chemicals.

Where can I go for additional information about this tool?

- Di Toro, D.M. and L.D. De Rosa. *Equilibrium Partitioning and Organic Carbon Normalization*. National Sediment Bioaccumulation Conference. Available online at http://www.epa.gov/waterscience/cs/ditoro.pdf.

- Di Toro, D.M., C. Zarba, D.J. Hansen, W. Berry, R.C. Swartz, C.E. Cowan, S.P. Pavlou, and H.E. Allen. 1991. Technical basis for establishing sediment quality criteria for nonionic organic chemicals using equilibrium partitioning. *Environ. Toxicol. Chem.* 10:1541-1583.

- Wong, C.S., P.D. Capel, and L.H. Nowell. 2001. National-scale, field-based evaluation of the biota-sediment accumulation factor model. *Environ. Sci. Technol.* 35:1709-1715.

- Parkerton, T. F., J. P. Connolly, R. V. Thomann, and C. G. Uchrin. 1993. Do aquatic effects or human health end points govern the development of sediment-quality criteria for nonionic organic chemicals? *Environ. Toxicol. Chem.* 12:507-523.

MODELLING TOOL #9

9.0 BIOAVAILABILITY ASSESSMENT MODELS

What does this tool consist of? The degree to which dietary contaminants are available for uptake by a consumer (bioavailability or bioaccessibility) can be estimated *in vitro* by measuring the fraction of dietary contaminant that is solubilized under conditions that mimic the consumer's gut. Approaches range from simply mimicking the pH of a consumer's gut, to including a full enzyme complement, to using real digestive fluid extracted from wild or cultured animals. The more elaborate approaches are sometimes called physiologically based extraction tests (PBETs). Bioavailability assessment models are mainly applied to soil- and sediment-feeding organisms, because bioavailability in soil and sediment is known to be highly variable among ecosystems.

Which ecosystem(s) would this tool typically be applied in? Any ecosystem in which soil- or sediment-feeding organisms are ROPCs.

How frequently is this tool used in a DERA? Rare in ecological risk assessment (more common in human health risk assessment), but increasing in application.

What are the benefits of using this tool in a DERA? These methods are a quick and inexpensive way to improve ecological relevance in assessment of dietary exposure to soil- or sediment-associated COPCs.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA? The digestive fluid extraction approach is probably not useful for compounds for which ingestion is likely to be a minor route of uptake (*e.g.*, hydrophilic organic compounds) or those for which intestinal absorption rather than solubilization constrains uptake (*e.g.*, chromium).

Where can I go for additional information about this tool?

- Oomen, A.G., A. Hack, M. Minekus, E. Zeijdner, C. Cornelis, G. Schoeters, W. Verstraete, T. Van de Wiele, J. Wragg, C.J. Rompelberg, A.J. Sips, J.H. Van Wijnen. 2002. Comparison of five in vitro digestion models to study the bioaccessibility of soil contaminants. *Environ. Sci. Technol.* 36:3326-3334.

- Weston, D.P., R.N. Millward, L.M. Mayer, I. Voparil, and G.R. Lotufo. 2002. *Sediment extraction using deposit-feeder gut fluids: A potential rapid tool for assessing bioaccumulation potential of sediment-associated contaminants.* ERDC/EL T R-02-18, U.S. Army Engineer Research and Development Center, Vicksburg, MS. Available online at: http://el.erdc.usace.army.mil/elpubs/pdf /trel02-18.pdf

MODELLING TOOL #10

10.0 METAL SPECIATION MODELS

What does this tool consist of? Water chemistry parameters are used to calculate the freely dissolved ion fraction of a metal in aqueous solution. Metals in aqueous solutions form numerous chemical species in solution of which only a proportion are freely dissolved (and are thereby more bioavailable).

Which ecosystem(s) would this tool typically be applied in? Deep Aquatic, Shoreline, and Rivers & Streams.

How frequently is this tool used in a DERA? Rare, but increasing in application.

What are the benefits of using this tool in a DERA?

- Metal speciation models provide an estimate of the bioavailable fraction of metals in an aqueous solution. Generally, only metals in the ionized form are considered to be bioavailable. Calculating the ionized form is superior to using total dissolved metal concentrations (which has often been used as a surrogate for the bioavailable portion) because the dissolved fraction contains a combination of metal ions, soluble complexes and small particles of insoluble precipitates.

- The estimate of the bioavailable fraction can help the risk assessor bound the exposure range of metals to aquatic receptors for a given site. In many cases, the model is used to show that the actual exposure is much less than what the measured dissolved concentration of metal in solution would indicate.

- Model results are useful for providing context for site-specific bioavailability and toxicity of contaminants relative to literature-based toxicity studies. The latter often report results based on soluble metal salts, which maximize uptake.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- Metal speciation models provide a measure of bioavailability, but do not provide any information on the interaction of the dissolved metal species and the site of action (or biotic ligand) on the receptor. Consequently, interpretation of a "low bioavailability" has to be exercised with care. Even if the bioavailable portion of a metal is low, the proportion of bioavailable metal interacting with the biotic ligand may be high. For this reason, biotic ligand models should be used where available.

- There are several metal speciation models available, and most require some knowledge of chemical thermodynamics. These models are based on dissociation constants for each of the potential metal to ligand complexes. There are several sources of these dissociation constants; scientific advancements result in the periodic modification of the dissociation constants.

- The models calculate metal speciation using different mathematic algorithms. One area where models diverge is in the description of interactions between metals and dissolved organic carbon. The interaction of metals with organic matter in water is highly complex and some models provide a more realistic description of this interaction than others. If organic binding is likely to account for a large proportion of the metal-ligand binding, then it is advisable to use a model that uses a more sophisticated approach to modeling this interaction. The Windermere Humic Aqueous Model (WHAM) is one example of a metal speciation model which provides a more sophisticated approach to modeling the metal to organic ligand binding.

- A detailed understanding of how water quality guidelines were derived for the metal in question. Specifically, it is necessary to consider how differences in water quality parameters (*e.g.*, pH, hardness, organic carbon, major ions) in toxicity tests used to derive the criteria vary from the conditions in the field. Metal speciation models are useful for instances in which the bioavailability of metals in the toxicity test upon which the criteria were based is high but the estimated bioavailability of a metal in the site water is low.

Where can I go for additional information about this tool?

- Schecher, W.D. & D.C. McAvoy. 2003. *MINEQL+: A Chemical Equilibrium Modeling System, Version 4.5 for Windows, User's Manual*. Environmental Research Software, Hollowell, Maine.

- Tipping, E., 2005. *Windermere Humic Aqueous Model (WHAM) - A Chemical Equilibrium Model And Computer Code For Waters, Sediments And Soils Incorporating A Discrete Site / Electrostatic Model of Ion-binding By Humic Substances*. Centre for Ecology and Hydrology. Available at: http://www.ife.ac.uk/aquatic_processes/wham/

- Tipping, E. 1994. WHAM - A chemical equilibrium model and computer code for waters, sediments, and soils incorporating a discrete site/electrostatic model of ion-binding by humic substances. *Computers Geosciences* 20:973-1023.

- USEPA. 2003. *2003 Draft Update for Ambient Water Quality Criteria for Copper*. U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, Washington, DC, USA.

MODELLING TOOL #11

11.0 BIOTIC LIGAND MODELS

What does this tool consist of? Biotic ligand models (BLMs) utilize ancillary water quality parameters (*e.g.*, pH, hardness, dissolved organic carbon, major ions) and the measured dissolved concentration of the metal of interest to derive a site-specific water quality criterion. BLMs predict the concentration of a metal bound to biotic ligands, which are located on the respiratory surfaces of aquatic organisms considered to be the cellular "site of action". The concentration of metal bound to the biotic ligand is directly related to the metal-mediated acute effect.. The estimated concentration of a metal bound to the biotic ligand for a given site is compared to toxicity reference values obtained from laboratory-based toxicity testing.

BLMs incorporate thermodynamically based metal speciation models in order to estimate the bioavailability of dissolved metals in water. Unlike metal speciation models, BLMs take one additional step by also estimating the competition for binding which occurs between the metal of interest and natural ions for the biotic ligand.

Which ecosystem(s) would this tool typically be applied in? Deep Aquatic, Shoreline, and Rivers and Streams.

How frequently is this tool used in a DERA? Rare, but becoming more common. The USEPA has recently provided a draft manual for deriving site specific water quality criteria for copper based on a biotic ligand (BLM) approach[1].

What are the benefits of using this tool in a DERA? BLMs provide improved estimates of dissolved metal concentrations unlikely to result in a deleterious effect. They are useful for reducing the uncertainty associated with using total concentrations for evaluating metal toxicity in freshwater ecosystems.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- BLMs are relatively new and therefore the availability of calibrated, validated models is limited. USEPA uses a BLM to derive site specific water quality criteria for copper, and anticipates developing BLMs for other metals in the future.

[1] http://www.epa.gov/waterscience/criteria/copper/2007/blm-tsd.pdf

- The existing USEPA BLM model is based on a complex metal speciation model, but for ease of use, the number of required water quality parameters was reduced. There is no ability for the user to modify thermodynamic dissociation constants, meaning that it is necessary to apply the USEPA default values for a substantial number of parameters rather than incorporate site-specific values.

- BLMs are largely based on the results of acute toxicity tests using a small number of freshwater aquatic organisms; they incorporate an acute-to-chronic ratio to extrapolate the model to chronic conditions. Research in the development of truly chronic BLMs as well as BLMs for marine organisms is ongoing.

Where can I go for additional information about this tool?

- Di Toro, D.M., H.E. Allen, H.L. Bergman, J.S. Meyer, P.R. Paquin and R.C. Santore. 2001. A biotic ligand model of the acute toxicity of metals I. Technical basis. *Environ. Toxicol. Chem.* 20:2383-2396.

- Niyogi, S. and C. M. Wood. 2003. Effects of chronic waterborne and dietary metal exposures on gill metal-binding: Implications for the biotic ligand model. *Human and Ecological Risk Assessment* 9:813-846.

- Paquin P.R., J.W. Gorsuch, S. Apte, G.E. Batley, K.C. Bowles, P.G.C. Campbell, C.G. Delos, D.M. Di Toro, R.L. Dwyer, F. Galvez, R.W. Gensemer, G.G. Goss, C. Hogstrand, C.R. Janssen, J.C. McGeer, R.B. Naddy, R.C. Playle, R.C. Santore, U. Schneider, W.A. Stubblefield, C.M. Wood and K.B. Wu. 2002. The biotic ligand model: a historical overview. *Comp Biochem Physiol, Part C* 133:3-35

- Santore, R.C., D.M. DiToro, P. R. Paquin, H.E. Allen and J.S. Meyer. 2001. Biotic ligand model of the acute toxicity of metals. 2. Application to acute copper toxicity in freshwater fish and *Daphnia*. *Environ. Toxicol. Chem.* 20:2396-2402.

- USEPA. 2003a. *2003 Draft Update for Ambient Water Quality Criteria for Copper*. U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, Washington, DC, USA

- USEPA. 2003b. The Biotic Ligand Model: Technical Support Document for Its Application to the Evaluation of Water Quality Criteria for Copper. U.S. Environmental Protection Agency, Office of Science and Technology, Health and Ecological Criteria Division, Washington, D.C. November 2003. Available at: http://www.epa.gov/waterscience/criteria/copper/2007/blm-tsd.pdf

MODELLING TOOL #12

12.0 QUANTITATIVE STRUCTURE ACTIVITY RELATIONSHIPS (QSARS)

What does this tool consist of? QSAR models are mathematical equations that describe a relationship between the toxicity (or other properties) of chemicals and their measured physico-chemical properties or structures. A QSAR derived for some members of a family of chemicals can then be used to predict unmeasured values for other members of the same family. QSARs are often used to estimate bioaccumulation or toxicity of new industrial chemicals/pesticides for which bioaccumulation or toxicity testing has not been conducted.

Which ecosystem(s) would this tool typically be applied in? All ecosystems.

How frequently is this tool used in a DERA? Rare, except when dealing with new or unusual chemicals.

What are the benefits of using this tool in a DERA? This tool is useful for new compounds about which little is known. QSARs provide a means to screen these chemicals, so that testing can be focused on chemicals that are most likely to be of ecological concern.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- Most QSARs are statistical models (*i.e.*, regression equations), not mechanistic models. They describe a statistical correspondence between structure and activity in the set of chemicals used to develop the model. In applying a QSAR to estimate properties of new chemicals, we are assuming that the correspondence will continue to hold. This may not always be true. If it is not true (*i.e.*, if the new chemical is different in some unknown way), then the predictions may be completely inaccurate.

- Regression-based QSARs also describe the strength of the statistical relationship (r^2). This information should be used to put confidence limits on the estimated value, although this is rarely documented.

Where can I go for additional information about this tool?

- Gobas, F.A.P.C., BC Kelly, and J.A. Arnot. 2003. Quantitative Structure Activity Relationships for Predicting the Bioaccumulation of POPs in Terrestrial Food-Webs. *QSAR & Combinatorial Science* 22:329–336.

- Arnot, J.A. and F.A.P.C. Gobas. 2003. A Generic QSAR for Assessing the Bioaccumulation Potential of Organic Chemicals in Aquatic Food Webs. *QSAR & Combinatorial Science* 22:337–345.

- Cronin, M.T.D. and D.J. Livingstone, Eds. 2004. *Predicting Chemical Toxicity and Fate*. CRC Press.

A relevant USEPA website on ECOSAR (Ecological Structure Activity Relationships) is also available at: http://www.epa.gov/oppt/newchems/tools/21ecosar.htm

MODELLING TOOL #13

13.0 TROPHIC TRANSFER FOOD CHAIN MODELS

What does this tool consist of? Conceptually, trophic transfer models calculate the daily ingested dose of a COPC to a wildlife receptor based on site-specific data (soil, tissue, water chemistry data), information about dietary preferences, and equations that predict food, soil and water consumption rates based on the receptor's body mass. These daily ingested dose estimates can be modified based on considerations such as habitat use, foraging range or COPC bioavailability. The calculated daily ingested dose is compared to a toxicity reference value (TRV, *e.g.,* point estimate or dose-response relationship) to characterize risks.

Trophic transfer models share the above commonalities, but in application the models can be modified extensively depending on the needs of the risk assessment. Potential customizations include:

- The use of site-specific tissue chemistry data from the site to replace the use of BAF/BCF models in the simulation of tissue concentrations of the base of the food web;

- Increasing the number of receptors evaluated in the model. Some models may focus on generic receptors to represent broad ROPC categories, whereas others may include large numbers of individual species to reflect site-specific habitat use patterns;

- Incorporating receptor-specific site use patterns based on habitat features throughout the site to estimate ROC exposure doses - Wildlife will use a site differently depending on their habitat and foraging requirements, which can affect their overall exposure at the site. Some models will base exposure on statistical point estimate concentrations (*e.g.*, 90^{th} percentile soil concentration); when greater site-specificity is required to represent complexity in ROC foraging, habitat-weighted and spatially-weighted exposure estimates can be used.,

- Increasing the number of dietary items in the model. Some models limit the categories to generic dietary categories (*e.g.*, "plants" and "soil invertebrates") while others include multiple specific dietary items.

- Calculating organism food ingestion rates based on caloric (metabolic) requirements rather than on the basis of organism mass.

- Introduction of probabilistic elements versus reliance on point-estimate values.

Which ecosystem(s) would this tool typically be applied in? Trophic transfer models are often applied in terrestrial ecosystems, although the procedure may be extended to coastal and marine birds and mammals.

How frequently is this tool used in a DERA? Frequent. The level of refinement and sophistication varies with the level of detail and complexity in the ERA.

What are the benefits of using this tool in a DERA? Trophic transfer food chain models are one of the few tools available for evaluating potential risks to wildlife associated with site contamination. Toxicity data from the literature are often expressed in units of milligrams contaminant per kilogram body weight per day, which is easily compared to output from the food chain models.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- The ecological realism of hazard quotients in evaluating trophic transfer model outputs is strongly influenced by the degree to which site-specific information can be incorporated into the model. A model that relies on default receptor parameters, simplistic assumptions about dietary items and literature based bioaccumulation factors has value for screening assessment, but may not be suitable to meet the objectives of the DERA.

- Hazard quotients resulting from a food chain model do not provide information about the relative risks of different contaminants (*i.e.*, a HQ of 10 is not five times worse than a HQ of 2). An HQ above 1 indicates that there is potential for an effect; further information on the magnitude and probability of an effect is required to assess risks.

- There is considerable uncertainty in the derivation of toxicity reference values (TRVs) for food chain models which can be masked by the availability of "look-up" tables of TRVs. It is important to understand the underlying dose-response relationship behind a TRV, even if a single-point estimate is used.

- Most trophic transfer models (and the associated HQ method) evaluate risks to individual organisms but cannot be used to directly assess effects to populations and/or communities. Risk assessors need to be clear about how they are defining populations (*e.g.,* site *vs.* local *vs*. regional) and ensure that interpretations are consistent with the narrative goals expressed in the assessment endpoints.

Where can I go for additional information about this tool?

- McDonald, B.G. and J.B. Wilcockson. 2003. Improving the use of toxicity reference values in wildlife food chain modeling and ecological risk assessment. *Human Ecol. Risk Assess.* 9:1585-1594.

- Nagy, K.A., I.A. Girard, T.K. Brown. 1999. Energetics of free-ranging mammals, reptiles and birds. *Annu. Rev. Nutr.* 19:247-277.

- Pascoe G., R. Blanchett, and G. Linder. 1996. Food chain analysis of exposures and risks to wildlife at a metals-contaminated wetland. *Arch. Environ. Contam. Toxicol.* 30:306-318.

- Sample, B., D. Opresko, G. Suter II. 1996. *Toxicological benchmarks for wildlife: 1996 revision*. Oak Ridge National Laboratory, Department of Energy. Oak Ridge, TN. ES/ER/TM-86/R3.

- USEPA. 1993. *Wildlife Exposure Factors Handbook*. Office of Research and Development, United States Environmental Protection Agency, Washington DC. December 1993. EPA/600/R-93/187

- USEPA. 2005. *Guidance for developing ecological soil screening levels.* Office of Solid Waste and Emergency Response. OSWER Directive 9285.7-55. Available at: http://www.epa.gov/ecotox/ecossl/.

APPENDIX III

INTERPRETIVE TOOLS

TABLE OF CONTENTS

INTERPRETIVE TOOL #1

1.0 HAZARD QUOTIENTS

What does this tool consist of? A hazard quotient (HQ; also known as the Risk Quotient) is the ratio of a receptor's observed or simulated exposure (Section 4) to a toxicity reference value (TRV; see also Section 5).

The hazard quotient for each combination of contaminant and receptor (plant or animal) of concern is calculated by dividing the estimated environmental concentration (EEC) by the toxicity reference value (TRV):

$$HQ = \frac{EEC}{TRV}$$

Over the years, in ecological risk assessments there has been implicit agreement that a hazard quotient ≤1.0 is the default *de minimis* range. HQs are often interpreted based on a binary decisions (*i.e.*, potential risks [hazards] are present if HQ > 1; risks are considered negligible if HQ ≤1). However, the uncertainties implicit in both exposure estimates and TRVs can vary widely, such that assuming that an HQ > 1 (*i.e.*, "bright line" interpretation) represents unacceptable risks is inappropriate. The HQ has value as a screening tool, which may be all that is required in some risk assessments. In other cases, when HQ > 1, further evaluation (*e.g.,* use of additional endpoints, weight-of-evidence evaluation, *etc.*) might be required to evaluate risks (or risk management can be put in place). Also, although order-of-magnitude differences in HQ values can be used to make qualitative inferences regarding potential for harm, an assumption of linearly increasing risk with increasing HQ cannot be made. Moving beyond the HQ is appropriate for more advanced ERAs where uncertainties are being iteratively addressed and other lines of evidence are used to augment screening.

Which ecosystem(s) would this tool typically be applied in? All ecosytems.

How frequently is this tool used in a DERA? Very common.

What are the benefits of using this tool in a DERA?

- Hazard quotients require few data, are easy to calculate and easy to interpret.
- Hazard quotients are useful in screening evaluations to determine whether follow-up investigations are warranted (see Section 6.6.2).

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- Hazard quotients are only as reliable as the exposure measures and TRVs used to calculate them. The calculations, extrapolations, and approximations used to estimate exposure and the TRV[1] from available data are a potentially large source of error and must be assessed carefully in the uncertainty analysis.

- Hazard quotients communicate very little information regarding the magnitude of possible effects. A larger HQ is presumably associated with more severe effects, but this is not quantifiable. As such, HQs are most useful in screening-level risk assessments to indicate when a more detailed assessment is warranted.

- HQs are applicable to individual contaminants, and combination of values through summation (hazard index approach) is not well-supported by existing toxicological data.

- HQs assume that a relatively definitive threshold for both exposure and toxic response can be derived. For example for toxic responses, the uncertainty and variability in toxicological data constraints the ability to identify a single "bright line" TRV. In addition, the experimental designs and endpoints of toxicity tests are variable, such that clear identification of the driver for a TRV is seldom straightforward.

Where can I go for additional information about this tool?

- Efroymson, R.A., M.E. Will, and G.W. Suter II. 1997. *Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Processes: 1997 Revision*. ES/ER/TM-126/R2, Oak Ridge National Laboratory, Environmental Sciences Division.

[1] Derivation of TRVs often requires calculations, approximations, and application (safety or uncertainty) factors, all of which present sources of uncertainty and potential error. For example, TRVs are commonly expressed in units of intake rates (daily dietary dose, mg/kg-day or similar). When available toxicity data are in terms of dietary concentrations (mg/kg-food), they must be converted to intake rates with an estimated or measured feeding rate. Application factors include estimation of the desired effects concentrations (*e.g.*, NOAELs) from other reported values (*e.g.*, LOAELs), sub-chronic to chronic extrapolations, interspecies extrapolations. Other conversions include consideration of allometric scaling and unit conversions. Producing an appropriate compilation of data from which to derive a reliable TRV can require a significant investment of resources. Although "off the shelf" TRVs are readily available, in DERAs it is often necessary to research the range of TRVs from various experimental organisms. It is anticipated that MOE will be developing guidance on development of TRVs in the future.

- McDonald, B.G. and J.B. Wilcockson. 2003. Improving the use of toxicity reference values in wildlife food chain modeling and ecological risk assessment. *Human and Ecological Risk Assessment* 9:1–10.

- Suter, G.W. II. 2006. *Ecological Risk Assessment*. Second Edition. CRC Press, Boca Raton, FL. 643 p.

- Tannenbaum, L.V., M.S. Johnson, and M. Bazar. 2003. Application of the hazard quotient method in remedial decisions: A comparison of human and ecological risk assessments. *Human and Ecological Risk Assessment* 9(1):387 - 401.

INTERPRETIVE TOOL #2

2.0 EC_X ASSESSMENT

What does this tool consist of? The EC_X approach is a method of evaluating the significance of toxicity test results, using an effect-size based approach. In some cases, the EC_X approach is extended beyond toxicity test endpoints to other biological responses. The approach is based on a policy decision that a defined level of effect is acceptable. Acceptable effect levels may depend on ecosystem and land use and are the subject of policy (see most recent MOE *Policy Decision Summary*). The user of this interpretive tool is advised to consult with the appropriate policy and/or regulatory agencies regarding applicable effect levels and acceptable endpoints.

The development of the EC_X approach comes from standard ecotoxicity testing protocols (Rand, 1995). Data from fixed times of observation are transformed so that least-squares optimization techniques can be used using linear models. Linearity is usually achieved by using the logged exposure concentration and transforming the response to a probit (probability unit) or logit (logistic unit). Although statistical details vary, all methods use some type of numerical interpolation to estimate the test concentration associated with a defined level of response.

Historically, many TRVs have been based on no-observed-adverse-effect levels (NOAELs) or lowest-observed-adverse-effect levels (LOAELs). This practice was supported by availability of LOAEL and NOAEL-based TRVs in easy-to-access compendia (*e.g.*, Oak Ridge National Laboratory, ECO-Soil Screening Levels); however, this practice has been replaced in recent years by the use of EC_X. For example, Efroymson and Suter (1999) and Pack (1993) have suggested that reductions in survival, growth, or reproduction of 20% or greater are indicative of significant effects to wildlife. Accordingly, the EC_X approach has been adopted as common risk assessment practice in several North American jurisdictions.

Which ecosystem(s) would this tool typically be applied in? This tool applies to: **All Ecosystems**

How frequently is this tool used in a DERA? This tool is frequently used in DERA, although the percent effect size threshold varies by land use, as a matter of provincial policy.

What are the benefits of using this tool in a DERA?

The EC_X provides a tool for evaluating toxicity data and determining the significance within an ecological context. Defining an EC_X as acceptable effect size for ecological risk assessments is not based on strict scientific principles, but rather reflects several practical considerations, including:

- *Limited utility of statistical significance measures* – Chapman *et al.* 1996 indicated that point estimates such as EC_X are more consistent, more reliable, and less variable estimates than NOECs. The NOEC, more than other parameters, is sensitive to the selection of the level of significance, the statistical procedures used, sample size, and other considerations.

- *Consistency* – Although somewhat arbitrary when applied across different organisms/populations, application of predefined EC_X (see MOE *Policy Decision Summary*) provides a degree of standardization to the interpretation of toxicity endpoint data.

- *Natural variations* – Most toxicity tests contain elements of natural variability that cannot be reduced even in a controlled laboratory environment. Because some low-level responses commonly occur in toxicity tests, it has been suggested that an "acceptable" response size can occur without necessarily being indicative of a contaminant-mediated effect. Accordingly, the EC_X method is thought to protect against false positives in some experimental designs.

- *Limitations of toxicity protocols* – The standard procedures for some toxicity tests do not incorporate sufficient statistical power to enable detection of very low effect sizes, particularly when the number of organisms per replicate is 10 or less. For some tests, the minimum significance difference (MSD) is close to 20% on average, although for some tests the MSD routinely exceeds 20%.

- *Negative control acceptability* – For many water and sediment toxicity tests, the acceptability standard is ≥80% of negative control performance, such that 20% reduction is operationally viewed to be within the range of experimental error. In other cases an alternate effect size is used (*e.g.*, 30% reduction in combined normal survival in bivalve larval development).

In summary, using EC_X values can provide a suitable comprise between sensitivity, confidence and reliability. However, because effects at or above a prescribed level may not always be concordant with ecological significance, these categorizations should be considered further in each situation.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- The EC_X effect size threshold varies according to existing risk assessment policy, which is determined largely by land use. Risk assessors must consider this provincial policy in light of regulatory requirement for other overlapping jurisdictions.

- The EC_X approach is a convention for operationally evaluating the significance of a toxic response, and should not be confused with statistical significance. Suter *et al.* (1995) recommend documenting both types of significance criteria. Any significant effects should be identified as either operationally significant (*e.g.*, > 20% effect) or statistically significant (*e.g.*, p value below critical α level of significance).

- The EC_X approach reflects a policy decision regarding a permissible level of effects, and may not always be synonymous with biological or ecological relevance.

- Some endpoints are ill-suited to EC_X analysis because they cannot be easy standardized to a "100% normal condition" (*e.g.*, presence of severe skeletal abnormalities in fish; amphipod emergence in number/jar/day; bird eggshell thickness). In these cases, alternate interpretive tools are required.

- Because the EC_X approach originated in the field of standardized toxicity testing, its application to other lines of evidence (*e.g.*, benthic community metrics) is unclear, particularly in terms of extrapolation of population or community measures as opposed to individual-level effects.

- There are cases where subtle differences in application of the EC_X approach can result in divergent conclusions. Consider the following table:

	Start Mass	**End Mass**	**% Effect (Mass)**	**Growth in Mass**	**% Effect (Growth in Mass)**
Control	1.0	1.2	-	0.2	-
Exposed	1.0	1.1	8%	0.1	50%

  In the table, the effect size could be interpreted as either 8% or 50%, depending on whether the endpoint is absolute mass or growth in mass. As with all scientific studies it is essential that a hypothesis is identified prior to conducting an experiment. This means that the measurement endpoint (absolute mass or growth in mass) must be selected prior to conducting the toxicity tests. Toxicity test results should then be evaluated in reference to the hypothesis (*i.e.*, the selected endpoints). The practitioner should select endpoints prior to testing and analyze data in context of the testing hypothesis, not in context of the results obtained.

Where can I go for additional information about this tool?

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INTERPRETIVE TOOL #3

3.0 SPECIES SENSITIVITY DISTRIBUTIONS

What does this tool consist of? A species sensitivity distribution (SSD) is the probability distribution of some measure of toxicity of a certain chemical to a set of animal species. Single-species toxicity data (*e.g.*, LC_{50} values, EC_X values, or NOECs) for many species are fit to a distribution such as the lognormal or log-logistic. From this distribution of species sensitivities, a hazardous concentration (HC_p) is identified at which a certain percentage (p) of all species is assumed to be affected. Selection by risk assessors of both the percentage of species and the effect level are matters of policy and require MOE input.

Which ecosystem(s) would this tool typically be applied in? An SSD can be derived for any ecosystem type for which sufficient toxicity data are available.

How frequently is this tool used in a DERA? Occasional, but increasing over time.

What are the benefits of using this tool in a DERA? SSDs provide a way to combine toxicity data for many species, reducing the effect of uncertainty in individual toxicity measurements. Calculating an HC_p allows for a simultaneous assessment of toxic effects in all potential receptors. As an added benefit, calculating an HC_p forces an explicit recognition of the magnitude of effect being considered (the chosen endpoint of the single-species toxicity tests) and the percentage of affected species that is judged to be acceptable (p).

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- The HC_p incorporates the limitations of the single-species toxicity data used to generate the SSD. If LC_{50}s are used, the HC_p will estimate the concentration at which 50% lethality occurs in p% of species, which may not be an adequately protective level. If the LC_{50}s are highly uncertain or have low ecological relevance (*e.g.*, due to unrealistic test conditions), the HC_p will be similarly limited. It is therefore important to establish agreement regarding the effect level with MOE.

- Calculating an HC_p explicitly recognizes that some fraction of species will be affected at any given concentration, but does not consider which species these are. If the species that fall into the affected p percentage are considered values ecosystem components, the resulting ecological effects may be greater than predicted. The practitioner must ensure that the species that fall into the affected p percentage is not protected by other provincial or federal legislation (*e.g.,* species at risk).

- This approach is based on the assumption that the toxicity data are from a random sample of species. In practice, the species for which data are available may not be representative of the real set of species in the ecosystem of interest. Whenever possible, the toxicity data used to generate the SSD should come from species representative of the system under consideration, or related indicator species. If the COPC has a particular target taxon, the SSD must include representatives of that taxon. Alternatively, it may be advisable to construct a SSD for the target taxon and another for non-target taxa.

- The SSD requires explicit articulation of the magnitude of effect and the percentage of affected species that is judged to be acceptable (p); the analysis is highly sensitive to these protection goals Selection by risk assessors of both the percentage of species and the effect level are matters of policy and require MOE input.

Where can I go for additional information about this tool?

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INTERPRETIVE TOOL #4

4.0 SUMMARY METRICS

What does this tool consist of? Summary metrics are numerical expressions of the characteristics of a biological community and are based on taxonomic data (*i.e.*, species identification and abundance). They can include attributes such as: abundance (number of organisms); richness/diversity (number of taxa); presence/absence of sensitive taxa; ratios of indicator taxa (*e.g.*, percent Ephemeroptera-Plecoptera-Trichoptera [EPT] in benthic invertebrate communities); and ratios of functional feeding groups. Numerous indices have also been developed as a means of describing biological communities (*e.g.*, Swartz Dominance Index; Bray-Curtis Index; Index of Biotic Integrity).

Which ecosystem(s) would this tool typically be applied in? Any ecosystem in which a biological community survey has been conducted. The tool is used most commonly for macroinvertebrate assessments based on field-collected samples and associated taxonomic enumerations.

How frequently is this tool used in a DERA? Commonly applied in DERAs in which biological community surveys have been conducted.

What are the benefits of using this tool in a DERA? Summary metrics simplify complex taxonomy data so that the patterns and relationships that describe the structure of a biological community can be assessed. They can be used as measurement endpoints for assessment endpoints involving biological community structure and can be incorporated into statistical analyses of differences between exposure and reference sites and correlations with habitat variables (*e.g.*, water depth, grainsize distribution) and measures of exposure to provide information about effects potentially related to contaminants of concern.

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- Because summary metrics by definition simplify complex data sets, a variety of metrics need to be used to assess the structure of a biological community to determine presence/magnitude of effects. Different metrics focus on different aspects of a community, for example, species richness versus evenness of distribution of individuals among the species identified.

- Summary metrics such as richness and diversity indices do not necessarily provide an evaluation of the ecological function of the organisms in a biological community. Diversity and other community level indices "can give very misleading biological interpretations of the data they are intending to summarize" (Boyle *et al.*, 1990; see also Washington, 1984; Izsáj and Papp, 2000; Thiebaut *et al.,* 2002).

- Biological communities are highly variable, so biotic indices typically have low statistical power to detect ecological effects. Indices are best used as one component of a weight-of-evidence approach.

Where can I go for additional information about this tool?

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- Bryce, S.A., R.M. Hughes, and P.R. Kaufmann. 2002. Development of a bird integrity index: using bird assemblages as indicators of riparian condition. *Environ. Manage.* 30: 294-310.

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INTERPRETIVE TOOL #5

5.0 MULTIVARIATE STATISTICAL ANALYSES

What does this tool consist of? Multivariate analysis refers to any of various statistical methods for analyzing more than two variables simultaneously. Assessing effects at the community and ecosystem levels usually involves measuring a large number of abiotic and biotic variables. Assessing each variable individually or with many pairwise bivariate analyses can be cumbersome and difficult to interpret. Multivariate techniques can be used to draw overall patterns from a large set of variables.

There are four broad types of applications for multivariate techniques: ordination, clustering/discrimination, investigating relationships between sets of variables (correspondence), and Bayesian methods.

- Ordination techniques (*e.g.*, principal components analyseis [PCA]) reduce a large set of variables into a smaller set of factors, each of which is a combination of variables that captures as much as possible of the information in the original variables. In this way, a multidimensional set of data can be reduced into a more interpretable form.

- Clustering/discrimination techniques identify natural groupings among sampling units (*e.g.*, most-similar groups of sampling sites) and the parameters that contribute most to this similarity (*e.g.*, abundances of certain species).

- Correspondence analysis techniques (*e.g.*, canonical correspondence analyses [CCA]) identify the degree of covariance between sets of variables (*e.g.*, concentrations of several chemicals versus abundances of several species), as well as identifying the variables within each set that contribute most to this covariance.

- Bayesian statistical methods are becoming increasingly popular in ERA as they are viewed by many as providing legitimate ways of incorporating subjective belief or expert opinion in the form of prior probability distributions. Although the Bayesian approach provides a logical and consistent method for melding prior probabilities with evidence in the form of data, issues regarding subjectivity in the choice of priors and parameterization of complex hierarchical models often arise (Bier, 1999).

Ordination, classification and canonical ordination techniques can be applied to any ecosystem, and are common in DERAs. Additional information is provided below for each group of techniques. Although Bayesian approaches are sometimes used in risk assessment, specialized training is required, and therefore, no generic guidance for its application in risk assessment is provided below.

Ordination

What is it? – Ordination techniques reduce a large set of variables into a smaller set of "derived factors", each of which is a combination of variables that captures as much as possible of the information in the original variables. In this way, a multidimensional set of data can be reduced into a more interpretable form. Commonly used ordination techniques include principal components and factor analysis (PCA and FA), correspondence analysis and detrended correspondence analysis (CA and DCA), and metric and nonmetric multidimensional scaling (MDS and NMDS).

How is it useful in risk assessment? – Ordination is usually treated as an exploratory tool for generating hypotheses and directing additional research. If the reduction in dimensionality is sufficient, the results can be plotted for a visual analysis of relationships among sites or among variables. In some cases, a derived factor is readily interpretable (*e.g.*, as a gradient of contamination) and can be used as a composite variable in additional analyses (*e.g.,* as an explanatory variable in multiple regression). Examples of ordination in DERA include exploring overall trends in a collection of response variables (measurement endpoints) such as in a set of chemical analyses, taxonomic data, or any other set of appropriately-related variables measured at a number of sites.

Issues to consider – Data sets frequently have missing values, skewed or bimodal distributions (*e.g.*, many zeroes for rare species), and categorical or semi-quantitative values. Different techniques have different sensitivities to these common issues. However, all ordination techniques de-emphasize the importance of individual variables (*e.g.*, a particularly sensitive receptor or high-priority COPC) and therefore may mask important information. Important information can also be masked when variables are subject to ordination techniques without consideration of how those variables relate to one another.

Classification

What is it? – Clustering and discrimination techniques identify natural groupings among sampling units (*e.g.*, most-similar groups of sampling sites) and the parameters that contribute most to this similarity (*e.g.*, abundances of certain species or concentrations of certain chemicals). The most commonly used clustering techniques are *k*-means clustering and two-way indicator species analysis (TWINSPAN). The most commonly used techniques to discriminate among established groups are linear discriminant analysis (LDA), Hotelling's T^2, and multivariate analysis of variance (MANOVA).

How is it useful in risk assessment? – Cluster analysis is useful as an exploratory tool to identify natural groupings of measured values in space or time, so that additional analysis or remediation can be stratified and/or focused on 'hotspots' of exposure or effects.

Cluster analysis produces a dendrogram (a tree diagram) in which sites may be grouped at varying levels of similarity. Discrimination techniques can be used to identify the variables that are most strongly associated with an established grouping scheme, to detect statistically-significant multivariate differences among groups, and to derive "rules" for predicting to which group (*e.g.,* impacted versus unimpacted) a new sample belongs.

Issues to consider – The results of cluster analysis may be sensitive to the particular technique used, such as the choice of distance measure (how similarity among cases is calculated). Clustering typically produces ambiguous and/or unstable results when samples are arranged continuously along gradients. As with their univariate counterparts (Student's t and ANOVA), T^2 and MANOVA are sensitive to the assumptions of multivariate normality and constant within-group variances and covariances. Discriminant analysis is often applied to the same set of data for which the rules were derived (the 'training' set), but this gives a highly inflated estimate of the success with which the categorization rules will determine group membership for a new sample. A better approach is to use cross-validation (split-sample or leave-one-out) to test the categorization rules. As with ordination techniques, cluster analysis de-emphasizes the importance of individual variables and may overlook important univariate trends.

Canonical Ordination

What is it? – Canonical ordination techniques explore the degree of covariance between two sets of variables, as well as identifying the variables within each set that contribute most to this covariance. Commonly used techniques include canonical correlation analysis, redundancy analysis (RDA), and canonical correspondence analysis (CCA).

How is it useful in risk assessment? – Canonical ordination techniques can be used to explore the relationship between exposure and effects when one or both of these are multivariate. For example, the data may include a concentration by site matrix for several chemicals and abundance by site matrix for several species. A technique such as CCA will reveal the strength of the overall correspondence (among sites) of abundances (effects) with concentrations (exposure). It is also common to include other site characteristics in this type of analysis to assess to what extent species' abundances are determined by habitat characteristics versus chemical concentrations. In CCA, explanatory variables can be of many types (*e.g.*, continuous, ratio scale, nominal) and do not need to meet distributional assumptions. Hypothesis testing is possible with CCA by means of a randomization test.

Issues to consider – As with regression, one cannot necessarily infer direct causation from canonical ordination techniques. In addition, the independent effects of highly correlated variables (*e.g.*, covarying concentrations of several metals) are difficult to disentangle. The outcome of CCA, in particular, is highly dependent on the scaling of the explanatory variables; logarthmic transformation of explanatory (exposure and

environmental) variables is often appropriate. CCA focuses more on species composition than RDA (which focuses on *relative* abundance); thus, if you have a gradient along which *all* species are positively correlated, RDA will detect such a gradient while CCA will not.

Multivariate techniques can be used to draw general patterns from very complex sets of data. Each technique has associated methods for graphical representation of these general patterns, which can aid in conveying complex ideas to non-technical stakeholders. Multivariate techniques can be used to assess community-level ecological effects, which have more ecological relevance than studies at lower levels of biological organization.

What are the common "pitfalls" or issues that should be considered when using multivariate statistical analyses in a DERA?

- Many multivariate techniques have no established method for determining the statistical significance of observed patterns, and are suitable only for exploratory data analysis.

- Like all statistical methods, there are assumptions that must be carefully assessed before applying multivariate methods. For example, most multivariate methods (except cluster analysis) assume multivariate normality. Most are sensitive to outliers.

- Application of multivariate techniques may require some modification of the field study design, including the appropriate level of replication, the endpoints to be measured, and the taxonomic resolution required.

Where can I go for additional information?

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- Sparks, T.H., W.A. Scott, and R.T. Clarke, R.T. 1999. Traditional multivariate techniques: potential for use in ecotoxicology. *Environmental Toxicology and Chemistry* 18**:** 128-137. (Note: the remainder of the Special Section in ET&C Volume 18 is also of relevance).

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INTERPRETIVE TOOL #6

6.0 PROBABILISTIC METHODS

What does this tool consist of? Probabilistic methods estimate the likelihood of adverse effects, and the probable magnitude of those effects, by incorporating statistical distributions for exposure and/or ecological effects profiles. If exposure concentrations have been measured, the distribution of observed values may be incorporated into the exposure profile to reflect either uncertainty or variability in exposure (but not both, as discussed below). If exposure concentrations are being estimated from a model, simulation methods can be used to generate a distribution of predicted exposures from variability or uncertainty in model structure or input parameters.

The most commonly-used simulation method is Monte Carlo analysis, a technique that randomly generates values for all uncertain or variable parameters and calculates the resulting exposure; many (usually > 10,000) such simulation scenarios give the range of possible exposures, each with an associated probability. Probability bounds analysis is another simulation technique. Ecological effects profiles can also incorporate statistical distributions as the cumulative distribution function of effects (*i.e.*, a dose-response curve) for a single species, or as a species sensitivity distribution for multiple species.

Which ecosystem(s) would this tool typically be applied in? All ecosystems.

How frequently is this tool used in a DERA? Occasional. The method is more frequently applied in complex risk assessments.

What are the benefits of using this tool in a DERA? Probabilistic methods produce very informative risk characterization statements that can include both a probability of observing a particular effect and the probable magnitude of that effect (*e.g.*, "a 90% likelihood of no more than 50% mortality").

What are the common "pitfalls" or issues that should be considered when using this tool in a DERA?

- One should avoid developing probability distributions that intermingle or try to represent both variability and uncertainty because a single probability distribution must be interpretable either as an expression of variability (*e.g.*, 90% of the time, or in 90% of the population) or as an expression of uncertainty (*e.g.*, with 90% confidence). An intermingling of these two interpretations would be meaningless.

- Simulation methods (*e.g.*, Monte Carlo) typically assume that all parameters are independent, and that a particular randomly-chosen value for one parameter will have no influence on the most likely value for another parameter. In reality, many ecological parameters are highly correlated (*e.g.*, feeding rate and growth rate of a species, or feeding rates of several species that are all a function of temperature). There are ways to account for these correlations in simulations, but this requires additional information about the form of the correlation, which is rarely available.

- Simulations do not easily take into account uncertainty in the structure of the model, and will therefore always underestimate to some degree the true uncertainty in model output.

- Monte Carlo simulations require accurate estimates of the magnitude of variability or uncertainty in parameters, and require that you know the form of the distribution of these parameters (*e.g.*, lognormal). These data are often unavailable.

Where can I go for additional information about this tool?

- Haas, C.N. 1999. On modeling correlated random variables in risk assessment. *Risk Anal.* 19:1205-1214.

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