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A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines

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READERS' COMMENTS

This protocol was published as a working document so that the revised methodology can be applied and tested. CCME recognizes that some refinements or changes may become necessary or desirable as scientific understanding of issues related to contaminated sites improves.

Comments on the content of the document may be directed to:

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NOTICE

This document provides the rationale and guidance for developing environmental and human health soil quality guidelines for contaminated sites in Canada. It was originally issued in 1996 in support of the National Contaminated Sites Remediation Program, and has now been revised based on experience gained from application and testing of the protocol, as well as advances in science. This document is intended for general guidance only, and does not establish or affect legal rights or obligations. It does not establish a binding norm, or prohibit alternatives not included in the document and is not finally determinative of the issues addressed. Decisions in any particular case will be made by applying the law and regulations on the basis of specific facts when regulations are promulgated or permits are issued.

OVERVIEW

In response to growing public concern over the potential ecological and human-health effects associated with exposure to contaminated sites, the Canadian Council of Ministers of the Environment (CCME) initiated in 1989 a five-year program entitled the National Contaminated Sites Remediation Program (NCSRP).

To promote consistency and provide guidance in assessing and remediating contaminated sites under this program, CCME released an interim set of numerical environmental quality guidelines in September 1991. The Interim Canadian Environmental Quality Criteria for Contaminated Sites (CCME, 1991a) were established for defined land uses by adopting existing criteria for soil and water used by various jurisdictions in Canada. However, many of the interim criteria for soil were based on professional judgement. This protocol for guidelines derivation was originally developed in 1996 to ensure that revised guidelines are scientifically defensible.

The protocol was subsequently used to develop several soil quality guidelines, most recently published in the Canadian Environmental Quality Guidelines (CCME, 1999 with updates). This document is a revised edition of the protocol. The revisions were based on experience gained while developing soil quality guidelines and the Canada-Wide Standards for Petroleum Hydrocarbons in Soil (CCME, 2000), along with recent advances in the understanding of contaminant fate, transport and toxicology.

The protocol considers the effects of contaminated soil exposure on human and ecological receptors for given land uses. The pathways and receptors of contaminated soil considered in the derivation of soil quality guidelines were selected based on exposure scenarios for agricultural, residential/parkland, commercial, and industrial land uses.

Procedures for deriving environmental soil quality guidelines were developed to maintain important ecological functions that support activities associated with the identified land uses. Guidelines are derived using toxicological data to determine the threshold level on key receptors. Exposure from direct soil contact is the primary derivation procedure for environmental guidelines for agricultural, residential/parkland, commercial and industrial land uses. Another procedure, exposure from contaminated soil and food ingestion, may be considered for certain land uses if there are adequate data. Protection of groundwater for both livestock watering use and transport to nearby surface water bodies with freshwater life are considered using a fate and transport model for certain chemicals. The lowest-value result for all applicable procedures is considered the environmental soil quality guideline.

Deriving human health based soil quality guidelines includes:

- assessing the hazard posed by a chemical;

- determining estimated daily intake (EDI) of that chemical unrelated to any specific contaminated site (i.e. normal "background" exposure); and

- defining generic exposure scenarios appropriate to each land use.

Soil guidelines must ensure that total exposure to a contaminant (EDI + on-site exposure at the guideline concentration) will present negligible risk.

Some of the steps employed to derive human health soil remediation guidelines are similar to those used in a site-specific risk assessment. However, to establish these generic guidelines, several basic assumptions were made about the sensitive receptor and the nature of chemical exposure for each land use. Guidelines derived for non-carcinogens are based on an assumed threshold for toxic effects. For carcinogens presenting some risk at any level of exposure, guidelines are derived based on estimated lifetime incremental cancer risk from exposure to soil.

Chemical constituents in soil can migrate and contaminate other media. For example, soil contaminants can:

- leach into a potable groundwater source or nearby surface water body;

- migrate in a vapour phase into basements and contaminate indoor air;

- be taken up by plants and garden produce.

These important indirect and direct soil exposure pathways are considered in this protocol. Some of these pathways are subject to considerable uncertainty and are evaluated using conservative models which may or may not adjust a guideline value (Management Adjustment Factors). SQGTG uses the term, Management Adjustment Factors, to acknowledge the necessarily imprecise nature of these models, which use conservative point estimates, based on data and professional judgement, for generic input values. Management Adjustment Factors are used in the check mechanisms for migration of soil contaminants into food, or migration of contaminants from industrial sites to more sensitive neighbouring land.

On a site-specific level, more sophisticated models and actual site data for input variables can reduce the uncertainty in these calculations. Generic guidelines can be altered to account for site-specific conditions by removing (or zeroing) exposure pathways, or recalculating management adjustment factors. For more information on setting site-specific objectives, see Section 1.1, Part A.

The final generic soil quality guideline is based on the lowest value generated by the environmental and human health approaches for each of the four land uses: Agricultural, Residential/ Parkland, Commercial, and Industrial.

VUE D'ENSEMBLE

Réagissant aux préoccupations croissantes du public au sujet des risques possibles, pour l’environnement et la santé humaine, d’une exposition aux lieux contaminés, le Conseil canadien des ministres de l’environnement (CCME) a lancé en 1989 un programme quinquennal intitulé Programme national d’assainissement des lieux contaminés (PNALC).

Afin de promouvoir une démarche cohérente et d’orienter l’évaluation et la remise en état des lieux contaminés dans le cadre de ce programme, le CCME a diffusé en septembre 1991 une série de recommandations numériques provisoires pour la qualité de l’environnement. Les Critères provisoires canadiens de qualité environnementale pour les lieux contaminés (CCME, 1991a) ont été établis pour certaines utilisations définies des terrains et s’inspiraient des critères existants pour le sol et l’eau utilisés par diverses instances au Canada. Toutefois, plusieurs des critères provisoires ayant trait au sol étaient basés sur le jugement professionnel. Le protocole a été élaboré à l’origine, en 1996, pour que les recommandations révisées soient scientifiquement justifiables.

Le protocole a par la suite été utilisé pour l’élaboration de diverses recommandations concernant la qualité des sols, récemment publiées dans les Recommandations canadiennes pour la qualité de l’environnement (CCME, 1999, avec mises à jour). Le présent document en constitue une version révisée qui prend en compte l’expérience acquise au fil de l’élaboration des recommandations pour la qualité des sols et des standards pancanadiens relatifs aux hydrocarbures pétroliers dans le sol (CCME, 2000), ainsi que les progrès réalisés récemment au chapitre des connaissances du devenir, du transport et de la toxicité des contaminants.

Le protocole s’intéresse aux effets de l’exposition aux sols contaminés sur les humains et les récepteurs écologiques, pour des utilisations données des terrains. Les voies d’exposition et les récepteurs des sols contaminés pris en compte aux fins de l’élaboration des recommandations pour la qualité des sols ont été choisis en fonction de scénarios d’exposition relatifs aux terrains à vocation agricole, résidentielle/parc, commerciale et industrielle.

Les méthodes d’élaboration des recommandations ont été mises au point afin de maintenir les fonctions écologiques importantes qui sous-tendent les activités liées aux utilisations définies des terrains. Les recommandations sont élaborées à partir de données toxicologiques afin de déterminer les seuils de concentration correspondant à des récepteurs clés. L’exposition par contact direct avec le sol est la principale procédure d’élaboration des recommandations en fonction de l’environnement pour les utilisations des terrains à vocation agricole, résidentielle/parc, commerciale et industrielle. Une autre procédure — l’exposition par ingestion de sol et d’aliments contaminés — pourrait être envisagée pour certaines utilisations des terrains, si nous disposons de données suffisantes. La protection des eaux souterraines pour l’abreuvement du bétail et leur transport jusqu’à des eaux de surface avoisinantes abritant des formes de vie aquatique sont examinés à l’aide d’un modèle du devenir et du transport de certains produits chimiques. Pour toutes les procédures applicables, la recommandation pour la qualité des sols en fonction de l’environnement correspondra à la valeur la plus faible obtenue.

L’élaboration de recommandations pour la qualité des sols en fonction de la santé humaine comprend les étapes suivantes :

- l'évaluation des dangers que présente une substance chimique;

- la détermination de la dose journalière estimée (DJE) de cette substance, sans égard à un quelconque lieu contaminé (c.-à-d. l'exposition « de fond » normale);

- la définition de scénarios génériques d'exposition appropriés pour chaque utilisation des terrains.

Les recommandations pour la qualité des sols doivent faire en sorte que l'exposition totale à un contaminant (DJE + exposition sur le lieu à la concentration prescrite par la recommandation) présentera un risque négligeable.

Certaines des étapes suivies pour élaborer les recommandations pour la qualité des sols en fonction de la santé humaine sont semblables à celles utilisées pour l'évaluation du risque propre à chaque lieu. Toutefois, pour élaborer ces recommandations génériques, plusieurs hypothèses de base ont été formulées au sujet du récepteur sensible et de la nature de l'exposition à une substance chimique pour chaque utilisation des terres. Les recommandations élaborées pour les substances non cancérogènes sont fondées sur un seuil hypothétique produisant des effets toxiques. Quant aux substances cancérogènes qui présentent un risque peu importe le degré d'exposition, les recommandations sont fondées sur la persistance estimée du risque additionnel de cancer attribuable à l'exposition au sol au cours d'une vie.

Les constituants chimiques des sols peuvent migrer et contaminer d'autres milieux. Ainsi, les contaminants du sol peuvent :

- percoler dans une source d'eau souterraine potable ou dans une masse d'eau de surface voisine;

- migrer en phase gazeuse dans les sous-sols et contaminer l'air à l'intérieur des bâtiments;

- être absorbés par les plantes et les produits maraîchers.

Ces voies indirectes et directes importantes d'exposition au sol sont prises en compte dans le présent protocole. Certaines d'entre elles sont extrêmement équivoques et doivent être évaluées à l'aide de modèles conservateurs qui ne seront pas nécessairement dotés de mécanismes d'ajustement (facteurs d'ajustement de gestion). Le GTRQS utilise l'expression « facteurs d'ajustement de gestion » afin de reconnaître la nature nécessairement imprécise des modèles qui utilisent des approximations ponctuelles conservatrices issues des données et du jugement d'un professionnel en guise d'intrants génériques. Ces facteurs sont utilisés dans les mécanismes de vérification de la migration des contaminants du sol dans les aliments, ou de la migration des contaminants d'un terrain à vocation industrielle vers des propriétés avoisinantes plus sensibles.

Pour réduire le degré d'incertitude des calculs associés à un lieu en particulier, on peut utiliser des modèles plus complexes et des données recueillies en conditions réelles. On peut également modifier les recommandations génériques pour prendre en compte les conditions locales en éliminant certains voies d'exposition (ou en les réglant à zéro), ou en recalculant les facteurs d'ajustement de gestion. Pour en savoir plus sur l'établissement des objectifs particuliers à un lieu, voir la partie A, sous-section 1.1.

Les recommandations génériques définitives pour la qualité des sols s’appuient sur la plus faible valeur obtenue à l’aide des méthodes servant à déterminer les effets sur l’environnement et la santé humaine pour chacune des quatre utilisations des terrains : terrains à vocation agricole, résidentielle/parc, commerciale et industrielle.

Table of Contents

READERS' COMMENTS iii
NOTICE iii
Overview iv
Vue d'ensemble vi
Acknowledgements xv
Document Organization xvi
List of frequently Used Acronyms and Abbreviations xvii
Glossary xviii

PART A 1
Section 1: Background and Context 2
1.1 Framework for the Screening and Assessment of Contaminated Sites 2
1.2 The Canadian Environmental Quality Guidelines 4
1.3 Terminology 4
1.4 Summary of Key Changes Since 1996 5

Section 2: National Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines 7
2.1 What is the Protocol? 7
2.2 Guiding Principles 7
2.2.1 Protecting the Environment 9
2.2.2 Protecting Human Health 9
2.3 Land Use 10
2.4 Chemical Classification 12
2.5 Soil Type and Depth 14
2.5.1 Soil type 14
2.5.2 Soil depth 14
2.6 Summary of Guideline Development Process 14

Section 3: Use of Canadian Soil Quality Guidelines 16
3.1 Primary Error in Model Input Parameters 16
3.2 Model Uncertainty 17

PART B 19
Section 1: Derivation of Environmental Soil Quality Guidelines 20

Section 2: Level of Ecological Protection and Relevant Endpoints for Deriving Soil Quality Guidelines 21
2.1 Level of Ecological Protection 21
2.2 Relevant Effects Endpoints 21
2.3 Endpoint Definition 22
2.4 Selection of Ecologically Relevant Endpoints for Guidelines Derivation 23
2.4.1 Short- and Long-term Tests for Soil-dependent Organisms 23
2.4.2 Short- and Long-term Tests for Mammalian and Avian Species 24

Section 3: Status of the Toxicological Database for Soil Related Exposures 25
3.1 Soil-dependent Organisms 25
3.2 Mammalian and Avian Species 25

Section 4: Potential Ecological Receptors and Exposure Pathways of Soil Contamination 26
4.1 Ecological Receptors of Soil Contamination 26
4.2 Exposure Pathways to Soil Contamination 26

Section 5: Exposure Pathways and Key Receptors According to Land Use and Data Availability 29
5.1 Agricultural Land Use 29
5.1.1 Growth of Crops and Plants 29
5.1.2 Maintenance of Livestock and Wildlife 31
5.2 Residential/Parkland Land Use 31
5.2.1 Growth of Ornamental and Native Flora 33
5.2.2 Maintenance of Resident and Transitory Wildlife 33
5.3 Commercial Land Use 33
5.4 Industrial Land Use 34
5.5 Groundwater Protection 34

Section 6: Uncertainty in Guidelines Derivation 38
6.1 Sources 38
6.2 Use of Uncertainty Factors in Guidelines Derivation 38

Section 7: Guidelines Derivation Process 40
7.1 Literature Review 40
7.2 Evaluation of Laboratory and Field Toxicological Data 40
7.2.1 Laboratory Data 42
7.2.2 Field Data 42
7.3 Minimum Toxicological Data Requirements 43
7.4 Environmental Behaviour Considerations 43
7.5 Derivation of Soil Quality Guidelines for Soil Contact 43
7.5.1 Overview 43
7.5.2 Level of Protection 45
7.5.3 Soil Type Considerations 45
7.5.4 Bioavailability Considerations 47
7.5.5 Weight of Evidence Method 48
7.5.6 Lowest Observed Effect Concentration Method 54
7.5.7 Median Effects Method 55
7.5.8 Insufficient Data for Soil Contact Guidelines Derivation 56
7.5.9 Confidence Ranking for the Soil Contact Guideline 56
7.6 Derivation of Soil Quality Guidelines for Soil and Food Ingestion 56
7.6.1 Derivation of Soil Quality Guideline for Soil and Food Ingestion for Protection of Primary Consumers 57
7.6.2 Derivation of Soil Quality Guideline for Soil and Food Ingestion for Protection of Secondary Consumers 64
7.6.3 Derivation of Soil Quality Guideline for Soil and Food Ingestion for Protection of Tertiary Consumers 69
7.6.4 Calculation of Final Soil Guideline for Soil and Food Ingestion 74
7.7 Using Microbial (Nutrient and Energy Cycling) Data for Derivation of Soil Contact Guidelines 74
7.8 Derivation of Soil Quality Guidelines for the Protection of Freshwater Life 74
7.9 Derivation of Soil Quality Guidelines for the Protection of Livestock Watering and Irrigation Water 75
7.10 Derivation of Environmental Soil Quality Guidelines for Offsite Migration 76
7.11 Consideration of Additional Exposure Pathways 76

Section 8: Derivation of the Final Environmental Soil Quality Guidelines 78
8.1 Agricultural Land Use 79
8.2 Residential/Parkland Land Use 79
8.3 Commercial and Industrial Land Use 79

PART C 80
Section 1: Derivation of Human Health Soil Quality Guidelines 81
1.1 Introduction 81
1.2 Guiding Principles for Establishing Human Health Soil Quality Guidelines 82
1.3 Uncertainty in Guidelines Derivation 83

Section 2: Investigation of Contaminant Toxicology 85
2.1 Non-threshold Contaminants 87
2.1.1 Guidelines for Classification of Carcinogenicity and of Mutagenicity in Germ Cells 88
2.2 Threshold Contaminants 89
2.3 Toxicity Benchmarks in the Absence of Health Canada Evaluations 90

Section 3: Exposure to Contaminants 91
3.1 Exposure to Mixtures of Chemicals 91
3.2 Determining Estimated Daily Intake 92

Section 4: Exposure Scenarios 93
4.1 Assumptions about Exposure 93
4.1.1 Threshold Contaminants 95
4.1.2 Non-threshold Contaminants 97
4.2 Absorption of Chemicals into the Body 97
4.3 Receptors, Exposure Pathways, and Land Uses 98
4.3.1 General 98
4.3.2 Defined Scenario for Agricultural Land Use 100
4.3.3 Defined Scenario for Residential/Parkland Land Use 100
4.3.4 Defined Scenario for Commercial Land Use 100
4.3.5 Defined Scenario for Industrial Land Use 101

Secton 5: Human Health Guidelines Derivation 106
5.1 Direct Human Health-Based Soil Guidelines Derivation for Threshold Substances 106
5.2 Direct Human Health-Based Soil Guidelines Derivation for Non-threshold Substances 108
5.3 Guidelines for the Protection of Potable Groundwater 109
5.3.1 General 109
5.3.2 Development of Source Guidance Values for Groundwater 110
5.3.3 Derivation of the SQG_{PW} 110
5.4 Guidelines for the Protection of Indoor Air Quality 111
5.5 Evaluation for Contamination of Produce, Milk, and Meat 112
5.6 Off-site Migration of Soil/Dust from Commercial or Industrial Sites 113
5.7 Consideration of Additional Exposure Pathways 113

Section 6: Derivation of the Final Human Health Soil Quality Guidelines 114
6.1 Agricultural Land Use 114
6.2 Residential/Parkland Land Uses 115
6.3 Commercial and Industrial Land Uses 115

PART D 116
Section 1: Derivation of the Final Soil Quality Guideline 117
1.1 Final Guideline Derivation 117
1.2 Considerations Other than Toxicity 117
1.3 Evaluation Against Plant Nutritional Requirement, Geochemical Background and Practical Quantitation Limits 118
1.4 Provisional Guidelines 121

1.5 Presentation of Soil Quality Guidelines ... 121
1.6 Scientific Supporting Documents ... 122

References ... 123

Appendix A: Partitioning of Contaminants Between Soil, Pore Water, and Soil Vapour ... 131
Appendix B: Nutrient and Energy Cycling Check ... 135
Appendix C: Model for the Protection of Groundwater from Soil Contamination ... 141
Appendix D: Multimedia Exposure Assessment ... 149
Appendix E: Migration of Contaminant Vapours into Buildings ... 152
Appendix F: Checking Protocol for Contaminant Ingestion Resulting from Produce, Meat, and Milk Produced at Remediated Residential and Agricultural Sites. ... 158
Appendix G: Evaluation of Industrial and Commercial Land Use Human Health Guidelines Relative to Adjacent Land Use ... 166
Appendix H: Summary of Models and Equations Used in Soil Quality Guideline Development ... 170
Appendix I: Default Parameters for Guideline Development ... 181

List of Tables

1 Receptors and Exposure Pathways for the Land Use Categories Considered in the Derivation of Environmental Soil Quality Guidelines

2 Required Exposure Pathways for Development of the SQG_E

3 Required Exposure Pathways for Development of the SQG_{HH}

4 Example of Soil Quality Guideline Presentation

F.1 Proposed Values for Local Garden Produce, Meat and Milk Consumption

I.1 Human Receptor Characteristics

I.2 Soil and Hydrogeological Parameters

I.3 Site Characteristics

I.4 Building Parameters

List of Figures

1 National Framework for Contaminated Site Assessment and Remediation
2 Contaminant Assessment Procedure for Deriving Soil Quality Guidelines
3 Concept of Generic Land Uses Envisioned for Guidelines Derivation
4 Simplified Diagram of Potential Ecological Receptors and Exposure Pathways of Contaminated Soil
5 Key Receptors and Exposure Pathways of Contaminated Soil Considered for Agricultural Land Use
6 Key Receptors and Exposure Pathways of Contaminated Soil Considered for Residential/Parkland Land Use
7 Key Receptors and Exposure Pathways of Contaminated Soil Considered for Commercial Land Use
8 Key Receptors and Exposure Pathways of Contaminated Soil Considered for Industrial Land Use
9 Overall Procedure for Deriving Environmental Soil Quality Guidelines for Agricultural, Residential/Parkland, Commercial and Industrial Land Uses
10 Procedure for Deriving Soil Quality Guidelines for Soil Contact for Agricultural and Residential/Parkland Land Use
11 Procedure for Deriving Soil Quality Guidelines for Soil Contact for Commercial and Industrial Land Use
12 Example of a Rank Probability Plot of Bioassay Data
13 Procedure for Deriving Soil Quality Guidelines for Soil and Food Ingestion
14 Generalized Framework for Human Health Risk Assessment
15 Significant Pathways of Human Exposure to Environmental Contamination
16 Conceptual Derivation of the Soil guideline for Threshold Substances from the Multimedia Exposure Assessment and Assumed Soil Apportionment (20%) from the Residual Tolerable Daily Intake
17 Exposure Assumptions for Defined Agricultural Land Use Scenario
18 Exposure Assumptions for Defined Residential/Parkland Land Use Scenario
19 Exposure Assumptions for Defined Commercial Land Use Scenario
20 Exposure Assumptions for Defined Industrial Land Use Scenario
21 Overview of Steps Leading to Derivation of a Final Soil Quality Guideline

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DOCUMENT ORGANIZATION

This document is divided into four parts. A glossary of terms is presented at the beginning of the document. Background information on the development of the Protocol, including the scientific tools that have been developed to help assess and remediate contaminated sites in Canada is provided in Part A. Information on the principles of the soil quality guidelines derivation protocol is also included in Part A. The processes for deriving environmental and human health guidelines are described in Part B and Part C. Part D concludes this document by providing guidance on derivation of the final soil quality guideline. Methods and models employed in the ecological sections of the document, and models and check mechanisms for indirect exposure from soil contaminants for the human health guidelines, are provided in the Appendices.

LIST OF FREQUENTLY USED ACRONYMS AND ABBREVIATIONS

AF	Relative absorption factor
BAF	Bioaccumulation factor
BCF	Bioconcentration factor
BW	Body weight
CCME	Canadian Council of Ministers of the Environment
DMIR	Dry matter intake rate
DTED	Daily threshold effects dose
ECL	Effects concentration – low
EC_X	Effective concentration – X%
EDI	Estimated daily intake
$ESSD_X$	Estimated species sensitivity distribution – X^{th} percentile
FIR	Food ingestion rate
IC_X	Inhibition concentration – X%
LC_X	Lethal concentration – X%
LO(A)EL	Lowest observed (adverse) effects level
LOEC	Lowest observed effects concentration
MRL	Maximum residue limit
NCSRP	National Contaminated Sites Remediation Program
NO(A)EL	No observed (adverse) effects level
NOEC	No observed effects concentration
RSC	Risk-specific concentration
RSD	Risk-specific dose
RTDI	Residual tolerable daily intake
SAF	Soil allocation factor
SIR	Soil ingestion rate
SQG	Soil quality guideline
SQG_E	Soil quality guideline – environmental
SQG_{HH}	Soil quality guideline – human health
SQGTG	Soil Quality Guidelines Task Group
TC	Tolerable concentration
TDI	Tolerable daily intake
TEC	Threshold effects concentration
UF	Uncertainty factor

GLOSSARY

Absorption: The process by which a chemical enters the circulatory system following ingestion, inhalation, or dermal exposure. If sufficient data are not available, it is typically assumed that absorption of the chemical from the exposure medium is equivalent to absorption of the chemical in the critical toxicity study, although this is expected to overestimate the actual rate of absorption in many cases.

Adsorption: The physical process of attracting and holding molecules of other substances or particles to the surfaces of solid bodies with which the former are in contact with.

Acceptable risk: A risk that is so small and consequences so slight, or the associated benefits (perceived or real) so great, that society is willing to take or be subjected to that risk.

Acute exposure: See short-term exposure.

Advective Flow: A process that transports a chemical from one location to another by virtue of the fact that the chemical is a component of a moving physical system (e.g. wind, flowing water, sediment transport).

Analytical detection limit: The lowest concentration which can be determined with confidence to be different from zero.

Appreciable risk: An estimated rate of incidence or frequency of disease, or level of chemical exposure considered significant. Appreciable risks must be defined chemical-by-chemical, consider all potential sources of exposure, and the critical hazard attributed to that exposure.

Aquifer: Groundwater-bearing formations sufficiently permeable to transmit and yield water in usable quantities.

Assessment endpoint: The characteristic of the ecological system that is the focus of the risk assessment. Formal expressions of the actual environmental value to be protected (e.g. fishable, swimmable water).

Average person: To conduct a risk assessment, many physical characteristics of "typical" Canadians have been measured and defined. Assumed characteristics of an average person are included in Appendix I. Data for other characteristics, age groups, and differences by sex are also available.

Background concentration: A representative ambient level for a contaminant in soil or water. Ambient concentrations may reflect natural geological variations in relatively undeveloped areas or the influence of generalized industrial or urban activity in a region.

Bioaccumulation: The process by which chemical compounds are taken up by terrestrial and aquatic organisms directly from the medium and through consuming contaminated food at a faster rate than the compounds are lost through excretion or metabolism.

Bioaccumulation factor: The ratio of the concentration of a chemical compound in an organism to the concentration in the exposure medium, based on uptake from the surrounding medium and food.

Bioassay: A controlled, experimental study in which organisms (typically rodents) are exposed to several different doses/concentrations of a chemical over a predetermined time (see long- and short-term exposure), and the resulting effects are identified and measured.

Bioavailability: The amount of chemical available to the target tissues following exposure.

Bioconcentration: The process by which contaminants are directly taken up by terrestrial and aquatic organisms from the medium. Normally refers to the situation where resulting concentrations in the organism are higher than concentrations in the medium (e.g. soil or water).

Bioconcentration factor (BCF): The ratio of the concentration of a chemical compound in an organism to the concentration of the compound in the exposure medium (e.g. soil or water).

Biodegradation: A microbiologically mediated process (e.g. due to the action of bacteria, yeasts and fungi) that chemically alters the structure of a chemical, the common result being the breakup of the chemical into smaller components.

Biomagnification: The process of bioaccumulation by which tissue concentrations of accumulated chemical compounds are passed up through two or more trophic levels so that tissue residue concentrations increase systematically as trophic level increases.

Biota: Biological organisms (including plants, microbes, invertebrates, and animals).

Biotransfer factor: The ratio of the chemical compound concentration in fresh (wet) weight tissue (e.g., meat, milk, etc.) to the intake of the chemical compound by the organism in mass per day.

Carcinogen: A substance or agent that causes the development or increases the incidence of cancer. A carcinogen can also act upon a population to change its total frequency of cancer in terms of numbers of tumours or distribution by site and age.

Carcinogenic: Of or pertaining to the ability to cause the development of cancer.

Carcinogenic potency: Expressed as a concentration or dose that induces a 5% increase in the incidence of, or death due to tumours or heritable mutations associated with exposure.

Cation exchange capacity: The total amount of exchangeable cations that a soil can absorb.

Cation exchange: The interchange between a cation in solution and another cation on the surface of any surface-active material (e.g., clay or colloidal organic matter).

Check mechanism: An exposure pathway which may or may not be considered in the determination of the final soil quality guideline, based on professional judgement or jurisdictional policy decisions.

Chemoautotroph: Organisms deriving energy from the oxidation of inorganic compounds and using carbon dioxide as the principal carbon source for organic synthesis.

Chronic exposure: See long-term exposure

Clay: Soil minerals of equivalent diameter < 0.002 mm usually consisting of clay minerals but commonly including amorphous free iron oxides and primary minerals.

Clay mineral: Finely crystalline hydrous aluminum silicates and hydrous magnesium silicates with a phyllosilicate structure.

Coarse-grained soils: Soil which contains greater than 50% by mass particles greater than 75 μm mean diameter (D_{50} > 75 μm).

Consumers: Organisms which require energy in the form of organic material from external food sources (heterotrophs).

Contaminant: Any substance present in an environmental medium at concentrations in excess of natural background.

Criteria: Generic numerical limits or narrative statements intended as general guidance for the protection, maintenance, and improvement of specific uses of soil.

Critical hazard: The "critical" hazard is the health effect that occurs at the lowest dose level defined by the most suitable bioassay or other health study, and is the effect from which the no observed (adverse) effect level/lowest observed (adverse) effect level is defined to derive soil guidelines.

Critical receptor: The taxon, cohort, and developmental stage believed to be the most biologically sensitive among a larger group that is potentially exposed to a contaminant (e.g. for humans, children (especially toddlers) are often critical receptors for non-cancer causing substances).

Critical threshold: The dose/concentration below which no adverse effect is expected to occur.

Cross-media transfer: When chemicals migrate and disperse from one medium to another. Chemicals in soil will leach into groundwater and volatilize, "transferring" contamination from one medium to another.

Decomposition: The chemical and physical breakdown of organic matter accompanied by mass loss.

De minimis risk: A trivial or negligible risk. In practical terms, when a risk is de minimis there is no incentive to modify the activity causing the risk.

Delivered dose: The amount or concentration of a substance at the targeted site within the body. The delivered dose may consider metabolic activation processes, pharmacodynamics, and tissue dosimetry.

Denitrification: The gaseous loss of nitrogen from soil, water, or sediment by biological and chemical reduction of nitrate to compounds other than ammonia.

Detritus: Organic debris from decomposing plants and animals.

DMIR (Dry matter intake rate): The combined rates of soil ingestion and food ingestion (dry weight) for an animal.

Dose: The amount or concentration of a substance taken in or absorbed into the body exposed to the substance.

DTED (Daily threshold effect dose): The dose of a chemical below which only minimal effects would be expected to occur in an animal.

EC_{25}: Effective Concentration 25: The concentration of a chemical in the medium that results in some sublethal effect to 25% of the test organisms. The EC_{25} is normally reported as a time dependent value with the sublethal endpoint observed (e.g., 5-day EC_{25}, reproduction). Effective concentrations can also be specified for other percentiles (e.g. the EC_{50} would result in an effect to 50% of the test organisms).

Ecological receptor: A non-human organism potentially experiencing adverse effects from exposure to contaminated soil either directly (contact) or indirectly (food chain transfer).

Ecosystem: A community of organisms, interacting with one another and with the environment.

Effects-based: The use of adverse effects data from toxicological studies to form the basis for guidelines derivation. In this document, effects-based pertains to data from toxicological tests on organisms to support guidelines derivation.

ECL (Effects concentration low): The concentration of a chemical in soil expected to result in some effects to soil organisms, but less than the concentration resulting in median lethality in populations. The desired level of protection for commercial and industrial sites.

EDI (Estimated daily intake): Total "background" exposure to a chemical experienced by most Canadians. Estimated daily intake arises from the low levels of contamination commonly found in air, water, food, soil, and consumer products (e.g., tobacco, paints, and medicines). Estimated daily intake of a chemical is determined through a multimedia exposure assessment.

Endpoint measurement: An effect on an ecological component that can be measured and described quantitatively.

Epidemiology: The study of the distribution and determinants of disease frequency in humans. Epidemiology may involve the observation of unusual clusters of a rare disease, descriptive statistics on morbidity and mortality patterns, ecologic studies correlating disease occurrence rates with geographic or spatial risk factors, and analytical studies of the relationship between disease occurrence rates and exposure to particular toxicants.

Evapotranspiration: The loss of water from a given area during a specified period by evaporation from the soil surface and by transpiration from plants.

Exposure: Contact between a substance and an individual or population. Exposure may occur via different routes including ingestion, dermal absorption, and inhalation.

Exposure characterization: Identification of the conditions of contact between a substance and an individual or population. Exposure characteristics may involve identifying concentration, routes of uptake, target sources, environmental pathways, and the population at risk.

Exposure estimation: Estimate of the amount and duration of contact between a substance and an individual or a population. Exposure estimates consider factors like concentration, routes of uptake, target sources, environmental pathways, population at risk, and time scale.

Exposure pathway: The route by which an organism comes into contact with a contaminant. In the ecological effects-based procedure, exposure pathways are restricted to organisms in contact with contaminated soil. In the human health-based procedure, exposure pathways include contact through consumption of contaminated food, direct soil ingestion, dust inhalation, dermal absorption, inhalation of contaminant vapours, and ingestion of contaminated groundwater.

Exposure route: The mode of entry of a chemical into the body. The three basic exposure routes are ingestion, inhalation, and dermal absorption.

Exposure scenario: A clearly and quantitatively defined description of all circumstances associated with a receptor that would permit the estimation of chemical exposure. These circumstances include amount of air breathed, food and water consumed, soil ingested, and the critical receptor's weight, age, sex, and all other relevant considerations.

Fine-grained soils: Soils which contain greater than 50% by mass particles less than 75 µm mean diameter (D_{50} < 75 µm).

Geo-environment: The vadose and saturated zones of the earth—excluding surface water bodies—participating in or communicating with the biosphere.

Guidelines: Generic numerical limits or narrative statements that are recommended to protect and maintain the specified uses of water, sediment, or soil, (referred to as criteria in some previous CCME publications).

Groundwater: Subsurface water beneath the water table in fully saturated geologic formations.

Groundwater recharge: Process which occurs when the water content of the unsaturated zone becomes high enough to cause excess water to percolate downward to the water table, usually as a result of the infiltration of snow melt or rainwater into surface soils. Using a water balance approach, recharge is equal to the total amount of precipitation less the amount of surface runoff and evapotranspiration.

Habitat: A particular type of environment inhabited by an organism.

Hazard: The adverse impact on health that can result from exposure to a substance. The significance of the adverse effect depends on the nature and severity of the exposure to the substance and the degree to which the effect is reversible. In some instances, the substance itself is also referred to as the hazard rather than the adverse effect that the substance can cause.

Hazard identification: Identification of effects capable of adversely affecting health as a result of exposure to a substance. Hazard identification may involve case reports, toxicological studies, epidemiological investigations, or structure/activity analysis.

Henry's Law constant: A partition coefficient defined as the ratio of a chemical's concentration in air to its concentration in water at steady state. The dimensionless Henry's Law constant is obtained by dividing the Henry's Law constant by the gas constant R.

Heterotroph: Organism that requires carbon in the organic form.

Hydraulic conductivity (K_H): The proportionality factor between hydraulic gradient and flux in Darcy's Law. Hydraulic conductivity measures the inherent ability of a porous medium to conduct water.

IC_{25}: Inhibition Concentration 25: The concentration of the chemical in a medium which results in a 25% response to a measured endpoint (e.g. growth, reproduction) for the test organism. The IC_{25} is normally reported as a time dependent value with the sublethal endpoint observed (e.g., 28-day IC_{25}, plant growth). Inhibition concentrations can also be specified for other percentiles (e.g. the IC_{50} would result in a 50% effect to the measured parameter).

Incremental risk: Risk due to exposure to a chemical in excess of the "background" risk.

LC_{50} (Median lethal concentration): The concentration of chemical in the medium that results in mortality to 50% of the test organisms. The LC_{50} is usually expressed as a time-dependent variable (e.g., 96-hr LC_{50}). The LC_{50} is normally statistically derived through analysis of mortality data from all test concentrations.

Leaching: The process by which contaminants in soil dissolve into percolating water (e.g., rainfall) and are gradually removed from the soil.

Lipophile: A substance that tends to dissolve in organic, non-polar solvents. Lipophilic substances generally have very low water solubility.

LO(A)EL (Lowest observed (adverse) effect level): The lowest dose in a bioassay that results in observed effects in the exposed organisms. In some cases, observed effects may be of questionable impact or possibly beneficial. Therefore, obvious negative effects may be differentiated as "adverse".

LOEC (Lowest Observed Effect Concentration): The lowest concentration of a chemical used in a toxicity test that has a statistically significant adverse effect on test organisms relative to a control.

Log K_{ow} (Log n-Octanol/Water Partition Coefficient): The logarithm (base 10) of the ratio of a substance's solubility in n-octanol and in water at equilibrium; also expressed as log P. Log K_{ow} indicates a substance's tendency to bioaccumulate in terrestrial and aquatic biota.

Long-term exposure: Exposure to a contaminant in a medium lasting from several weeks to years and often includes a reproductive or life cycle of the test organism. Usually referred to as a chronic exposure. Absolute definitions for this term vary among studies (see Part A, 3.1).

Macronutrient: A chemical necessary in large amounts, usually greater than 1 ppm, for plant growth.

Measurement endpoint: An effect on an ecological component that can be measured and described in some quantitative fashion (e.g., EC_{50}).

Mineralization: The decomposition of an element from an organic to an inorganic state.

Mineral soil: A soil consisting predominantly of mineral matter (organic carbon < 17%), except for an organic surface layer that may be up to 40 cm thick.

Multimedia exposure assessment: The quantitative estimate of total exposure to a chemical arising from all sources (air, water, soil, food, consumer products), by all routes (ingestion, inhalation, dermal absorption).

Mutagenicity: The ability of a chemical to produce a permanent change in the genetic material.

N-fixation: The conversion of elemental nitrogen (N_2) to organic combinations or to forms readily useable in biological processes.

Negligible risk: Risk that is considered negligible if the estimated carcinogenic potential of the contaminant in the soil is very small compared to exposure risks from other media (i.e., air, water, food). See also *de minimis* risk.

Nitrification: The biological oxidation of ammonium to nitrate.

NO(A)EL {No observed (adverse) effect level}: The highest dose in a bioassay with no observed effects in the exposed organisms. In some cases, observed effects may be of questionable impact or may possibly be beneficial. Therefore, obvious negative effects may be differentiated as "adverse".

NOEC (No Observed Effect Concentration): The highest concentration of a contaminant used in a toxicity test that has no statistically significant adverse effect on the exposed population of test organisms relative to a control.

Non-threshold contaminant: A contaminant for which there is considered to be some probability of human harm at any level of exposure.

Objective: A numerical limit or narrative statement that has been established to protect and maintain a specified use of soil at a particular site by taking into account site-specific conditions.

Pathway-specific guideline: A pathway-specific value which is always considered in the determination of the final soil quality guideline if data requirements are met.

pH: A measure of acidity -- technically, the negative logarithm of hydrogen ion activity.

Porewater: The water occupying the space between particles of sediment or soil.

Porosity: The volume proportion of the total bulk not occupied by solid particles.

Practical quantitation limit: The lowest concentration that can be quantified with a suitable level of accuracy and precision.

Probability: The likelihood or frequency of occurrence of an adverse health effect.

Producers: Organisms which undergo photosynthesis to convert CO_2 and H_2O into sugars (autotrophs).

Receptor/critical receptor: A receptor is the person or organism exposed to a chemical. For human health risk assessment, it is common to define a critical receptor as the person expected to experience the most severe exposure (due to age, sex, diet, lifestyle, etc.) or most severe effects (due to state of health, genetic disposition, sex, age, etc.) as a result of that exposure.

Reference concentration (RfC): An estimate (with uncertainty spanning perhaps an order of magnitude) of continuous inhalation exposure to the human population, including sensitive subgroups, that is likely to be without appreciable risk of deleterious effects during a lifetime. The RfC is used to evaluate potentially non-carcinogenic effects only. See also tolerable concentration.

Regolith: The unconsolidated mantle of weathered rock and soil overlying solid rock.

Remediation: The management of a contaminated site to prevent, minimize, or mitigate damage to human health or the environment. Remediation may include both direct physical actions (e.g., removal, destruction, and containment of contaminants) and institutional controls (e.g., zoning designations or orders).

Required pathway: An operative pathway without which a final soil quality guideline cannot be determined (e.g. ecological soil contact or human direct soil contact pathways).

Respiration: Metabolic processes leading to the production of carbon dioxide from reduced organic substrates.

Risk: In this protocol, risk is a measure of both the severity of health effects from exposure to a substance and the probability of its occurrence. Risk may involve quantitative extrapolation from animals to humans or from high dose/short exposure time to low dose/long exposure time. Risk may consider potency (physical/chemical properties, biological reactivity), susceptibility (metabolic activation, repair mechanisms, age, sex, hormonal factors, immunological status), level of exposure (sources, concentration, initiating events, routes, pathways), and adverse health effects (nature, severity, onset, reversibility).

Risk analysis: The process of risk assessment, management, and communication. In addition to the scientific considerations involved in risk assessment, risk analysis considers such factors as risk acceptability, public perception of risk, socio-economic impacts, benefits, and technical feasibility.

Risk assessment: A procedure designed to determine the qualitative aspects of hazard identification, and usually a quantitative determination of the level of risk based on deterministic or probabilistic techniques.

Risk estimation: Estimate of the level of risk involving statistical analysis of toxicological and epidemiological data and of the level of human exposure. Risk estimation examines the severity, extent, and distribution of the effects of an event or activity and leads to a specific numerical point estimate or a range of values.

Risk management: The selection and implementation of a strategy for control of a risk, followed by monitoring and evaluation of the effectiveness of that strategy. The decision to select a particular strategy may involve considering the information obtained during risk assessment. Implementation typically involves a commitment of resources and communication with affected parties. Monitoring and evaluation may include environmental sampling, post-remedial surveillance, prospective epidemiology, and analysis of new health risk information, as well as ensuring compliance.

Risk perception: An intuitive judgement about the nature and magnitude of a risk. Perceptions of risk involve the judgements people make when they characterize and evaluate hazardous substances, activities, and situations.

RSC (Risk-specific concentration): The concentration of a chemical (normally in air) expected to lead to a specified cancer risk (e.g. 1 in 1 million). The RSC can only be specified for chemicals with non-threshold effects (i.e. carcinogens).

RSD (Risk-specific dose): The dose of a chemical expected to lead to a specified cancer risk (e.g. 1 in 1 million). The RSD can only be specified for chemicals with non-threshold effects (i.e. carcinogens).

RTDI (Residual tolerable daily intake): The dose of a chemical above the background exposure to which a person could be exposed without expected adverse effects. RTDI = TDI – EDI.

Runoff: The portion of the total precipitation on an area that flows into stream channels. Surface runoff does not enter the soil. Groundwater runoff or seepage flow enters the soil before reaching the stream.

Safety factor: A unitless numerical value applied to a reference toxicological value (e.g., EC_{50}) to account for the uncertainty in the estimate of a final soil quality guideline. Uncertainty factors may be applied, for example, when there is a need for extrapolation to long-term values from short-term data, extrapolation from laboratory to field conditions, or to account for inter- or intra-specific variation between individual test organisms and species.

Sand: A soil particle between 0.075 and 2 mm in diameter.

Short-term exposure: A short-term exposure to a contaminant in a medium, usually severe enough to rapidly induce an effect. Often referred to as an acute exposure. Absolute definitions for short-term exposure vary from study to study.

Silt: A soil particle between 0.002 and 0.075 mm in equivalent diameter.

Slab-on-grade: Building foundation built as a concrete slab directly on the ground surface with no basement.

Soil: Normally defined as the unconsolidated material on the immediate surface of the earth that serves as a natural medium for terrestrial plant growth. Here limited to unconsolidated, surficial, mineral materials.

Soil allocation factor (SAF): The relative proportion which it is allowable for soil to constitute in the RTDI (Residual Tolerable Daily Intake) from various environmental pathways (air, soil, food, water, consumer products).

Soil-dependent organisms: Organisms that use the soil medium as primary habitat, have direct contact with soil, and require soil to sustain normal biological function. In this document, these organisms are limited to plants, invertebrates, and microorganisms.

Soil organic matter: The organic fraction of soil; includes plant and animal residues at various stages of decomposition, cells and tissues of soil organisms, and substances synthesized by the soil population. It is usually determined on soils that have been sieved through a 2.0 mm sieve.

Solubility: The maximum concentration of a chemical that can be dissolved in a liquid (usually water, unless otherwise specified) when that liquid is both in contact and at equilibrium with the pure chemical.

Subsoil: Unconsolidated regolith material above the water table not subject to soil forming processes. Nominally includes vadose zone materials below 1.5 m depth.

Surface Soil: Unconsolidated regolith material near the ground surface; nominally includes topsoil and vadose zone materials to a depth of 1.5 m.

TEC (Threshold effects concentration): The concentration of a chemical in soil below which minimal effects to soil organisms is expected. The desired level of protection for agricultural and residential/parkland land uses.

Teratogenicity: The ability of a chemical to change the normal development processes of an unborn organism, resulting in permanent alterations in the biochemical, physiological, or anatomical functions of the organism.

Texture: Categorical description of the proportions of sand, silt, and clay present in a soil.

Threshold: The dose/concentration of a chemical below which no adverse effect is expected to occur.

Threshold contaminant: A contamination for which there is a dose/concentration below which no adverse effects are expected to occur.

Tolerable concentration (TC): The concentration of a chemical (normally in air) to which a person may be exposed with no expected adverse effects. A tolerable concentration can only be determined for chemicals with threshold effects (i.e., non-carcinogens).

Tolerable daily intake: The level/rate of chemical exposure to which a person may be exposed with no expected adverse effects. A tolerable daily intake can only be determined for chemicals with threshold effects (i.e., non-carcinogens).

Tolerable incremental exposure: The additional exposure which a person may experience, over and above background estimated daily intake, but not exceeding the tolerable daily intake for that substance.

Toxic: Adverse effect (e.g., reduced survival of a population, growth inhibition, or reduced reproduction rates) which occurs in an organism, or population of organisms due to exposure to a contaminant.

Trophic level: Position in the food chain determined by the number of energy transfer steps to that level.

Uncertainty factor: See safety factor.

Unconfined aquifer: A region of saturated ground material unbound by an impermeable or low-permeability layer such as clay. These systems allow for the draining of soil porewater and the subsequent movement of air (or water) to fill the spaces vacated by the moving water.

Vadose zone: Refers to the upper portion of the unsaturated zone in the subsurface environment, where both air and water are present between mineral grains.

Volatilization: The chemical process by which chemicals spontaneously convert from a liquid or solid state into a gas and then disperse into the air above contaminated soil.

PART A

SECTION 1
BACKGROUND AND CONTEXT

1.1 Framework for the Screening and Assessment of Contaminated Sites

In response to public concern over the potential environmental and human health effects associated with contaminated sites, the Canadian Council of Ministers of the Environment (CCME) initiated the National Contaminated Site Remediation Program (NCSRP) in October 1989. This five-year federal-provincial/territorial program was intended to provide a common framework and scientific tools for the consistent, scientifically defensible and cost-effective assessment and remediation of contaminated sites.

In 1990, CCME held multi-stakeholder workshops to discuss key factors in the development of a national framework for the management of contaminated sites. Key recommendations from the workshops included the need for:

- a consistent risk-based approach to evaluate and set priorities for remediation of contaminated sites;

- a tiered approach to assessment and remediation with generic national criteria (or guidelines) and guidance on site-specific objectives; and

- equal protection of human health and the environment.

It was also recognized that effective implementation of these recommendations would require the development of a number of supporting scientific tools. As a result of the workshop, CCME established the Subcommittee on the Classification of Contaminated Sites and the Sub committee on Environmental Quality Criteria for Contaminated Sites. These subcommittees were responsible for the production of a number of scientific tools between 1991 and 1996, including the following:

- National Classification System for Contaminated Sites (CCME, 1992)

- Interim Canadian Environmental Quality Criteria for Contaminated Sites (CCME, 1991)

- Subsurface Assessment Handbook for Contaminated Sites (CCME, 1994)

- Guidance Manual on Sampling, Analysis and Data Management for Contaminated Sites, Volume I and II (CCME, 1993)

- A Framework for Ecological Risk Assessment: General Guidance (1996) and Technical Appendices (1997) (CCME, 1996a; 1997a)

- A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines (CCME, 1996b)

- Guidance Manual for Developing Site-specific Soil Quality Remediation Objectives for Contaminated Sites in Canada (CCME, 1996c)

The above tools are integral to a tiered framework for the screening and assessment of contaminated sites, through which site managers develop remediation objectives. The framework relies on generic guidelines (Tier 1) and site-specific objectives (Tiers 2 and 3) and is illustrated in Figure 1. The generic guidelines are simple numerical values, based on generic scenarios developed for different land uses, and employ conservative assumptions. The development of generic guidelines is the focus of the original Protocol document (CCME, 1996b). Generic guidelines help evaluate the relative risk posed by contaminants at a site, but may not always be an appropriate remediation goal. To tailor clean-up levels to a site, site-specific remediation objectives may be developed, either by modifying (within limits) the generic remediation objectives based on site-specific conditions (Tier 2) or by conducting a human health and/or ecological risk assessment (Tier 3).

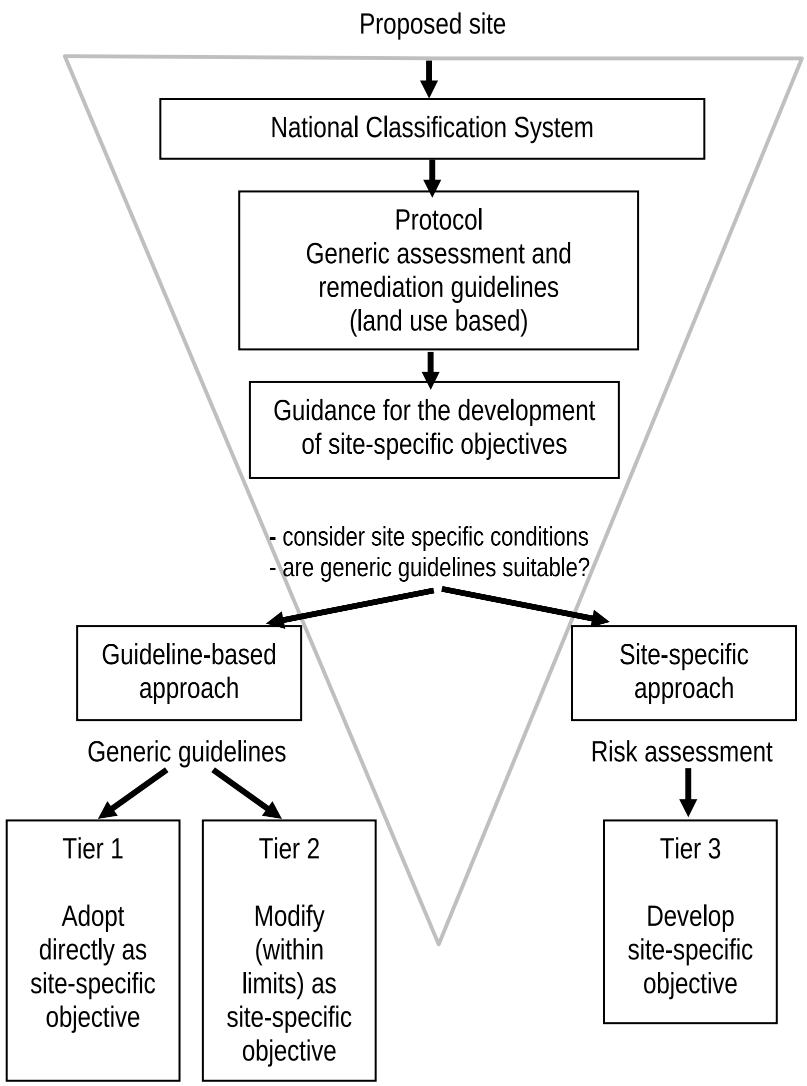


Figure 1: National Framework for Contaminated Site Assessment and Remediation

1.2 The Canadian Environmental Quality Guidelines

In 1997, CCME's Soil Quality Guidelines Task Group (SQGTG) issued the following document:

- Recommended Canadian Soil Quality Guidelines (CCME, 1997b)

The document contained soil quality guidelines, determined using the 1996 protocol, for 20 substances and four land uses, which were intended to replace the 1991 interim criteria for these substances.

Since 1996, SQGTG and a number of other task groups, working groups and committees have been working towards the creation of an integrated set of national environmental quality guidelines for all environmental media (water, soil, sediment, tissue residue and air). These were first published in 1999 in a document entitled Canadian Environmental Quality Guidelines (CCME, 1999). This document contained updated soil quality guidelines for a further 12 substances determined according to the 1996 protocol. Since 1999, soil quality guidelines have been determined for additional substances and adopted as replacements for the corresponding1991 criteria.

The *Canada-wide Accord on Environmental Harmonization*, signed in 1998 by all CCME members with the exception of Quebec, provides an additional mechanism for the development of environmental quality guidelines through the *Canada-wide Environmental Standards Sub-agreement*. The focus of the latter is the development of ambient environmental standards for the quality of various environmental media. Under this mechanism, the Canada-wide Standards (CWS) for Petroleum Hydrocarbons (PHC) in Soil were endorsed in 2001 (CCME, 2001), together with a number of supporting documents. The development of the PHC CWS was based in part on the 1996 protocol, but also incorporated revised procedures and features of other approaches and protocols.

Over the period since 1996, a number of modifications and improvements have been made to the protocol based on scientific and regulatory experience gained through the development of the subsequent soil quality guidelines and the PHC CWS, and as a result of shifts or developments in the science. However, these modifications and improvements have not been hitherto formalized. The present version of the protocol, therefore, is intended as an update to the 1996 version, reflecting the most current approaches to the development of soil quality guidelines.

1.3 Terminology

The use and interpretation of the terms *guidelines*, *objectives* and *standards* vary among different agencies and countries. Previous CCME publications about the National Contaminated Sites Remediation Program used the term *soil criteria*. This term has now been replaced by *guidelines* for consistency with other environmental media (water, sediments, etc.). For the purpose of this document, these terms are defined as follows:

Guidelines – numerical limits or narrative statements recommended to support and maintain designated uses of the soil environment.

Objectives – numerical limits or narrative statements established to protect and maintain designated uses of the soil environment at a particular site.

Standards – guidelines or objectives recognized in enforceable environmental control laws of one or more levels of government.

1.4 Summary of Key Changes Since 1996

General

- Separate guidelines are now determined for coarse (sand) and fine (silt/clay) soils where appropriate (generally for organic chemicals).

- Default parameters for some models have been revised. All equations are summarized in Appendix H, and default parameters are summarized in Appendix I.

- The applicable exposure pathways for various types of chemicals have been clarified.

- Provision for non-toxicity based endpoints (aesthetics, free product formation, etc.) as a check mechanism has been incorporated into the Protocol.

- Adjustment of the soil allocation factor for chemicals which may not be present in all media (soil, water, air, food and consumer products) is now permitted.

Ecological Soil Contact

- The preferred approach for the weight-of-evidence method is to use ecological toxicity endpoint response levels standardized to the 25% level (EC_{25} values).

- Nutrient and energy cycling processes are now evaluated separately from the ecological soil contact pathway.

Protection of Indoor Air Quality (Vapour Inhalation) Pathway

- A vapour intrusion model developed by Johnson and Ettinger (1991) is now applied to calculate guidelines for this pathway.

Protection of Groundwater

- Protection of freshwater life in nearby water bodies, livestock watering, and irrigation water are now considered in addition to the protection of potable groundwater. These pathways are considered as guideline values, as opposed to check values, and therefore are included in the overall soil quality guideline.

Soil and Food Ingestion by Livestock/Wildlife

- Secondary and tertiary consumers are now considered for substances that biomagnify.

Offsite Migration

- Offsite migration is evaluated for both environmental and human health soil quality guidelines, and is applied for commercial as well as industrial land uses.

SECTION 2
NATIONAL PROTOCOL FOR THE DERIVATION OF ENVIRONMENTAL AND HUMAN HEALTH SOIL QUALITY GUIDELINES

2.1 What is the Protocol?

The protocol was originally developed to provide a method for replacing the 1991 interim remediation criteria for soil with scientifically defensible generic guidelines for contaminated sites. The protocol provided stakeholders (i.e., public, industry, and regulatory agencies) with the basic concepts and methods employed in guidelines development. Both scientific and management considerations are necessary when deriving generic guidelines. CCME is committed to document and present all technical, scientific, and management considerations that are used to derive the guidelines. Under the leadership of SQGTG, the protocol has been updated to reflect modifications and improvements in the methods for developing generic guidelines, based on recent scientific and regulatory experience.

The protocol examines the steps needed to generate effects-based soil remediation guidelines including a rationale for the choice of receptors, exposure pathways under certain land uses, assumptions, and describes acceptable and minimum data requirements for guidelines derivation.

The guidelines are developed and/or revised on a substance-by-substance basis as required, in accordance with the protocol, after a comprehensive review of the physical/chemical characteristics, background levels in Canadian soils, toxicity and environmental fate and behaviour of each substance (see Figure 2). This supporting information is presented in a series of guideline-supporting technical documents from Environment Canada and/or Health Canada.

2.2 Guiding Principles

Soil is a complex heterogeneous medium consisting of variable amounts of minerals, organic matter, water and air, and is capable of supporting organisms, including plants, bacteria, fungi, protozoans, invertebrates and other animal life. Ideally, soil with contaminants present at the guideline levels will provide a healthy functioning ecosystem capable of sustaining the current and likely future uses of the site by ecological receptors and humans.

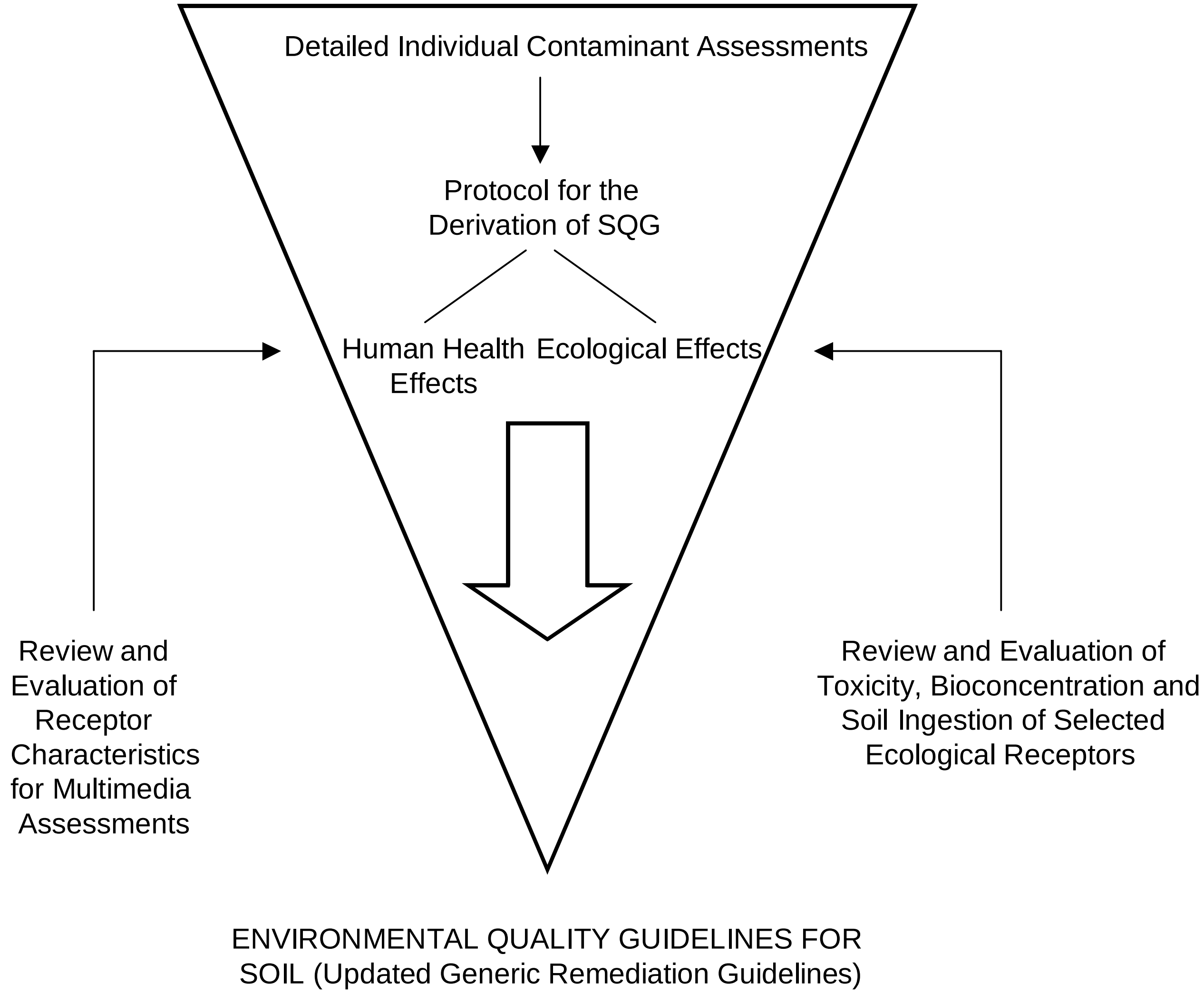


Figure 2: Contaminant Assessment Procedure for Deriving Soil Quality Guidelines

2.2.1 Protecting the Environment

To protect the terrestrial ecosystem, the protocol derivation process considers the adverse effects from direct contact exposure to soil-based contaminants as well as those resulting from ingestion of contaminated soil and food. Potential exposure pathways, receptor arrays, and exposure scenarios are assumed for major land uses. Based on these exposure scenarios, ecological receptors that sustain the primary activities for each land use category are identified.

A literature review is conducted to determine the environmental fate and behaviour of the contaminant as well as its toxicity in soil. A standard procedure is used to derive an effects-based soil quality guideline for soil-dependent organisms (i.e., invertebrates, plants and microbes) from acceptable toxicity data. For higher trophic level consumers (i.e., livestock, terrestrial wildlife and predators where applicable), pathways have been identified to derive environmental quality guidelines which consider the ingestion of contaminated soil and food, as well as the ingestion of contaminated water by livestock at agricultural sites. Wind erosion resulting in deposition on a more sensitive neighbouring property is also considered in the derivation of environmental quality guidelines.

The protocol also addresses the potential impacts on aquatic ecosystems from contaminants originating in soil that may enter the groundwater and subsequently discharge to a surface water body. This pathway may be applicable under any land use category, where a surface water body sustaining aquatic life is present (i.e., within 10 kilometres of the site). Where the distance to the nearest surface water body is greater than 10 kilometres, application of the pathway should be evaluated on a case-by-case basis by considering the site-specific conditions.

2.2.2 Protecting Human Health

Human health soil quality guidelines provide concentrations of contaminants in soil, at or below which no appreciable human health risk is expected. To protect human health, derivation processes for threshold and non-threshold toxicants are differentiated, taking into account daily background exposure from air, water, soil, food, and consumer products. Indirect exposure routes resulting from contaminated soils, such as contaminated groundwater, contaminated meat, milk and produce, contaminated produce from private gardens, infiltration into indoor air, and wind erosion resulting in deposition on neighbouring properties are also considered during the derivation of human health guidelines. These indirect exposure routes are evaluated conservatively by applying simplified transport and redistribution models using generic site characteristics in a variety of site conditions.

Key components of the risk based generic human health guidelines include a multimedia assessment of background exposure unrelated to contaminated sites and a generic human exposure scenario relevant to each land use. In the multimedia exposure assessment, total background exposure from all sources (i.e., air, water, food, soil, and consumer products when appropriate) and by all pathways (i.e., inhalation, ingestion, and skin absorption) is estimated.

The human health soil quality guidelines are established after accounting for this background exposure to ensure that the tolerable daily intake (TDI) of the contaminant is not exceeded.

2.3 Land Use

Generic soil quality guidelines are derived to protect human and key ecological receptors that sustain normal activities for four land use categories: agricultural, residential/parkland, commercial, and industrial. Generic land use scenarios are envisioned for each category based on how the land is used and on how sensitive and dependent the activity is on the land. Sensitivity to contamination increases among ecological or human health components most dependent on land use activities (i.e., agricultural and residential/parkland) (see Figure 3).

The definition of each land use accommodates generic conditions and puts boundaries on the receptors and exposure pathways considered in the guideline derivation for that land use. The four defined land uses are as follows:

Agricultural: where the primary land use is growing crops or tending livestock. This also includes agricultural lands that provide habitat for resident and transitory wildlife and native flora.

Residential/Parkland: where the primary activity is residential or recreational activity; parkland is defined as a buffer between areas of residency, and also includes campground areas, but excludes wildlands such as national or provincial parks.

Commercial: where the primary activity is commercial (e.g., shopping mall) and not residential or manufacturing. This does not include zones where food is grown.

Industrial: where the primary activity involves the production, manufacture, or construction of goods.

Key biological receptors and exposure pathways were identified for each land use to protect soil quality and maintain activities performed on these lands. Recognizing differences in analyzing human health and ecological issues, soil quality guidelines for each chemical are developed for both ecological and human receptors. For each of the four land uses, to protect both human health and the environment, the most protective guideline is chosen as the recommended soil quality guideline.

The defined exposure scenarios used to develop the soil quality guidelines do not cover the full spectrum of the types of sites, environments, and organism-site interactions that can exist (see Figure 3). For example, a natural areas land use has not been defined. The applicable exposure pathways for natural areas could vary considerably among different sites due to differences, for example, in: resident wildlife species; presence of campers, hikers, trappers, etc.; consumption of country foods by local residents. As a conservative approach, the soil quality guidelines for agricultural land use could be applied to natural areas; alternatively, site-specific exposure scenarios could be considered. Additional land uses and exposure scenarios may be developed

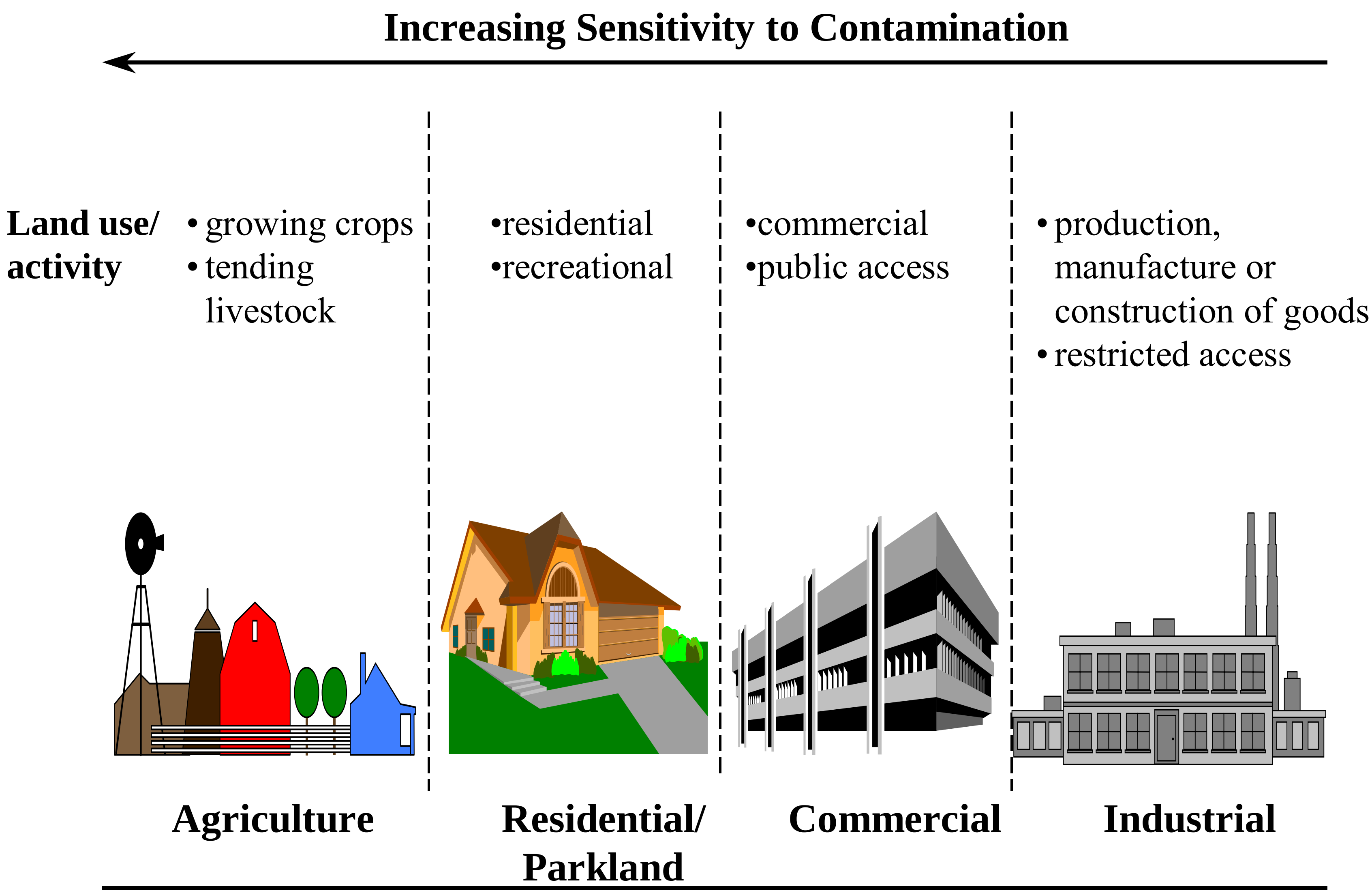


Figure 3: Concept of Generic Land Uses Envisioned for Guidelines Derivation

by jurisdictions to reflect situations that commonly occur in various regions; additional land uses may also be incorporated by CCME in future revisions of this Protocol.

SQGTG recognizes that there are areas where the science dealing with soil-based human and ecological toxicology and exposure is still developing. SQGTG therefore acknowledges the uncertainty (i.e., Part B, Section 6.0) in some of the data used in the protocol. Assumptions made in defining the characteristics of the generic land uses are therefore conservative in order to protect human health and the environment under a broad range of possible conditions.

2.4 Chemical Classification

Different types of contaminants exhibit different fate and transport characteristics in the environment. As a result, not all contaminants can be treated in exactly the same manner when developing soil quality guidelines. In particular, although a number of potential exposure pathways are common to most or all types of contaminants, some pathways may not be relevant for specific classes of chemicals.

For the purpose of guideline development, it is recommended that contaminants be classified according to the following categories:

Organic or Inorganic
Compounds which contain carbon atoms (and usually hydrogen atoms) are classified as organic compounds; all others are classified as inorganic. In generic guideline development, groundwater protection pathways are normally only considered for organic compounds, due to the highly site-specific nature of partitioning for inorganic compounds (see Appendix A).

Dissociating or Non-Dissociating
For purposes of guideline development, dissociating chemicals are considered to be those that form cations and anions in solution, including organic acids. Partitioning relationships for dissociating compounds (see Appendix A) need to consider both the ionized form and the non-ionized form (if applicable).

Volatile or Non-Volatile
Volatile chemicals are those which may be found in the vapour phase in significant quantities; this classification determines whether effects on indoor air quality due to vapour migration into buildings needs to be evaluated for a contaminant. If there is doubt as to whether a contaminant should be classified as volatile, a simple check can be made by comparing the product of the pure-phase solubility and the unitless Henry's Law constant (i.e. the theoretical vapour-phase concentration at saturation) with the published or derived reference concentration or risk-specific concentration (see Part C, Section 2). If the vapour-phase concentration cannot exceed the toxicity benchmark, then there is no need to evaluate the vapour migration pathway for the protection of indoor air quality and the contaminant can be treated as non-volatile for purposes of guideline development.

Soluble or Non-Soluble
For purposes of soil quality guideline development, a contaminant is considered to be soluble if it may be present in water at a concentration high enough to pose a human health or environmental risk. As a general rule, any substance for which a published water quality guideline is available in Canada can be considered to be soluble. For substances with no published water quality guideline, if the pure-phase solubility is equal to or greater than a derived concentration for the protection of potable groundwater (Part C, Section 5.3.2) or livestock watering (Part B, Section 7.8), then the contaminant is considered to be soluble. Groundwater protection pathways are only evaluated for soluble contaminants.

Biomagnifying or Non-Biomagnifying
Biomagnifying contaminants are those which may increase in concentration as they move through the food chain. Ingestion by secondary and tertiary consumers must be evaluated for these contaminants during guideline development. The Environment Canada Toxic Substances Management Policy (TSMP) indicates that any compounds with a reported bioaccumulation factor or bioconcentration factor greater than 5000, or an octanol-water partitioning coefficient (K_{ow}) greater than 10^5 should be treated as biomagnifying. Recent research in Northern Canada (Kelly & Gobas, 2001) has indicated that some substances which do not meet these criteria may still biomagnify; therefore, if the literature review indicates that a substance biomagnifies, it should be treated as biomagnifying regardless of whether or not it meets the TSMP criteria. It should also be noted that some substances with a K_{ow} greater than 10^5 do not biomagnify. If studies on a substance with a high K_{ow} demonstrate a lack of biomagnification in upper trophic levels, then guidelines for ingestion by secondary and tertiary consumers may not be needed for this substance.

Guidance on the exposure pathways that should be considered for the different classes of chemicals are given in later sections of the protocol.

Every chemical has peculiarities that cannot be adequately documented in this protocol. These peculiarities will be identified and discussed in individual supporting documents. There may also be some contaminants where the guideline derivation process described in the protocol is not suitable. Deviations from the protocol will be fully documented. Each responsible agency will have to decide how and when to best incorporate new data, information and approaches into generic guidelines.

While it is recognized that contaminants are likely to occur in mixtures, not enough is known about contaminant mixtures for them to be considered explicitly in the protocol. Some substances, such as petroleum hydrocarbons, comprise a large number of compounds having varying physical and chemical properties and toxicity, and are therefore more appropriately addressed as mixtures. Other compounds may normally coexist with different congeners having similar physical and chemical properties but significantly different toxicity. The procedures outlined in this protocol are intended primarily for individual compounds, although they can be used with surrogates or indicator compounds representing mixtures. Certain classes of chemicals have been addressed in this manner, although the methods of applying the resultant guidelines to mixtures vary according to the substances being considered. Reference is made to some of these methods in later sections of the protocol.

2.5 Soil Type and Depth

2.5.1 Soil type

The protocol recognizes that contaminant fate and transport, as well as bioavailability, are dependent to varying degrees on soil texture, moisture content and other factors. To minimize the uncertainty in guideline derivation introduced by soil variability, the protocol considers two generic soil types: coarse-textured soils (soils containing predominantly sand and gravel sizes) and fine-textured soils (soils containing predominantly silt and clay sizes). The criterion distinguishing the two categories is a median grain size of 75 microns. Generic soil properties representative of typical soils in each category have been assigned for the purposes of guideline development; these are summarized in Appendix I.

The influence of soil texture on physical transport processes in soil is reasonable well understood and can be quantified, at least on a generic basis, using contaminant-specific information and the generic soil properties provided. The effect of soil texture on bioavailability and toxicity to soil-dependent organisms is less well known, and the extent of scientific data is likely to be limited. Therefore, insufficient data may be available for all contaminants to develop soil quality guidelines for both coarse-textured and fine-textured soils.

It should also be noted that an individual jurisdiction may choose to take soil type into account only on a site-specific basis.

2.5.2 Soil depth

The protocol does not specify the depth to which the generic soil guidelines apply. Most direct human and ecological exposure pathways apply to soil located at or near surface. Soils at depth are less accessible for human contact and are typically not required to perform the same level of ecological function, although such soils may still be sources of indirect exposure through vapour and groundwater pathways. Surface soils are often defined as those within the uppermost 1.5 m of the soil profile.

An administrative problem may arise if certain pathways are modified or excluded, based on depth, for the development of generic guidelines; specifically, soil disturbance can result in subsurface soils being relocated at or near the ground surface. For this reason, the protocol does not explicitly address the determination of generic subsurface soil quality guidelines. However, guidance for determining subsoil guidelines is provided, since subsoil guidelines may be developed and implemented by an individual jurisdiction or on a site-specific basis in conjunction with a suitable risk management policy.

2.6 Summary of Guideline Development Process

The guideline development process is detailed in Parts B through D of this document. A brief summary of the process is presented below.

Separate soil quality guidelines are developed for the protection of environmental and human health. Guidelines are developed based on four defined land use scenarios, though other land use scenarios may be defined by jurisdictions or on a site-specific basis.

The environmental soil quality guideline (SQG_E) is determined by evaluating direct soil contact for plants and soil invertebrates, nutrient and energy cycling processes, ingestion of contaminated food and soil by wildlife, and the transport of contaminants through groundwater to potential livestock watering sources and surface water bodies inhabited by freshwater life. The lowest of the soil concentrations deemed protective of each of these pathway-receptor combinations becomes the SQG_E. The level of protection required for each pathway is dependent on the land use scenario; some of the receptor-pathway combinations are not evaluated for all land uses or contaminant types.

Similarly, the human health soil quality guideline (SQG_{HH}) is determined by evaluating direct soil exposure (soil ingestion, dermal contact, and particulate inhalation), transport of contaminants through groundwater to potential potable water sources, intrusion of contaminant vapours into buildings, and human consumption of contaminated food. The lowest of the soil concentrations deemed protective of each of these potential exposure pathways becomes the SQG_{HH}. The specific exposure scenario is dependent on the land use; some of the exposure pathways are not evaluated for all land uses or contaminant types.

The lowest of the SQG_E and SQG_{HH} becomes the final soil quality guideline (SQG_F) for each land use scenario. The SQG_F is also checked against non-toxicity considerations and typical background soil concentrations.

SECTION 3
USE OF CANADIAN SOIL QUALITY GUIDELINES

The soil quality guidelines derived using the protocol replace the Interim Environmental Quality Criteria for Contaminated Sites (CCME, 1991a), where applicable. Soil quality guidelines represent "clean down to levels" at contaminated sites and not "pollute up to levels" for less contaminated sites. They are not intended to be used to manage pristine sites.

The Canadian Soil Quality Guidelines are intended to be used for assessing in-place contaminants in soil. They are not intended for evaluating the quality of soil amendments (e.g., compost, synthetic fertilizers, manures, etc.) and are not directly comparable to quality criteria for these types of materials. It is also not recommended that the soil quality guidelines be used for waste management of fill materials (e.g., slags, foundry sands, mining wastes, etc.). Use of the soil quality guidelines for anything other than their intended purpose should only be done with great care and an understanding of the guideline development process and its relevance to the proposed use. Jurisdictions should consider the relationship of these guidelines to soil concentrations that may result from long-term applications of materials to the soil. The soil quality guidelines must also not be considered as permission to contaminate up to a certain level.

The guidelines should be used in combination with acceptable sampling and analytical methods. Guidance documents on sampling methods and site characterization have been published by CCME (1993) and ASTM (2002), as well as by numerous jurisdictions. Typical analytical methods are summarized in the scientific supporting documents prepared for each guideline.

The development of ecological effects-based soil quality guidelines is, in a sense, a scaled down risk assessment for generic conditions and therefore the following uncertainties apply.

3.1 Primary Error in Model Input Parameters

Model error created from the inappropriate aggregation of variables (i.e., multiple species toxicity data and endpoints) used to determine acceptable threshold effect concentrations (TECs), and the error associated with the input variables (individual toxicity data) themselves must be considered in guidelines derivation.

An examination of the toxicological data reported for soil-dwelling organisms and terrestrial animals revealed that common reference toxicity values (i.e., LOEC, NOEC, LC_{50}, EC_{50}, EC_{25}) were available for guidelines derivation. While it is possible to quantify the error associated with predictions of LC_{50} and EC_{50} data using the confidence intervals reported, it is difficult to estimate the uncertainty associated with improper use of a statistical model (e.g., probit or logit) applied to the test data (e.g., when data display hormesis).

Perhaps the most significant source of uncertainty in soil toxicity data available for guidelines derivation is attributed to LOEC and NOEC data. No observable effects concentration and LOEC data are hypothesis driven and are thus subject to Type I and Type II error as well as variation of the test design itself. Again, these data have been generated from the improper use of statistical models (usually by ANOVA, paired means comparisons, etc.) A sizable proportion of the soil toxicity data available for guidelines derivation are LOEC and NOEC, and because these data are still considered useful in the absence of more meaningful EC_X data, a discussion on the reasons for their uncertainty is warranted.

Traditionally, LOEC and NOEC data were estimated without considering the dose-response curve (i.e., for LOECs, by using the lowest test concentration that is significantly different from the controls; or for NOECs, the highest test concentration not significantly different from controls). Some researchers view this as problematic (Bruce and Versteeg, 1992) and suggest that LOEC or NOEC concentrations suffer from the fact that:

- They must be one of the test concentrations used in the study and are therefore dependent on the range of concentrations used, the sensitivity of the test controls, replicate number and replicate variability.

- High test variability can lead to inaccurate estimation of these concentrations and loss of information on the dose-response of the chemical to the test organism and variability of the data set.

Therefore, NOEC and LOEC data can vary significantly from study to study given the same test conditions and may not reflect the "true" concentration for these endpoints. Alternative methods to hypothesis-based NOECs and LOECs have been proposed (e.g., Mayer, 1991; Bruce and Versteeg, 1992; Hoekstra and van Ewijk, 1993). The essence of these methods is low-dose/concentration interpolation based on calculated doses/concentrations on the response curve. Therefore, for reasons above, LOEC or EC_X data interpolated or extrapolated from the dose-response curve are preferred, but since most older soil toxicity data are not calculated using this method, arbitrary uncertainty factors accommodate for the lack of confidence with using these data in guidelines derivation.

3.2 Model Uncertainty

Due to the restrictions on model input parameters, the error associated with the guidelines derivation can only be qualitatively assessed. The primary concern here is the ability of the statistical model employed to accurately predict the concentration at which no effects or low-level effects are observed. This of course is subject to the overall error inherent in toxicity data used to derive the guideline, but also to the error in model design. For instance, if a large proportion of NOEC and LOEC data are used for derivation, the degree to which these data affect output is a function of the statistical design used in the model.

Another source of model uncertainty has been termed "stochasticity of species sensitivity" (Suter, 1993). That is, species sensitivity is assumed to be a stochastic variable and can be characterized by fitting an empirical or probabilistic distribution of test endpoints for several species to determine guidelines. There is also uncertainty about the true distribution of species sensitivity. The Netherlands (e.g., Kooijman, 1987; van Straalen and Denneman, 1989; Aldenberg and Slob, 1991) and Denmark (Wagner and Lokke, 1991) have proposed statistical methods in an attempt to quantify the uncertainty of the "true" stochastic multi-species sensitivity distribution. Recently, species sensitivity distributions have been used in the development of soil guidelines in some countries (e.g. Suter et al., 2002).

Depending on the method employed for deriving environmental guidelines, test endpoints may be used that are not considered concentrations that pose low-level effects to a species (i.e., LC_{50}, EC_{50}). The goal of guidelines derivation is to estimate a concentration at which no significant adverse effects are observed in field populations (agricultural and residential/parkland) or significant low-level effects in field populations (commercial and industrial land). Therefore, uncertainty arises from the need to extrapolate from median lethal or effective concentrations to areas of the species sensitivity distribution required for guidelines derivation. Uncertainty factors are generally used for this extrapolation.

Use of statistically meaningful low-level effects data (e.g. EC_{25} values) reduces the uncertainty in guideline derivation, when compared to the use of NOEC/LOEC data or LC_{50}/EC_{50} data with uncertainty factors.

PART B

SECTION 1
DERIVATION OF ENVIRONMENTAL SOIL QUALITY GUIDELINES

It is important to understand what is meant by ecological effects when deriving environmental soil quality guidelines. A wide range of effects can be considered when examining various components of a terrestrial ecosystem. These include both abiotic and biotic factors which influence ecosystem structure and function. Assessing the magnitude of potential effects of all of these factors usually requires an ecological risk assessment. This protocol focuses on the effects of chemical stressors on the biotic component of a terrestrial ecosystem; specifically, the potential for adverse effects to occur from exposures to soil-based contaminants at point-of-contact or by indirect means (e.g., food chain transfer or transport to nearby surface water). Adverse effects data may come in a variety of forms, ranging from data collected in the field (e.g., mesocosm studies) to single species tests performed in the laboratory (i.e., using bioassays). Specific land uses are studied and guidelines based on the availability of terrestrial toxicity information are developed in this protocol.

The following sections examine topics related to the "initial staging" of guidelines development and provide a detailed description of the guidelines derivation process. These topics include narrative and illustrative descriptions of the level of ecological protection envisioned for guidelines, relevant endpoints, availability of soil toxicity data for guidelines derivation, and the formulation of exposure scenarios for contaminated soil. Potential exposure pathways, receptor arrays, and exposure scenarios for individual land use categories are also examined. Based on information from the initial staging, the process for deriving soil quality guidelines is then presented.

SECTION 2
LEVEL OF ECOLOGICAL PROTECTION AND RELEVANT ENDPOINTS FOR DERIVING SOIL QUALITY GUIDELINES

2.1 Level of Ecological Protection

Before soil quality guidelines can be derived, an understanding of the level of ecological protection must be established if protection goals for the environment are to be effective and sustainable. The level of protection provided by the guidelines depends on the protection goals sought for individual land use categories. Therefore, for agricultural and residential/parkland land use, it is necessary to achieve a level of ecological functioning that sustains the primary activities associated with these land uses. To make this possible, soil quality guidelines for these land uses are derived using laboratory and field toxicological data that make predictions on the adverse effects (effects that undermine a species' ability to survive and reproduce under normal living conditions) of chemicals on key ecological receptors. The protection goals and endpoints for agricultural and residential/parkland land use are described in Sections 5.1 and 5.2.

On commercial and industrial lands, the primary land use activities are not directly dependent on the need to sustain a high level of ecological processes (see Figure 3). The same key ecological receptors and endpoints examined for agricultural and residential/parkland land uses are also examined for commercial and industrial land use. However, SQGTG has decided that the level of protection for commercial and industrial land use does not need to be as stringent as for agricultural or residential/parkland land uses. Accordingly, the degree to which commercial and industrial receptors may experience adverse effects is increased to correspond with the lower protection levels required for the land use category. For more information on the key receptors and level of protection sought for commercial and industrial land use see Section 5.3 and 5.4.

Despite the different levels of protection sought for individual land uses, an important common principle exists for all land use categories. For each land use, the level of ecological protection provided by the soil quality guidelines ensures that the remediated land has the potential to support most activities likely to be associated with that land use (see Figure 3).

2.2 Relevant Effects Endpoints

The term "endpoint" is often used in chemical effects assessment to define the single response factor that characterizes the impact of the test chemical on a selected organism(s) (SECOFASE, 1993). In developing generic environmental soil quality guidelines, only the endpoints related to the "direct effects" of chemical stressors to receptors can be examined, and these do not account for the "indirect effects" (e.g., avoidance of polluted food items) that may occur from sublethal exposures. Consequently, soil quality guidelines derived using endpoints from data on direct effects, may mask other less observable responses that have a cumulative negative effect on organism survival. While this is an area for further research in terrestrial effects assessment, it is possible to examine these interactions at a site-specific level.

2.3 Endpoint Definition

Suter (1993) describes two basic types of endpoints used in ecological risk assessment: assessment and measurement endpoints. Assessment endpoints are formal expressions of the environmental values to be protected (Suter, 1989) (e.g., decline in soil arthropod abundance). Two steps are required to properly define these endpoints in a research project:

- identifying the valued attributes of the environment considered to be at risk, and
- defining these attributes in operational terms (Suter, 1993).

Suter (1993) indicated that there is no universal set of assessment endpoints, but that there are five criteria that any endpoint should satisfy:

- societal relevance,
- biological relevance,
- unambiguous operational definition,
- accessibility to prediction and measurement, and
- susceptibility to the hazardous agent.

Examples of assessment endpoints are given in Suter (1993), who recommends that the aforementioned criteria be seriously considered when operationally defining assessment endpoints so that ecological effects assessment is effective and understood at the societal level.

Measurement endpoints are measurable, quantifiable responses (e.g., EC_{50}) to a chemical stressor that are related to a valued attribute of the ecological component (Suter, 1990). These measurable responses are usually estimated in monitoring studies of laboratory toxicity tests and are generally referred to as indicators (Suter, 1993). Since it is often difficult to measure assessment endpoints most data are used quantitatively or qualitatively. Assessment and measurement endpoints are seldom the same since assessment endpoints are usually defined on a large scale (e.g., populations, ecosystems) and measurement endpoints on an individual level. Nevertheless, measurement endpoints should be consistent with assessment endpoints (e.g., predictions on population decline based on mortality estimates). Examples of measurement endpoints related to assessment endpoints are given in Suter (1993).

In terrestrial toxicity testing, most data are focused on mortality (LC_{50}) as a short-term endpoint and reproduction, growth, development, behaviour, activity, lesions, physiological changes, respiration, nutrient cycling, contribution to decomposition, genetic adaptation, and physiological acclimatization as long-term, sublethal endpoints (EC_X, NOEC, LOEC) (SECOFASE, 1993). Mortality can be recognized as the ultimate measurement endpoint in ecotoxicological testing and is most often used in soil invertebrate and terrestrial avian and mammalian testing. Among sublethal effects, reproduction and growth are common endpoints for soil invertebrates and plants, and to a lesser degree with avian and mammalian species.

2.4 Selection of Ecologically Relevant Endpoints for Guidelines Derivation

It is generally accepted that ecotoxicology attempts to prevent local species extinction (SERAS, 1992). This concept of ecotoxicology should be maintained during the development of "safe" concentrations in the environment for regulatory purposes. In the development of soil quality guidelines, assessment endpoints at the community or ecosystem level, such as structure and function, ideally would provide the best measure of ecological impact. However, as discussed and agreed upon at the OECD workshop for ecological effects assessment (OECD, 1988), studies at this level are difficult and expensive to perform. In addition, physicochemical and biological conditions of terrestrial systems are highly variable in space and time making results of these studies difficult to interpret and limiting their validity for other areas (van Straalen and van Gestel, 1992; Pederson and Samsoe-Petersen, 1993). Currently, it is not practical to establish generic soil quality guidelines using endpoints from this level of biological organization. Further verification of chemical effects on terrestrial ecosystems is needed. Therefore, for the purposes of generic guidelines derivation assessment, endpoints must be defined by directly extrapolating measurement endpoints to field populations. The Threshold Effects Concentration (TEC) (soil-dependent biota) or Daily Threshold Effects Dose (terrestrial animals), provides the measurement endpoint data, that if exceeded is expected to result in adverse effects on populations in the field. Therefore, in this protocol, assessment endpoints can be regarded as the biological impairment of a species' ability to survive or reproduce.

For generic guidelines derivation, the protection levels sought in Section 2.1 are determined from information from laboratory tests, and a suitable extrapolation method. To this end, environmental soil quality guidelines employ sensitive measurement endpoint data from key receptors that act as "predictive sentinel species". Extrapolation to assessment endpoints is therefore restricted to the population level since single species measurement endpoint data are used in guidelines derivation. Information from laboratory studies should therefore involve endpoints critical to the maintenance of a species. Specifically, these endpoints include those that are critical for a species to complete a normal lifecycle, and to produce viable offspring. Traditionally, in soil ecotoxicology, these endpoints have been limited to mortality, reproduction, and growth.

2.4.1 Short- and Long-term Tests for Soil-dependent Organisms

Current definitions of short- and long-term exposure and endpoints for soil toxicity testing are either lacking or vary from agency to agency. Whereas numerous short-term tests are available for earthworms and plants, few long-term tests were historically available for these or other soil-dependent organisms or microbial processes (Environment Canada, 1994). In recent years, however, an increasing number of standardized methods and protocols (e.g. Environment Canada, 2004a, 2004b, 2004c) have been developed for short- and long-term toxicity tests.

There is consistent use and acceptance of the 7 and 14 day mortality test as a short-term test for earthworms (e.g., OECD, 1984; Greene et al. 1989; ISO, 1991) and for the five-day seed germination/root elongation test as a short-term test for plants (e.g., U.S. EPA, 1982; Porcella, 1983; Ratsch and Johndro, 1986; Thomas and Cline, 1985; Miller et al., 1985; Wang, 1987; Wang and Williams, 1988; ASTM, 1990a, 1990b). However, these short-term tests are now considered to be outdated and insensitive.

Long-term standardized toxicity test methods for earthworms and isopods have recently been developed (Environment Canada, 2004a; Environment Canada 2004c; ISO, 1999; ISO, 1998; ISO, 1992; OECD, 1993; NISRP, 1991). Long-term plant toxicity tests have also come into use in recent years, based on endpoints including plant growth and life cycle flowering (Environment Canada, 2004b; ASTM, 1996; ASTM, 1991). Usually, long-term exposure tests for soil-dwelling organisms include at least one reproduction stage (in the case of soil invertebrates), or one growth cycle (in the case of plants).

The acceptability of short- and long-term tests for general use in this protocol should be determined on a study-by-study basis using the information cited above as a guide. Data from long-term studies are preferred for deriving soil quality guidelines. However, given the limited availability of such data, short-term toxicity data may be accepted for guidelines derivation.

Studies showing damage or visible injury to ornamental plants (including trees), where they exist, should also be included for the derivation of soil quality guidelines if the experimental protocols are sufficiently rigorous, since healthy appearance of plants is of importance to many property owners.

2.4.2 Short- and Long-term Tests for Mammalian and Avian Species

A considerable number of standardized toxicological tests have been carried out on traditional laboratory animals using different endpoints for various types of exposure. One of the most common toxicity tests is the LD_{50} (i.e., the lethal dose which results in 50% mortality of the test population). This test is normally an acute exposure often involving a single administration of various chemical doses to laboratory animals followed by a 7 to 14 day examination of lethal effects (Klassen, 1986). Unfortunately, standardized methods for conducting toxicity tests for wildlife and livestock species are generally lacking, with the exception of the five-day dietary LC_{50} test for avian species (Hill and Hoffman, 1984).

Historically, in mammalian and avian subacute and chronic lethality tests, the number of dose regimes often does not provide sufficient information to calculate a dose that results in a 25% or 50% response. Thus, endpoints for chronic exposures are often reported as the no observed (adverse) effect level (NO(A)EL) and the lowest observed (adverse) effect level (LO(A)EL). Chronic adverse effects endpoints can be related to reproduction, growth, or viability resulting from a continuous exposure over a significant portion of the organism's lifespan. Subchronic exposures normally involve the same endpoint as chronic exposures, but are conducted over a shorter time period.

SECTION 3
STATUS OF THE TOXICOLOGICAL DATABASE FOR SOIL RELATED EXPOSURES

3.1 Soil-dependent Organisms

Most available toxicological information for soil-based exposures was generated using soil-dependent biota. A compilation of toxicological data for soil-dependent organisms (plants, invertebrates and microbes) by Dennemen and van Gestel (1990) indicates that well-characterized soil toxicity information is lacking for many contaminants. However, there is considerable research effort going into establishing standardized soil toxicity test procedures to generate new toxicity data for a broader range of soil-dependent organisms (e.g., soil isopods) (NISRP, 1991).

Currently, soil toxicity data for soil-dependent organisms are better characterized for inorganic rather than organic contaminants (Dennemen and van Gestel, 1990). Long-term studies, for the most part, have historically reported the no observed effect concentration (NOEC) and the lowest observed effect concentration (LOEC), though more recent studies may report effective concentrations (EC_X) or inhibition concentrations (IC_X). Short-term studies typically report a median lethal (LC_{50}), median effective (EC_{50}) concentration, or a NOEC and LOEC (estimated from the highest test concentration producing no observed effects and the lowest test concentration producing an observed effect, respectively).

3.2 Mammalian and Avian Species

The bulk of available mammalian toxicological data for environmental contaminants was generated using laboratory animals, particularly rodents. Significantly less information is available for terrestrial wildlife and livestock species for soil-based exposures, and most of these studies were performed using oral dosages via food. Few toxicity studies have been conducted on avian wildlife, and most were performed on poultry and game birds (Walker and MacDonald, 1992). Little or no information exists on dermal contact toxicity from contaminated soil exposures.

Soil ingested directly or adhered to vegetation can account for most, if not all, of the contaminant ingested by an animal (Beresford and Howard, 1991). Soil ingestion was also found to be a more significant pathway of exposure in animals than ingestion of contaminated forage (Zach and Mayoh, 1984). Unfortunately, relatively little data exist on the effects of contaminated soil ingestion by birds and mammals.

SECTION 4
POTENTIAL ECOLOGICAL RECEPTORS AND EXPOSURE PATHWAYS OF SOIL CONTAMINATION

4.1 Ecological Receptors of Soil Contamination

As a first step in the development of environmental soil quality guidelines, it is essential to identify the ecological component(s) potentially at threat from contamination. This protocol addresses adverse effects posed by chemical stressors to the biotic component (receptors) of a terrestrial ecosystem. By characterizing potential receptors of contaminated soil it is possible to identify those requiring protection from soil contamination as well as evaluating potential exposure scenarios. A simplified exposure scenario identifying potential receptors of contaminated soil in a greatly simplified terrestrial ecosystem is illustrated in Figure 4. It is evident that exposure spans a range of trophic levels including soil-dependent organisms (plants, soil invertebrates, soil microbes) and higher order consumers (wildlife, livestock).

Ideally, the selection of receptors should be compatible with, and reflect important characteristics of the ecosystem (i.e., ecologically relevant). Because of the paucity of ecological effects information for terrestrial organisms, however, the selection of ecological receptors for this protocol must focus on key receptors that maintain land use activities. It is also necessary to devise an array that includes ecologically relevant receptors that are sensitive to chemical stressors, so that guidelines derived through this protocol reflect sensitive measurement endpoints. In this sense, the receptors of choice serve as predictive sentinel species from which guidelines can be derived. The ecological receptors selected for this protocol are summarized in Table 1 and discussed in more detail for each land use in Sections 5.1 to 5.5.

4.2 Exposure Pathways to Soil Contamination

Once the receptor array has been identified, soil exposures for these receptors must be evaluated. For the purposes here, evaluation depends on the production of a set of facts, assumptions, and inferences about how exposure to ecological receptors from contaminated soil takes place. In the scenario depicted in Figure 4, soil-dependent organisms and their consumers are exposed to contamination directly from the soil, water, and air and indirectly through the food chain. Those organisms dependent on soil for survival (e.g., plants, soil invertebrates, soil microbes) come into contact with the soil directly as a function of their life cycle and therefore will likely receive the greatest threat from contaminated soil. Higher trophic level consumers receive their largest exposure through consumption of contaminated food and ingested soil particles, although absorption of contaminants from dermal contact with soil and inhalation of vapours and suspended soil may also occur.

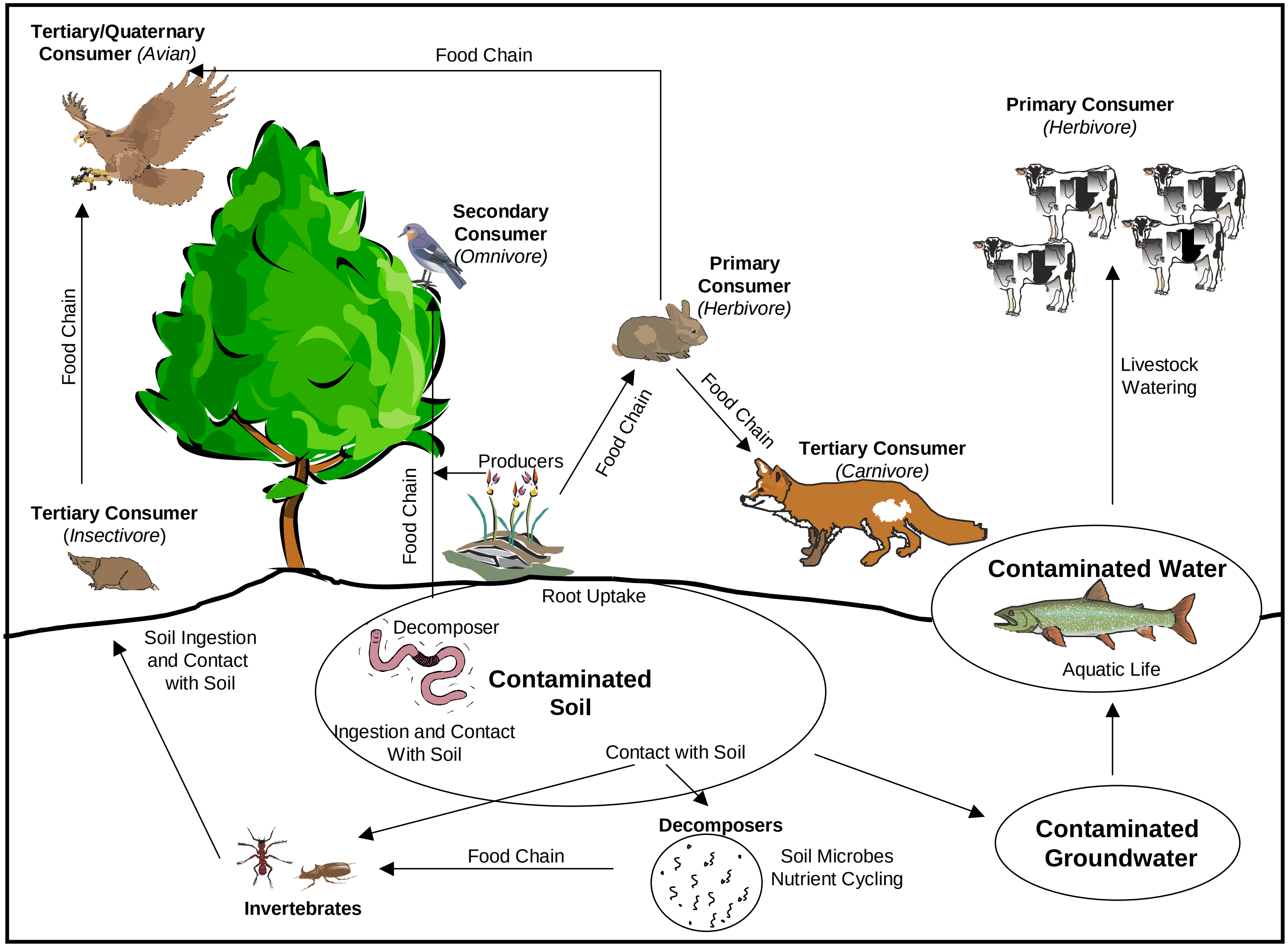


Figure 4: Simplified Diagram of Potential Ecological Receptors and Exposure Pathways of Contaminated Soil

Ideally, it would be desirable to consider all influential variables for all potential exposure pathways at all trophic levels (as might be done at the site level). This is not possible in a generic protocol and therefore the exposure pathways must include those expected for the receptors selected for each land use. The exposure pathways considered in this protocol are summarized in Table 1, and discussed in more detail for each land use in Sections 5.1 to 5.5. It should be noted that the absence of wildlife considerations for ingestion exposures under residential/parkland, commercial, and industrial land use reflects the state of information availability and quality in these areas. If acceptable and relevant toxicological and exposure information is available to allow further examination of these scenarios, then guidelines may be derived for these categories. Although only ingestion by livestock and herbivorous wildlife is normally considered for agricultural land use, ingestion by secondary and tertiary consumers should be evaluated for substances that are known to bioaccumulate or biomagnify, provided that sufficient data are available. Ingestion by primary, secondary and tertiary receptors should also be considered for residential/parkland land use for these substances, again subject to availability of adequate data.

Table 1. Receptors and Exposure Pathways for the Land Use Categories Considered in the Derivation of Environmental Soil Quality Guidelines

Route of Exposure	Agricultural	Residential/ Parkland	Commercial	Industrial
Soil Contact	Soil Nutrient Cycling Processes, Soil Invertebrates, Crops/Plants, Livestock/Wildlife	Soil Nutrient Cycling Processes, Soil Invertebrates, Plants, Wildlife	Soil Nutrient Cycling Processes, Soil Invertebrates, Plants, Wildlife	Soil Nutrient Cycling Processes, Soil Invertebrates, Plants, Wildlife
Soil and Food Ingestion	Herbivores, Secondary and Tertiary Consumers[a]	Herbivores[a], Secondary and Tertiary Consumers[a]		
Ingestion of Contaminated Water	Livestock			
Contact with Contaminated Water	Freshwater Life, Crops (irrigation)	Freshwater Life	Freshwater Life	Freshwater Life

Note: a – Herbivores (residential/parkland) and Secondary and Tertiary Consumers (agricultural and residential/parkland) are considered for substances that bioaccumulate and/or biomagnify

SECTION 5
EXPOSURE PATHWAYS AND KEY RECEPTORS ACCORDING TO LAND USE AND DATA AVAILABILITY

Land use, by definition, implies a human measure of the requirement for or capability of a portion of the earth's surface to sustain some human activity, not necessarily an ecological one. However, the maintenance of primary ecological functions is usually required for most land use activities (except some commercial and industrial processes). The decision to derive environmental soil quality guidelines according to selected human land uses in no way subordinates the protection of ecological values to human values. The following sections describe the receptor and exposure scenarios for agricultural, residential/parkland, commercial, and industrial land use.

5.1 Agricultural Land Use

In general, the primary activities associated with agricultural land use include the ability to grow crops and raise livestock. Although agricultural land use varies, the development of soil quality guidelines must protect key receptors that permit or maintain crop growth and livestock production against adverse effects. (Note: guidelines selection may include organisms that are subject to pesticide control.) Protection must also be offered to resident and transitory wildlife and native flora because, in some areas (e.g., agroecosystems), this may be the only viable habitat for these organisms. A generic exposure scenario considered for the derivation of soil quality guidelines for agricultural land use is provided in Figure 5.

5.1.1 Growth of Crops and Plants

To ensure crop production at agricultural sites, it is essential to maintain soil-dependent biota whose ecological function sustains crop and plant growth. Contact of crops and native plants directly with contaminated soil must also be considered in guidelines development (Figure 5).

Sufficient toxicological information exists to consider dermal soil contact by microbes (and their effect on nutrient cycling), soil invertebrates (e.g., decomposers), and crops and plants (e.g., seeds and roots) for guidelines derivation for the protection of crop and plant growth. Currently, there is not enough information to incorporate dermal absorption and translocation of contaminants by crops and plants via aerial deposition into guidelines derivation, but this pathway should be examined where information is available. Root uptake and accumulation of contaminants by crops grown on-site and used as feed, or native flora used as pasture, must also be examined when they relate to livestock and wildlife ingestion scenarios (Figure 5).

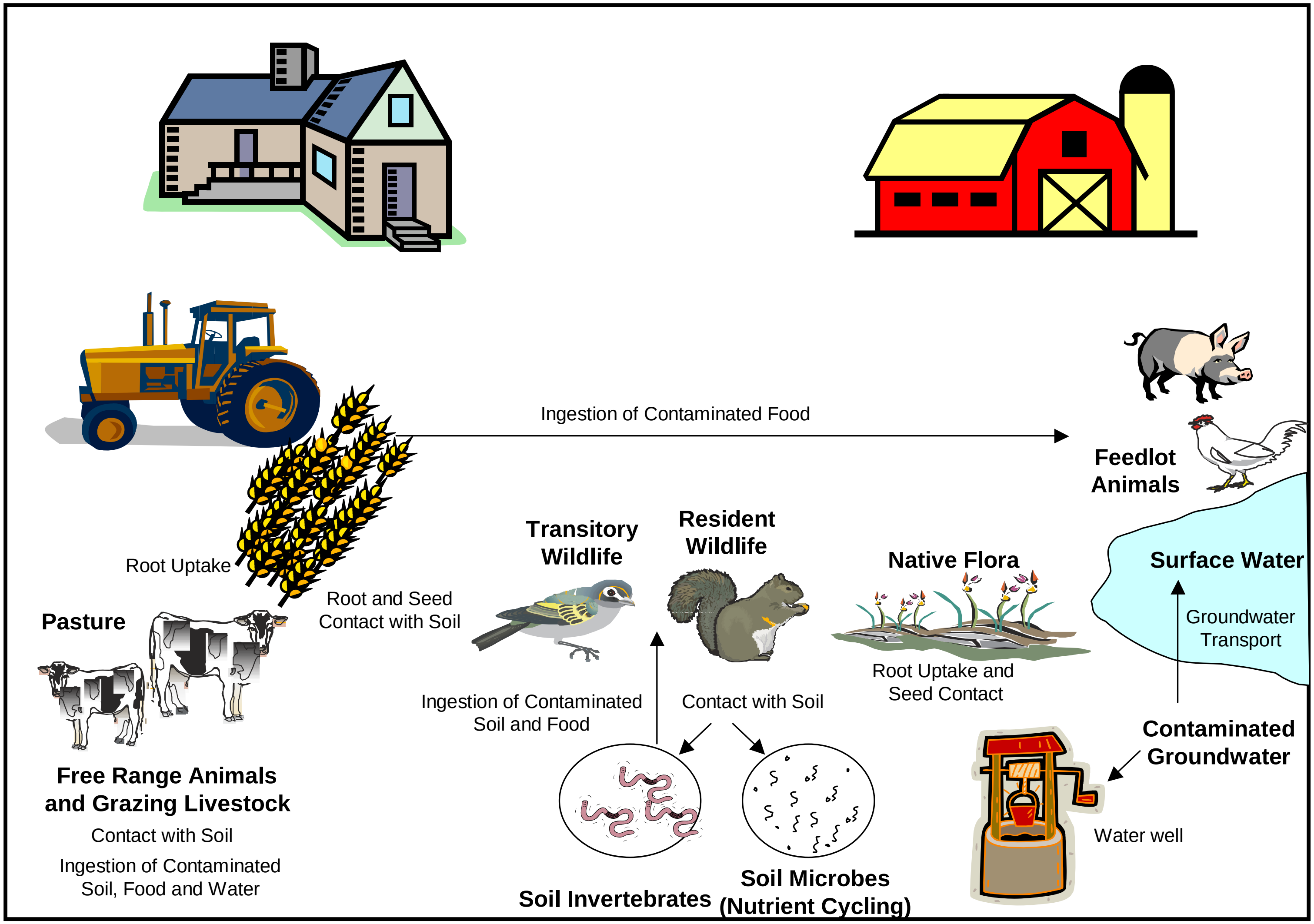


Figure 5: Key Receptors and Exposure Pathways of Contaminated Soil Considered for Agricultural Land Use

5.1.2 Maintenance of Livestock and Wildlife

To ensure adequate soil quality for the maintenance of grazing and feedlot livestock, potential adverse effects from the incidental ingestion of contaminated soil and ingestion of contaminated food must be considered for guidelines derivation. These pathways of exposure are the most significant source of contaminant uptake by livestock (Thorton and Abrahams, 1983, Fries, 1987, Paustenbach, 1989). Sufficient toxicological information exists to consider grazing herbivore soil and food ingestion exposures on agricultural lands for guidelines derivation. These pathways also pertain to herbivorous wildlife that may graze on agricultural lands (Beyer et al., 1994).

For substances that are persistent and have a strong tendency to bioaccumulate and/or biomagnify in the food chain, the consideration of herbivore exposures does not provide sufficient protection for all ecological receptors. Therefore, for agricultural land use, the derivation of soil quality guidelines for these substances should also consider the protection of secondary and tertiary consumers from ingestion of contaminated soil and food.

Dermal contact by livestock and wildlife (resident and transitory) with contaminated soil may pose a significant health risk to these organisms. In spite of this, information on the effects from dermal contact with contaminated soil by livestock and wildlife is severely lacking (OECD, 1988). Because of these data limitations, it is assumed that the level of protection offered to soil-dependent organisms from dermal exposure is adequate to protect livestock and wildlife from the same exposure. This assumption is based on the notion that soil-dependent organisms are more directly in contact with the medium for a large portion of their life cycle, and will experience adverse effects sooner than most organisms at higher trophic levels. This assumption will be held except where explicit information to the contrary exists. However, effects from dermal contact should be considered for guidelines derivation when information is available.

5.2 Residential/Parkland Land Use

The combination of different activities under one land use category can complicate the decision of which key receptors should be evaluated in an exposure scenario for residential/parkland land use. However, a common requirement among these land uses is to provide landscape and ecological settings that support the main land use activities (e.g., residential and parkland landscaping). Similar to agricultural land use, the development of soil quality guidelines for residential/parkland land use must ensure that the soil is capable of sustaining soil-dependent species and does not adversely affect wildlife from dermal contact and ingestion of contaminated soil or food. A generic exposure scenario for the residential/parkland land use category reflecting exposure pathways and receptors of choice is illustrated in Figure 6.

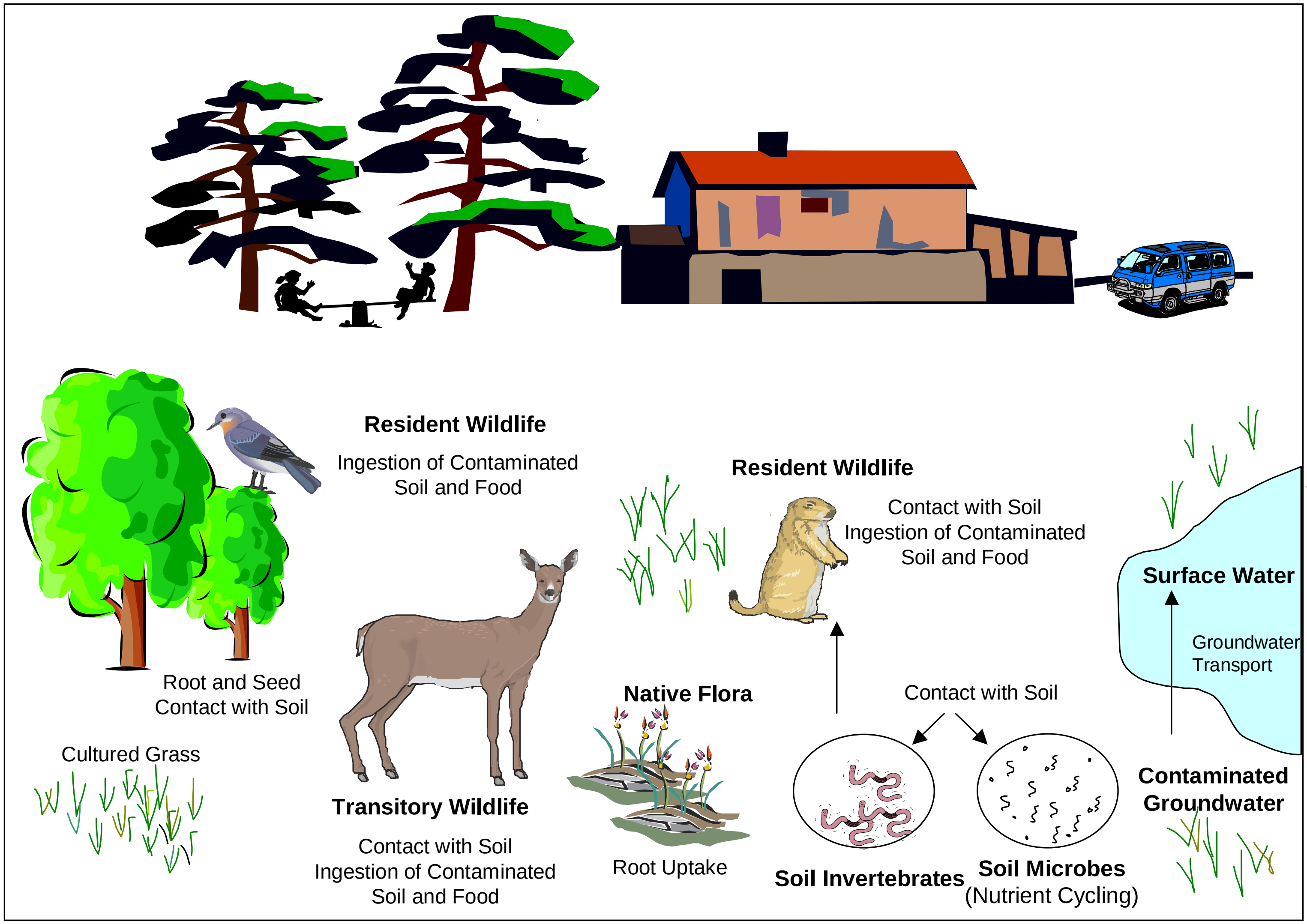


Figure 6: Key Receptors and Exposure Pathways of Contaminated Soil Considered for Residential/Parkland Land Use

5.2.1 Growth of Ornamental and Native Flora

To ensure that residential/parkland land use can support both ornamental and native flora, soil-dependent biota (whose ecological function helps sustain plant growth) must be protected from adverse effects as a result of dermal contact with contaminated soil (see Figure 6). Dermal contact by plant roots and seeds must also be examined. Root uptake and accumulation of contaminants will be examined as it relates to the ingestion of plant matter by wildlife.

Sufficient toxicological information exists to consider dermal soil contact by microbes (and their effect on nutrient cycling), soil invertebrates (e.g., decomposers), and crops and plants (e.g., seeds and roots) for guidelines derivation for the protection of ornamental and native plant growth (Figure 6). Currently, there is insufficient information to incorporate dermal absorption and translocation of contaminants by crops and plants via aerial deposition into generic guidelines derivation, but these exposure pathways should be examined when information becomes available.

5.2.2 Maintenance of Resident and Transitory Wildlife

To ensure that residential/parkland land use can sustain resident wildlife populations and transitory wildlife, an exposure scenario must consider the primary routes of exposure to wildlife on these lands. Dermal contact and contaminated soil and food ingestion by wildlife pose a significant health concern to these organisms. Information on the effects from these exposures to wildlife (other than some grazing herbivores) is severely lacking (OECD, 1988). Because of these data limitations, it is assumed that the level of protection offered to soil-dependent organisms from direct contact exposures is adequate to protect wildlife from dermal and ingestion exposures. This assumption is based on the notion that soil-dependent organisms are more directly in contact with the medium for a large portion of their life-cycle and will therefore be a more sensitive indicator of adverse effects than organisms at higher trophic levels. This assumption will be held except where explicit information to the contrary exists (e.g., molybdenum, selenium). However, effects from dermal contact and soil and food ingestion should be considered for guideline derivation for residential/parkland land use when information becomes available.

For substances that are persistent and have a strong tendency to bioaccumulate and/or biomagnify in the food chain, the consideration of direct contact with soil dependent organisms may not provide sufficient protection for all ecological receptors. Therefore, for residential/parkland land use, similar to agricultural land use, the derivation of soil quality guidelines for these substances should also consider the protection of herbivores and secondary and tertiary consumers from ingestion of contaminated soil and food.

5.3 Commercial Land Use

The nature of commercial land use is variable and can range from lands that approximate residential conditions (e.g., local gas stations) to lands that border on industrial activities (e.g., warehouses) (see Figure 3). This makes it difficult to describe key ecological receptors and exposure pathways for commercial land use. However, using the description of commercial land use in Section 2.3 of Part A, the degree to which maintenance of ecological functions is required will depend on the degree to which the site has been developed. From an ecological standpoint, SQGTG envisions generic commercial land to include managed (e.g., cultivated lawns, flowerbeds) as opposed to natural ecological areas (e.g., forests). The ecological receptors predicted to be present on commercial lands are similar to those identified for residential/parkland lands (i.e., soil-dependent biota, wildlife) since these receptors sustain the managed ecological areas of commercial lands. However, on commercial lands, it is assumed that the normal land use activities do not depend on the maintenance of ecological functioning to the same degree as on agricultural or residential/parkland lands.

The main route of exposure for soil-dependent biota and wildlife is most likely to be direct contact with contaminated soil. Due to the lack of information on the effects of direct soil contact on wildlife, the protection assumption used in the residential/parkland and agricultural land use categories is also used here. Soil ingestion by wildlife on commercial lands, at a generic level, is not thought to be significant because residence time on commercial lands is predicted to be low relative to agricultural and residential/parkland lands. Therefore guidelines are derived for commercial land use based on the direct contact of contaminated soil to soil-dependent biota and wildlife. A generic exposure scenario and receptor array for a commercial site is shown in Figure 7.

5.4 Industrial Land Use

It is assumed that on industrial lands, activities may not rely on protecting key ecological receptors to the same degree as agricultural and residential/parkland land use. However, it is not the recommendation of SQGTG that areas of industrial lands not be able to support any ecological activity, and consequently be viewed as a portion of the landscape in which high levels of contamination are permitted. Therefore, soil quality guidelines will be developed for industrial land use, but will not offer the same level of protection from adverse effects as guidelines for agricultural and residential/parkland land use. Industrial land use guidelines will be derived for direct soil contact by soil-dependent biota and wildlife and will offer the same level of protection as commercial land use guidelines. A generic exposure scenario and receptor array for an industrial site is shown in Figure 8.

5.5 Groundwater Protection

Contaminants present in the soil may be leached into shallow groundwater, which in turn may discharge to, or be intercepted by, a surface water body upon which ecological receptors depend. Therefore, there is a potential requirement for the protection of groundwater under all of the above land use scenarios.

A surface water body that supports aquatic life may exist within any land use classification. Therefore, soil quality guidelines for the protection of groundwater for aquatic life are required for all land uses. Toxicological information and/or surface water quality guidelines related to direct contact by aquatic life are available for many contaminants.

Under agricultural land use, shallow groundwater may be intercepted by dugouts or wells used for livestock watering and by wells or surface water used for irrigation purposes. For this land use soil guidelines should also be protective of water ingestion by livestock and wildlife, and water used for irrigation. Toxicological information is available to consider ingestion of contaminated water by livestock and wildlife.

Guidelines developed for the protection of freshwater life in nearby surface water bodies include saturated zone transport. It is assumed for generic guidelines that the nearest surface water body is at least 10 m away from the remediated soils; if surface water bodies are located closer to the remediated soils than 10 m, then a site-specific evaluation may be necessary. The saturated zone transport model is not considered to be appropriate for use at distances less than 10 m.

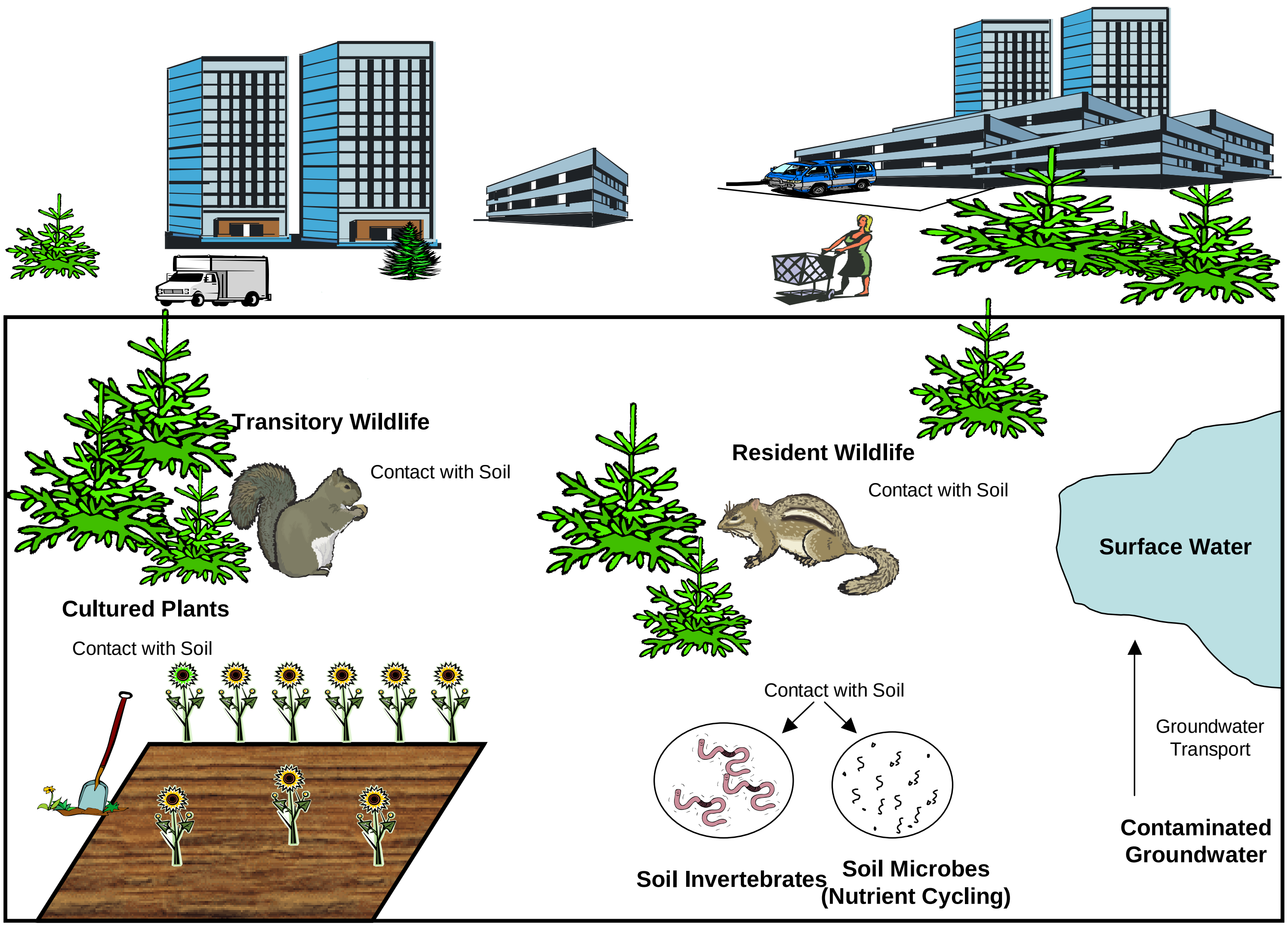


Figure 7: Key Receptors and Exposure Pathways of Contaminated Soil Considered for Commercial Land Use

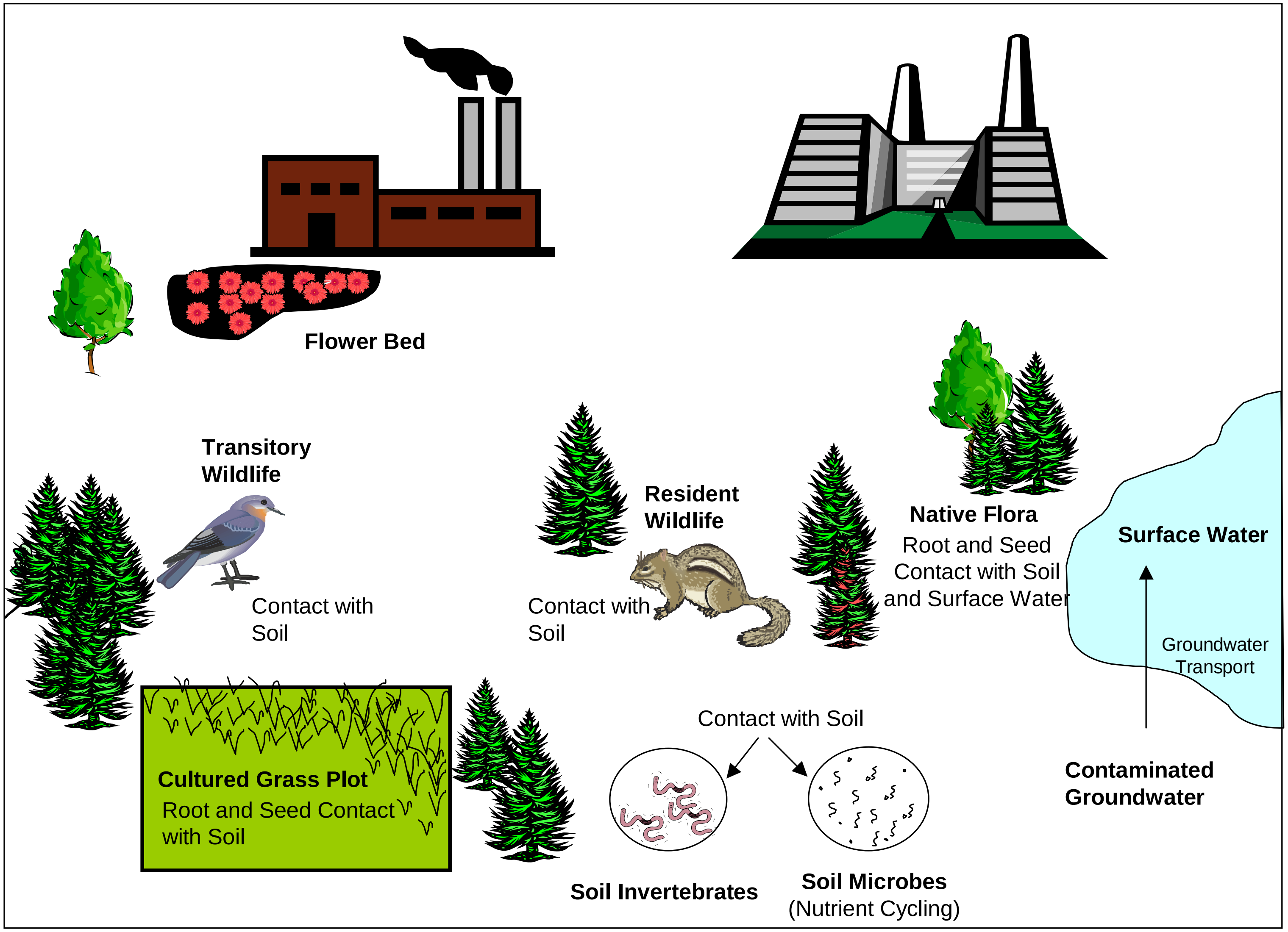


Figure 8: Key Receptors and Exposure Pathways of Contaminated Soil Considered for Industrial Land Use

SECTION 6
UNCERTAINTY IN GUIDELINES DERIVATION

6.1 Sources

Developing environmental soil quality guidelines requires a model that estimates (predicts) a level in soil considered to be acceptable for given land use exposure scenarios. The models employed in the ecological portion of this protocol are predictive in nature. Predictions based on model input parameters or mechanisms create aggregate uncertainties when the model is applied. In most regulatory settings, it is desirable to assess qualitatively, if not quantitatively, the level of uncertainty attached to these predictive models. This is done so that decision-makers understand the uncertainties associated with the scientific data on which the decision was based (Suter, 1993).

A wide spectrum of methods can assess the uncertainty associated with model predictions. In general, the major distinction between various approaches depends on whether the cause of concern is the propagation of error by models (sensitivity analysis methods), or the causes of prediction uncertainty (uncertainty or error analysis methods) (Summers et al., 1993). Suter (1993) describes three basic sources of uncertainty in ecological risk assessment:

- the inherent randomness in the world (stochasticity),
- imperfect or incomplete knowledge of things that could be known (ignorance), and
- mistakes in execution of assessment activities (error).

Finally, there is uncertainty in model error concerning the relationship between measurement endpoints used in guidelines derivation and the actual field population response or assessment endpoint. This uncertainty can be expressed as variance in the proportion of species to be protected at the level of the guideline and the confidence in that degree of protection (Suter, 1993).

6.2 Use of Uncertainty Factors in Guidelines Derivation

Traditionally, the development of environmental quality standards or guidelines, has relied on the application of uncertainty factors (UFs) (also referred to as safety factors) to a reference toxicological value (e.g., LOEC, $LC(D)_{50}$, EC_{50}) to arrive at a "safe" concentration that is extrapolated to field conditions. Uncertainty factors account for various sources of uncertainty associated with the model input parameters, model design, and to extrapolate to field conditions. It is important to note that, in the beginning, the use of uncertainty factors to account for model error was intuitive rather than scientific and it was stated emphatically that:

...the margin of safety concept is a reasonable approach to the matter, but that its acceptance should not fool researchers and/or the public into believing that there is an experimental or theoretical basis for its existence.

J.M. Barnes and F.A. Denz (1954) cited in Calabrese (1978)

The methods for determining the magnitude of specific uncertainty factors (UFs) are, in most cases, arbitrary. This approach assumes that the sources of uncertainty are independent of each other and may be combined in a multiplicative scheme (Calabrese and Gilbert, 1993). It has been argued (Calabrese and Gilbert, 1993) that a lack of independence exists between certain UFs used in human and ecotoxicological models, which results in an error in double counting of UFs.

Arbitrary UFs are used in environmental guideline derivation in place of probabilistic randomized block designs such as Monte Carlo Analysis to accommodate input parameter and model uncertainty. Although the three basic sources of uncertainty described by Suter (1993) are inherent in the development of environmental guidelines, the largest source of uncertainty may exist in model input parameters (toxicity data) and the result this has on model output (guidelines). It is this source of error that is primarily accounted for in guidelines derivation through the use of arbitrary UFs.

The protocol does not advocate the multiplicative use of UFs to account for other sources of uncertainty (extrapolation to field conditions), the result of which generally leads to environmental standards well below practicable levels. Other sources of uncertainty are assumed to be accounted for in the generally conservative approach incorporated in the development of environmental soil quality guidelines. It is acknowledged that this practice is subject to the disadvantages outlined by Paustenbach [cited in Suter (1993)], but given the status of toxicological data and model development for soil organisms, a quantifiable approach (e.g., probabilistic analysis) is not considered a "reliable" tool at this time. The specific application of uncertainty factors in guidelines derivation is discussed under individual derivation methods (Section 7.5).

SECTION 7
GUIDELINES DERIVATION PROCESS

The process for deriving soil quality guidelines for non-human biota according to the key receptors and exposure pathways previously described are found in Section 7. The general process for deriving soil quality guidelines is summarized in Figure 9.

7.1 Literature Review

For each contaminant, an extensive literature search of all published and non-proprietary data is conducted to obtain information on:

- physical and chemical properties,
- sources and emissions,
- distribution in the environment,
- environmental fate and behaviour,
- short- and long-term toxicity, and
- existing guidelines, standards, or criteria.

7.2 Evaluation of Laboratory and Field Toxicological Data

Because the quality of soil toxicity information is variable, toxicological data obtained from the literature must be screened for acceptability in generating soil quality guidelines. This ensures that studies selected will provide scientifically verified information. Candidate data are screened according to whether they are considered "acceptable" (referred to as selected) or "unacceptable" (referred to as consulted) for deriving soil quality guidelines. Similar procedures were used for the derivation of Canadian Water Quality Guidelines (CCME, 1999) and Ontario Water Quality Objectives (OMOE, 1988) and for screening ecotoxicological data for deriving soil standards in The Netherlands (van de Meent et al., 1990). The requirements for selected laboratory and field data are outlined below. When available, field data should be used in conjunction with laboratory data.

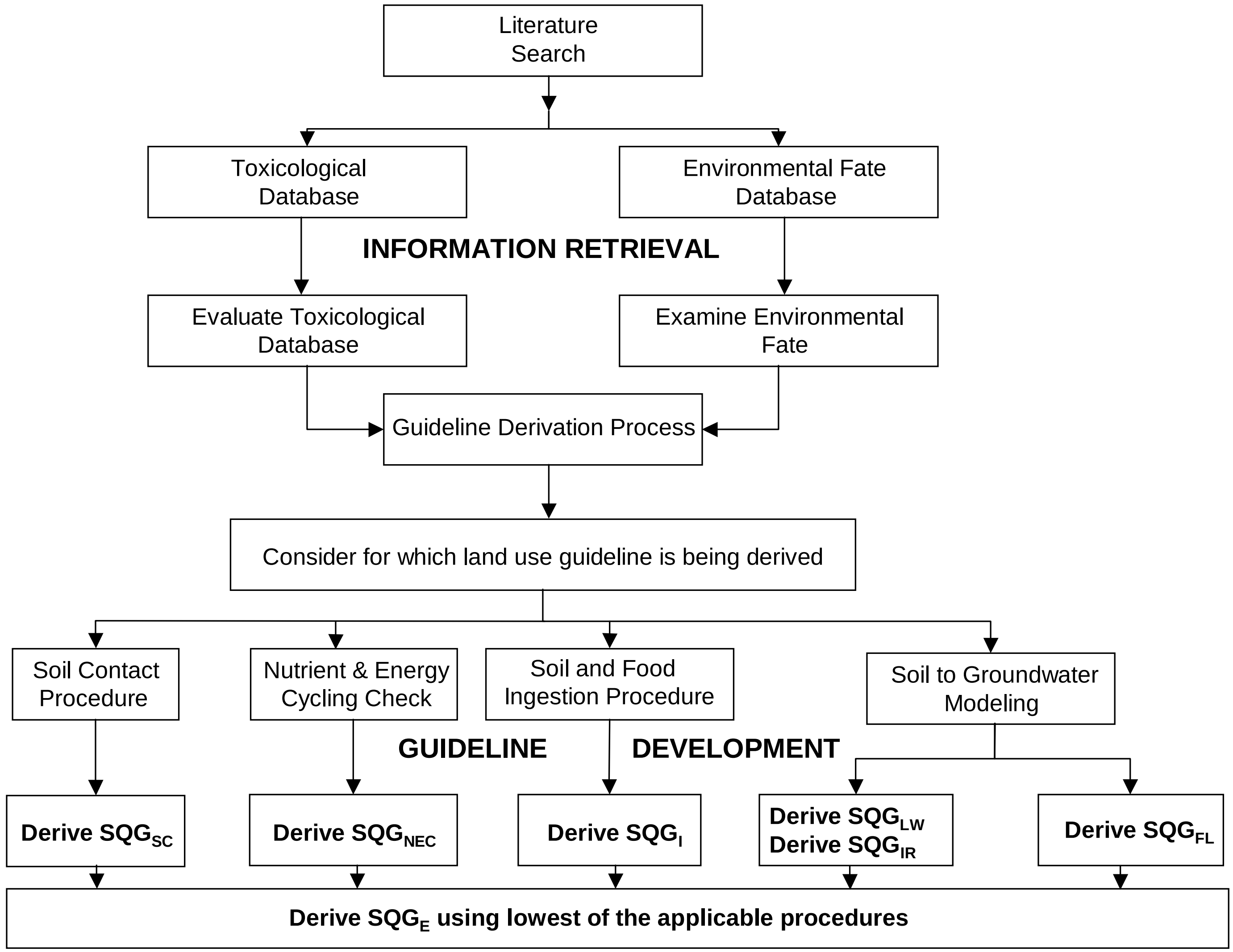


Figure 9: Overall Procedure for Deriving Environmental Soil Quality Guidelines for Agricultural, Residential/Parkland, Commercial and Industrial Land Uses

7.2.1 Laboratory Data

Selected Data

- Bioassay test procedures should conform to currently acknowledged and accepted soil toxicity testing practices or protocols (e.g., Environment Canada, 2004a, 2004b, 2004c; OECD, 1984, 1993; Green et al., 1989; ASTM, 1996, 1990a, 1990b; ISO, 1999; ISO, 1998, 1991, 1992). Data generated using non-standardized testing procedures should be evaluated case-by-case.

- Exposure time and recognized toxicological endpoints (e.g., mortality, reproduction, growth) for soil contaminants must be identified. Information from the dose-response curve should be used to estimate the IC_{25} or EC_{25} (or, failing that, the LOEC and NOEC endpoints).

- Environmental test conditions (e.g., soil type, pH, organic matter and clay content, moisture content, temperature, etc.) should be recorded so that factors affecting contaminant availability and toxicity can be evaluated.

- Appropriate statistical analysis should be performed and reported in the study.

- Tests that measure contaminant soil toxicity in combination with other conditions considered to be environmental stressors to the test organism (e.g., soil temperature changes), can be used provided that these stressors have been accounted for in the test design.

- Experimental effect must be attributable to contaminant of concern (avoid contaminant mixtures, such as sludges, unless clearly evident that the effect is due to the contaminant of concern).

- Studies which report measured values of contaminants in the soil must use comparable analytical methods for use in the derivation process, and should report an actual exposure concentration, not just an applied concentration (especially for volatile chemicals).

7.2.2 Field Data

Data from field studies can be used in the derivation of guidelines provided the preceding requirements are met as well as the following requirements:

- Effects data must be collected from the same site during the same time period and must be confirmed with matching soil chemistry data.

- Collection, handling, and storage of samples should conform to standardized or accepted practices (e.g., Greene et al., 1989).

- The acceptability of other field related variables (e.g., sampling design) should be evaluated on a case-by-case basis.

7.3 Minimum Toxicological Data Requirements

After compilation, review, and evaluation of the available information, selected data fulfilling the minimum toxicological data requirements specified for each of the procedures are used to derive soil quality guidelines. Minimum data requirements are designed to ensure guidelines are derived based on effects-data from a variety of organisms. In situations where there is a strong weight of evidence to suggest that the minimum data requirements do not apply, professional judgement may be used to derive a soil quality guideline based on a single class of organisms (e.g., when scientific evidence suggests that a single organism group is the most threatened).

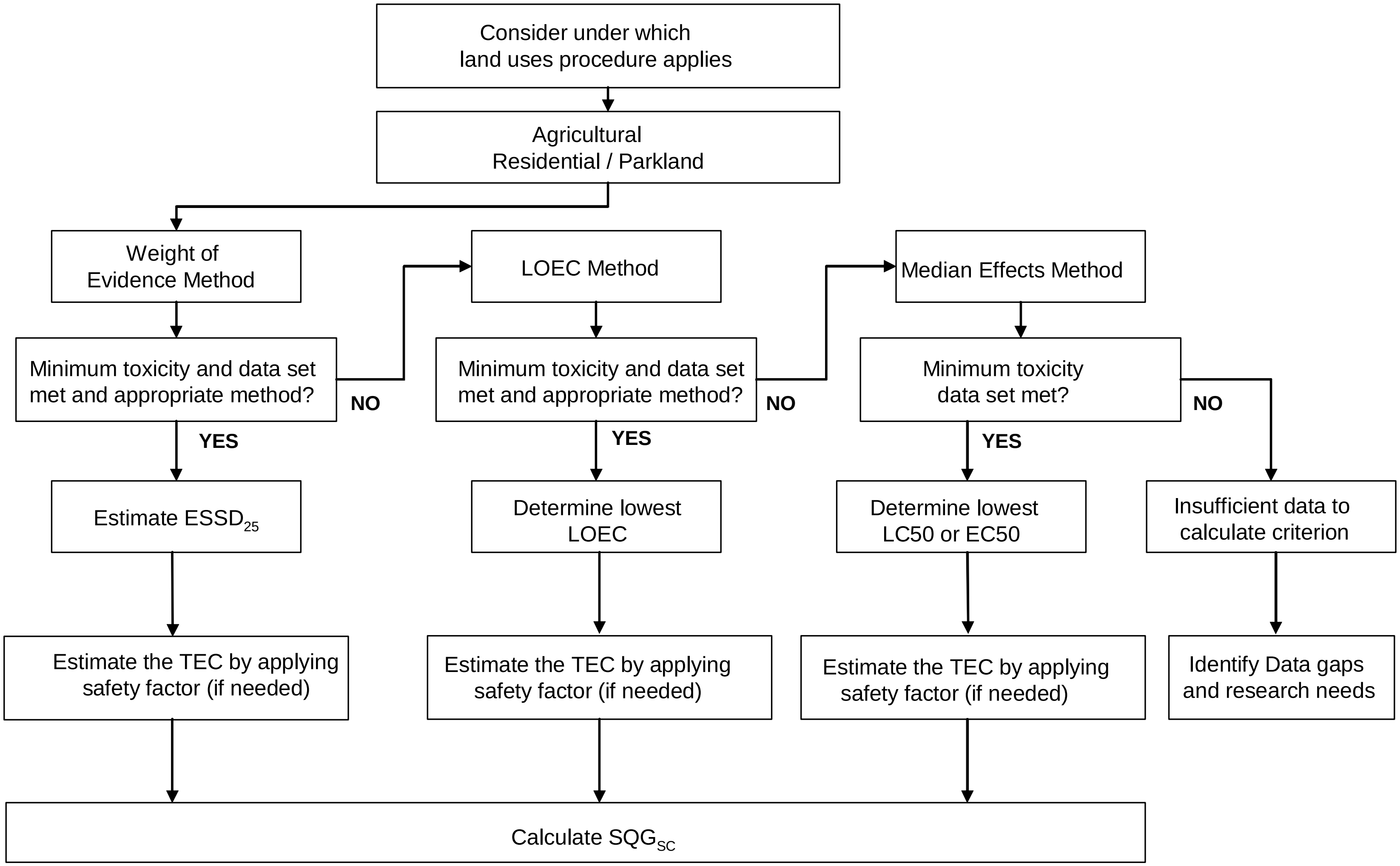
7.4 Environmental Behaviour Considerations

Studies which have examined the environmental fate and behaviour of contaminants in soil and terrestrial biota must be critically reviewed. Fate processes such as biodegradation, photolysis, hydrolysis, volatilization, adsorption, mobilization, and leaching should be examined. Physiological processes in biota including uptake, accumulation, metabolism, and elimination should also be reported. It is not necessary to have all the environmental fate information listed above to derive a guideline. However, it is necessary to have descriptive information on the major environmental factors that influence the potential hazard of the contaminant and its fate in the environment. Although derivation of environmental soil quality guidelines relies on toxicological data only, environmental behaviour data are critically examined and evaluated for pertinent information. During the assessment of individual contaminants, this information may play a key role in determining, for example, the derivation approach applied, chemical form of importance, usefulness of available toxicological data, or relevance of the final guideline in light of environmental behaviour.

7.5 Derivation of Soil Quality Guidelines for Soil Contact

7.5.1 Overview

The following sections describe the methods for deriving soil quality guidelines for soil contact (SQG_{SC}) by soil-dependent organisms. Based on the exposure scenarios previously discussed, guidelines derived using this procedure apply to all land use categories (i.e., agricultural, residential/parkland, commercial, and industrial). The methods are presented in order of preference. When minimum data are not available for a particular method, a measure of conservatism is added to each subsequent method to account for the inherent uncertainties of deriving guidelines from a less preferable data set.



Where: $ESSD_{25}$ = Estimated Species Sensitivity Distribution – 25th Percentile, TEC = Threshold Effects Concentration, LOEC = Lowest Observed Effect Concentration, LC_{50} = Median Effective Concentration.

Figure 10: Procedure for Deriving Soil Quality Guidelines for Soil Contact for Agricultural and Residential/Parkland Land Use

There are three methods to derive soil contact guidelines. In the first method, guidelines are derived using a weight-of-evidence approach to estimate the Threshold Effects Concentration (TEC) for agricultural or residential/parkland guidelines, and the Effects Concentration – Low (ECL) for commercial and industrial guidelines. Alternatively, if minimum data requirements cannot be met in order to use this method (as specified below), the second method is used. In the second method, the TEC and ECL are estimated by extrapolating from the lowest observable "adverse" effect concentration (LO(A)EC). If the minimum data requirements cannot be met using this method, then the third method is used. In this method, the TEC and ECL are determined by extrapolating from the median effective concentration (EC_{50}) or median lethal concentration (LC_{50}). If minimum data requirements cannot be met using this method, no environmental guideline can be set for soil contact, but data gaps and research needs will be identified.

The TEC or ECL is compared to a check value for nutrient and energy cycling (see Section 7.5.8 and Appendix B) to develop the soil contact guideline.

An overview of the soil contact guideline derivation procedure for agricultural and residential/parkland land uses is shown in Figure 10; the procedure for commercial and industrial land uses is shown in Figure 11.

7.5.2 Level of Protection

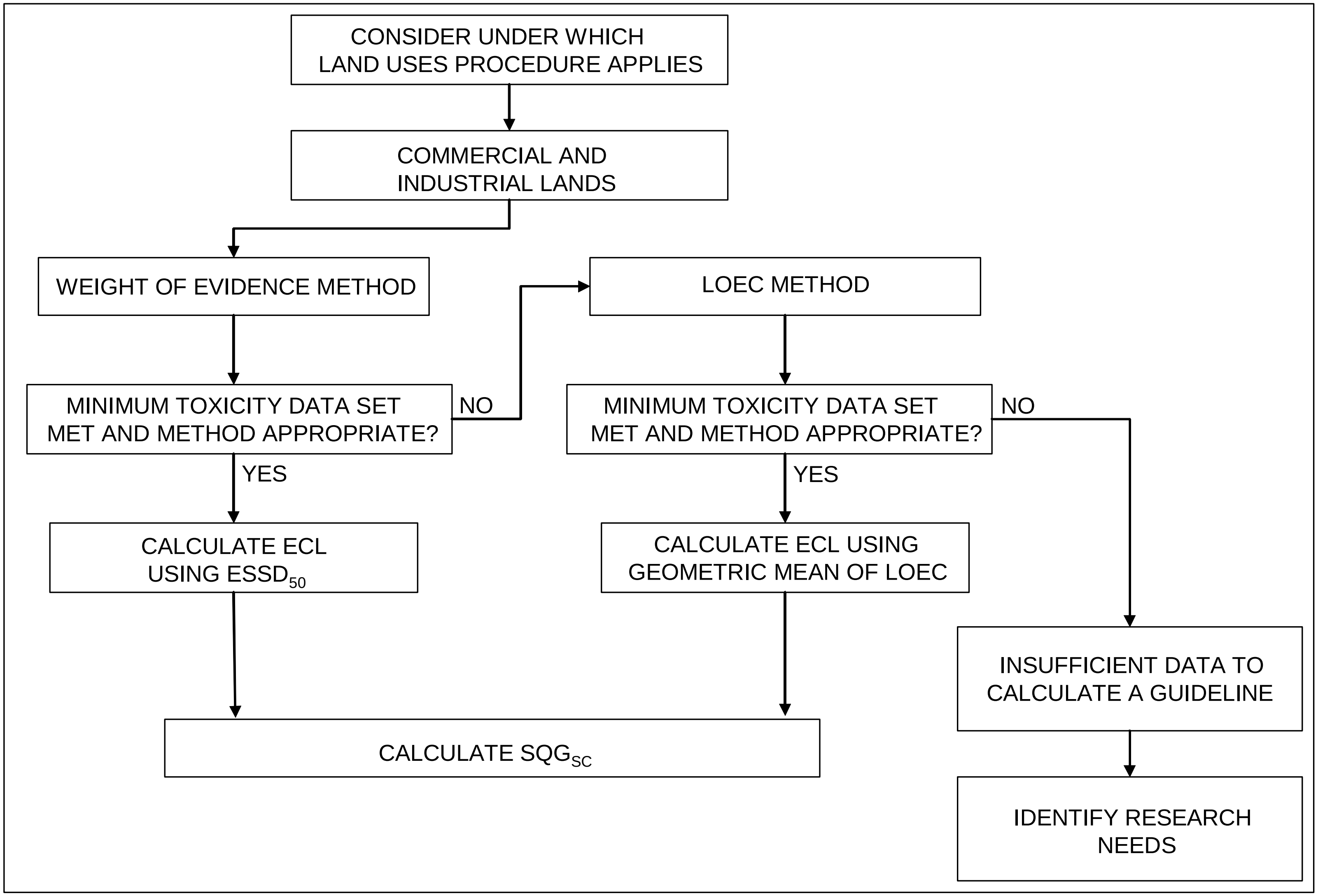
As discussed in Sections 5.1 through 5.4, the level of ecological function which needs to be preserved is dependent on land use. To account for the different levels of ecological protection needed, two separate values are calculated from the toxicity data: the Threshold Effects Concentration (TEC) and the Effects Concentration – Low (ECL).

The TEC represents the concentration of a contaminant in soil at which only minimal effects on ecological function would be observed. This concentration is used to derive guidelines for the agricultural and residential/parkland land uses.

The ECL represents a concentration of a contaminant in soil at which only a low level of adverse effects would be expected to occur in less than half of the species in the terrestrial community. The ECL is used to derive guidelines for the commercial and industrial land uses.

7.5.3 Soil Type Considerations

Where sufficient data exist, coarse-grained and fine-grained soils should be considered separately, and guidelines developed for each soil type. However, in many cases there may not be sufficient data to develop separate soil contact guidelines for both soil types, or the data requirements of a preferred derivation method may only be met for one soil type. It should be noted that some jurisdictions may only allow consideration of soil type on a site-specific basis.



Where: $ESSD_{50}$ = Estimated Species Sensitivity Distribution – 50th Percentile, ECL = Effects Concentration Low, LOEC = Lowest Observed Effect Concentration, SQG_{SC} = Soil Quality Guideline for Soil Contact.

Figure 11: Procedure for Deriving Soil Quality Guidelines for Soil Contact for Commercial and Industrial Land Use

The data set should be evaluated on a case-by-case basis, and professional judgement applied. Possible solutions may include:

- Data for the two soil types can be combined, and the resulting guideline applied for both soil types.
- A soil contact guideline can be developed for a soil type for which sufficient data are available. This guideline can be applied to the other soil type as a provisional guideline.
- A soil contact guideline can be developed for a soil type for which sufficient data are available, and the guideline adjusted for the other soil type by a factor determined by professional judgement. This factor could be based on toxicity studies for the chemical that consider the effects of soil type, or on the behaviour of similar chemicals. The resulting soil contact guideline would be considered a provisional guideline.

The approach taken should consider any apparent differences in the toxicity of the chemical in different soil types.

7.5.4 Bioavailability Considerations

The bioavailability (and toxicity) of contaminants to soil organisms can be affected by numerous factors, including organic carbon content, pH, ion exchange capacities, clay content, and aging (US EPA, 2003). In addition, for inorganic contaminants the bioavailability can be affected by chemical speciation. The form in which the contaminant is added (e.g., as a metal oxide versus a salt) may also affect its bioavailability.

The soil contact guidelines should be developed based on data reflective of typical Canadian soils. Ideally, the toxicity data used to develop the soil contact guidelines should include a wide range of soil conditions. However, in some cases the data may reflect only a limited range of conditions, or be biased towards data from a particular soil type.

It is anticipated that, in most cases, data are likely to be biased towards conditions of relatively high bioavailability. However, in some cases data may be biased towards conditions of relatively low bioavailability; this could lead to the development of a soil contact guideline which is not protective of most Canadian sites. Therefore, the bioavailability conditions for toxicity studies used to develop the soil contact guideline should be evaluated.

The soil organic carbon content and pH, both of which can have a significant effect on bioavailability, are routinely reported in soil toxicity studies (US EPA, 2003). In general, a soil organic carbon content of 6% or greater indicates that bioavailability is likely low. A high pH (greater than 7) can result in low bioavailability of inorganic cations, while a low pH (less than 5.5) can result in low bioavailability of anions. Studies conducted under conditions of very high bioavailablity (i.e., very low pH and low organic carbon content) may not be relevant when deriving guidelines for agricultural land uses in particular. For inorganic compounds, the bioavailability of the particular species (if applicable) should also be evaluated based on the literature reviewed during guideline development. If, based on professional judgement, more than 50% of the data used to develop the soil contact guideline reflect low bioavailability conditions, consideration should be given to applying an uncertainty factor. If all of the data reflect low bioavailability conditions, the soil contact guideline should be classified as a provisional guideline.

7.5.5 Weight of Evidence Method

This method is a modification of an approach used for calculating sediment quality guidelines for the National Status and Trends Program (NSTP) (Long and Morgan, 1990), and an approach proposed by Smith and MacDonald (1993) for deriving Canadian Sediment Quality Guidelines. These methods use a percentile of the effects data set, or combined effects and no effects data set, to estimate a concentration in the sediment expected to cause no adverse biological effects.

There are two approaches to the Weight of Evidence Method for ecological soil contact guidelines. The preferred approach is to compile IC_{25} and EC_{25} data. If, however, insufficient IC_{25} and EC_{25} data are available and these values cannot be derived from the dose-response curves, then the combined set of "effects" and "no observed effects" data can be used instead.

7.5.5.1 EC_{25} Distribution

The preferred approach to developing soil contact guidelines is to compile EC_{25} and/or IC_{25} effects-endpoints from all selected studies. This approach has been adapted from CCME (2000). If an EC_{25} (or IC_{25}) value is not available for a selected study and cannot be determined from the information presented in the study (or by contacting the authors of the study for additional data), the effects-endpoint (IC_X or EC_X) value for the point where 'X' is closest to 25% (generally between 20% and 30%) should be used. NOEC and LOEC data should not be used with this approach.

Data for plants and invertebrates should, where possible, be evaluated separately, with the lower of the generated guidelines being taken as the SQG_{SC}. However, this requires that the data requirements for the method be met by each of the plant and invertebrate data sets. If the data are inadequate, then plant and invertebrate data sets may be combined. Professional judgement should be used to determine whether there are sufficient data to evaluate plants and invertebrates separately while still retaining statistically valid data distributions.

At least ten data points from at least three studies are required to perform this procedure. When three studies are not available, professional judgement should be used to determine whether or not the data available are sufficient in breadth, scope and quality to derive a suitable guideline. If multiple data points from a single study are used, they must be discrete end points. For example, two EC_{25} values for different plant species from the same study would be considered discrete, whereas an EC_{20} and EC_{30} for a single plant species from the same study would not be (and only one of these values should be used). A minimum of two soil invertebrate and two crop/plant data points must be represented. Single studies reporting data for multiple species and/or multiple endpoints will be considered as separate data entries. Where data are reported for a large number of different endpoints on the same species, loading the distribution with all of the data may not be appropriate because it could give too much weighting to that one species, rather than representing the distribution of species sensitivities. Therefore, professional judgement should be used in selecting the most relevant, sensitive endpoints for inclusion.

Data points for the same species that are redundant should be combined into a single composite response concentration calculated as the geometric mean of the individual values. Individual toxicity data points are considered redundant if they:

- represent the same type of response and response level (e.g., two different EC_{25} values for earthworm reproduction) under the same or highly similar exposure conditions; or,

- were based on different response data which are known to be directly, causally connected (e.g., plant wet weight and dry weight).

If toxicity data are available for the same species and response type, but different response levels (e.g., EC_{25} and EC_{50}), only the value closest to an EC_{25} should be used. If toxicity data are available for the same species, response type, response level and exposure conditions, based on different exposure periods, then the data for the longer exposure period should be given preference. In some cases, data points may also be combined if the data are for the same species and response type but for different soil types, particularly if including all of the data points will result in a significant bias of the EC_{25} distribution towards a single species, though it should be noted that variations in toxicity due to the effects of exposure conditions are a valid part of the overall sensitivity distribution. Professional judgement should be used in these cases.

The resulting data are ranked, and rank percentiles determined for each data point. For data points with the same concentration, it is recommended that these be assigned separate, sequential ranks, rather than calculating tied ranks. Rank percentiles should be calculated using the equation:

$$j = \frac{i}{(n+1)} \times 100$$

where,
j = rank percentile
i = rank of the data point in the data set
n = total number of data points in the data set.

It should be noted that some commonly used software applications have a pre-set function for calculating rank percentile, but these may use different equations which would result in slightly different values.

It is recommended that a graph of rank percentile versus concentration of the chemical be prepared (see Figure 12 for an example); these graphs assist with scrutiny of the underlying data distribution. The data should be evaluated for anomalies, to ensure that this method is appropriate. The 25th percentile of the rank distribution, identified as the "estimated species sensitivity distribution – 25th percentile" ($ESSD_{25}$), is used as the basis for soil contact guidelines for the agricultural and residential/parkland land uses. The 50th percentile of this distribution, identified as the “estimated species sensitivity distribution – 50^{th} percentile” ($ESSD_{50}$), is used as the basis for soil contact guidelines for the commercial and industrial land uses. The $ESSD_{25}$ and $ESSD_{50}$ should be estimated from the graph or calculated from the regression equation for the line in the graph.

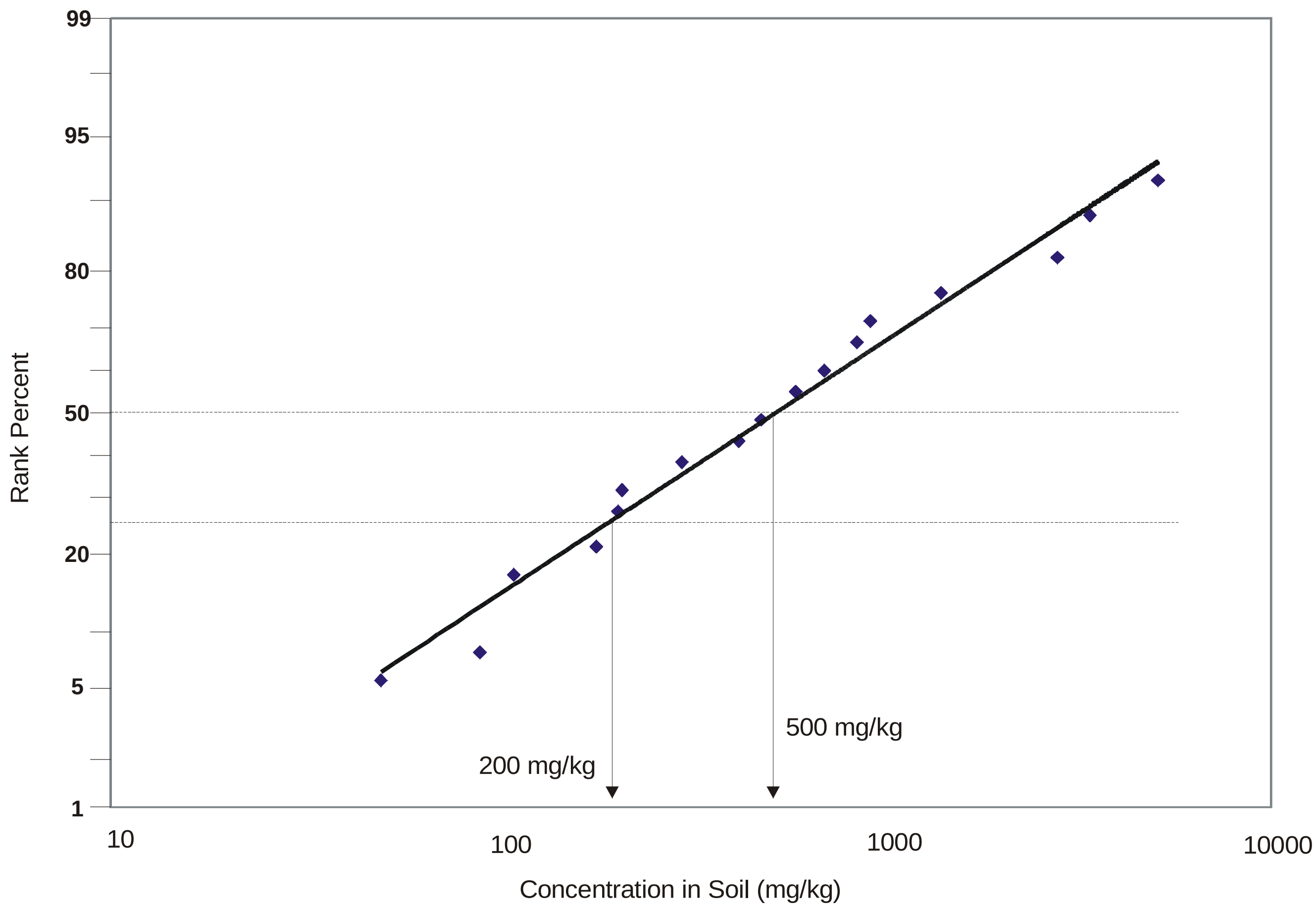


Figure 12: Example of a Rank Probability Plot of Bioassay Data

The TEC and ECL are calculated using the following equations:

$$TEC = ESSD_{25}/UF$$
$$ECL = ESSD_{50}$$

where
TEC = threshold effects concentration (mg/kg)
ECL = effects concentration - low
$ESSD_{25}$ = estimated species sensitivity distribution – 25^{th} percentile (25^{th} percentile of the distribution) (mg/kg)
$ESSD_{50}$ = estimated species sensitivity distribution – 50^{th} percentile (50^{th} percentile of the distribution) (mg/kg)
UF = uncertainty factor (if needed)

An uncertainty factor can be applied after examining the data used to estimate the $ESSD_{25}$. A UF between one and five is suggested using the criteria below as a guide to determine the magnitude of the uncertainty if:

- only the minimum of three studies is available.
- more than three studies are available, but fewer than three taxonomic groups are represented.
- greater than 50% of the data for either plant or soil invertebrate toxicity is below the 25^{th} percentile of the distribution if these data sets are combined.
- short-term toxicity studies were used to meet the minimum data requirements.
- more than 50% of the data are based on toxicity studies with low bioavailability conditions.

It should be noted that an uncertainty factor need not always be applied. Expert judgement should determine the magnitude of the uncertainty factor based on the above criteria. An uncertainty factor greater than five for other sources of uncertainty (e.g., field extrapolation) is not recommended since the point of departure for the $ESSD_{25}$ is considered conservative and accounts for these and other (e.g., model error) uncertainties. Uncertainty factors are not normally applied to the $ESSD_{50}$.

7.5.5.2 Effects/No Effects Data Distribution

In the event that EC_{25} values cannot be determined for a sufficient number of toxicity studies, but all other requirements of the above approach are met (at least ten data points from three studies, including at least two invertebrate and two plant/crop data points), then an effects/no effects distribution can be used instead. As for the EC_{25} method described above, if possible the plant and soil invertebrate data should be evaluated separately.

For this method, all selected "effects" and "no observed effects" data are collected and a frequency distribution using the empirical distribution function with averaging is created. Effects data should include lowest observable effect concentration (LOEC) and median effective concentration (EC_{50}) data. Other types of selected "effects" data (e.g., LC_{50} or $E(L)C_{<50}$) must be evaluated for suitability before inclusion in the distribution. As in Section 7.5.4.1 above, a geometric mean is determined for redundant data points.

Some concerns have been raised with respect to this approach (CCME, 2000), particularly relating to the use of NOEC and LOEC data, which are affected by the experimental protocol and may be subject to statistical issues. Also, the combination of mortality endpoints with sublethal endpoints may result in a highly variable level of environmental protection for guidelines derived using this approach.

Even though minimum data requirements have been met for this method, it may not always be an appropriate approach to derive guidelines. The main concern is the bias created in the overall distribution when data in any particular quantile greatly outnumber data in other quantiles. For example, if greater than 50% of the "effects" data entries are dominated by median effective or median lethal concentrations, or if greater than 75% of the distribution is dominated by NOEC data, a bias is created in the area of the curve where *n* is greatest. In the case where the curve is dominated by LOEC and NOEC (e.g., >75% LOEC and NOEC data), additional concerns are raised about the uncertainty of this data set (see Section 6.1). While a normal distribution would best suit the guidelines derivation using this method, asymmetric distributions predominate in biological effects data. Expert judgement should determine if the degree of bias in the data makes this method inappropriate for guidelines estimation. Where a high degree of bias is created from the preponderance of LOEC and NOEC or EC_{50}/LC_{50} data, the Lowest Observed Effect Concentration Method (see Section 7.5.5) or the Median Effects Method (MEM) (see Section 7.5.6) are preferable for guidelines derivation due to expected lower levels of uncertainty.

The 25th percentile of the effects/no effects frequency distribution (the $ESSD_{25}$) was chosen as the basis for soil contact guidelines for agricultural and residential/parkland scenarios based on the analysis of data from cadmium, pentachlorophenol, and arsenic. The $ESSD_{25}$ represents a point estimate in the distribution below which the proportion of definitive effects data (EC_X, LC_X) does not exceed "acceptable levels" (25%). Definitive effects data below the $ESSD_{25}$ are "minimal effects concentrations" that overlap the range of LOECs and NOECs used to determine the $ESSD_{25}$. Norberg-King (1988) used a 25% effect level as an estimate of the minimal effect level. Due to the variability inherent in LOEC data and its proximity to NOEC estimates on distribution curves (often overlapping NOEC points [see Figure 11]), it is not considered a definitive effect, but rather as a potential effect estimate.

The 50^{th} percentile of the effects/no effects frequency distribution (the $ESSD_{50}$) was chosen as the basis for soil contact guidelines for the commercial and industrial scenarios. It is expected that some effects will be incurred by soil-dependent biota at this level, but not at the level of median lethality in the population(s). It should be noted that this approach is slightly different than the version of this method presented in the 1996 version of the protocol.

The TEC and ECL are calculated using the following equations:

$$TEC = ESSD_{25}/UF$$
$$ECL = ESSD_{50}$$

where:

TEC	=	threshold effects concentration (mg/kg)
ECL	=	effects concentration – low (mg/kg)
$ESSD_{25}$	=	estimated species sensitivity distribution – 25th percentile (25th percentile of the distribution) (mg/kg)
$ESSD_{50}$	=	estimated species sensitivity distribution – 50th percentile (50th percentile of the distribution) (mg/kg)
UF	=	uncertainty factor (if needed)

An uncertainty factor can be applied after examining the data used to estimate the $ESSD_{25}$. A UF between one and five is suggested using the criteria below as a guide to determine the magnitude of the uncertainty if:

- only the minimum of three studies is available.
- more than three studies are available, but fewer than three taxonomic groups are represented.
- more than 25% of the data below the 25th percentile are definitive effects data.
- short-term toxicity studies were used to meet the minimum data requirements.
- more than 50% of the data are based on toxicity studies with low bioavailability conditions.

It should be noted that an uncertainty factor need not always be applied. Expert judgement should determine the magnitude of the uncertainty factor based on the above criteria. An uncertainty factor greater than five for other sources of uncertainty (e.g., field extrapolation) is not recommended since the point of departure for the $ESSD_{25}$ is considered conservative and accounts for these and other (e.g., model error) uncertainties. Uncertainty factors are not normally applied to the $ESSD_{50}$.

7.5.6 Lowest Observed Effect Concentration Method

When the minimum data requirements for the weight of evidence method cannot be met, the TEC is derived by extrapolating from the lowest available, lowest observed effect concentration (LOEC) divided by an uncertainty factor (if needed), and the ECL is derived using the geometric mean of the available LOEC data. In this method, the TEC is estimated to be somewhere below the lowest reported effects concentration (see Figure 9) while the ECL is estimated to be somewhere in the range of low level observable effects.

The TEC is calculated using the following equation:

TEC = lowest LOEC/UF

where
TEC = threshold effects concentration (mg/kg soil)
LOEC = lowest observed effect concentration (mg/kg soil)
UF = uncertainty factor (if needed)

The ECL is calculated according to the equation:

$$ECL = (LOEC_1 \times LOEC_2 \times ... \times LOEC_n)^{1/n}$$

where
ECL = effects concentration low (mg/kg)
LOEC = lowest observed effect concentration (mg/kg)
n = the number of available LOECs

A minimum of three studies reporting LOEC endpoints must be considered. Requirements also include at least one terrestrial plant and one soil invertebrate study.

If expert judgement determines that an UF is warranted for the calculation of the TEC, the following criteria should be used as a guide for application to determine an UF between one and five:

- The LOEC is considered "biologically significant" and not just statistically different from controls, and therefore extrapolation below this level of effect is required.
- The LOEC is taken from an acute lethal or sublethal study.
- Only the minimum number of studies (three) was available to select the lowest LOEC.
- Fewer than three taxonomic orders are represented when selecting the lowest LOEC.

An uncertainty factor greater than five for other levels of uncertainty (e.g., model error, intra-species variation) is not recommended since a large measure of conservatism is already added by selecting the species with the lowest available LOEC.

7.5.7 Median Effects Method

Alternatively, if the minimum data requirements cannot be met for the weight of evidence and LOEC methods, the TEC is derived by extrapolating from the lowest available EC_{50} or LC_{50} datum using an uncertainty factor (UF). In this method, the TEC is estimated in the region of predominantly no effects (see Figure 11). The TEC is calculated as follows:

TEC = lowest EC_{50} or LC_{50}/UF

where
TEC = threshold effects concentration (mg/kg soil)
EC_{50} = median effective concentration (mg/kg soil)
LC_{50} = median lethal concentration (mg/kg soil)

UF = uncertainty factor

A minimum of three studies must be considered to select the lowest EC_{50} or LC_{50}, including one terrestrial plant, and one soil invertebrate study. If the lowest datum is an EC_{50} value, then an uncertainty factor of five should be initially applied to derive the TEC. If an LC_{50} is used as the lowest datum, then an uncertainty factor of 10 should initially be applied. The selection of these uncertainty factors is based on median acute/chronic ratios determined for EC_{50} and LC_{50} data versus NOEC data for 38 inorganic and organic contaminants for soil-dependent organisms (Bonnell, 1992). Uncertainty factors of 5 and 10 have also been proposed for use in deriving guidelines for soil from short-term data (Dennemen and van Gestel, 1990; van de Meent, 1990; van der Berg and Roels, 1991). An additional uncertainty factor between one and five may be applied if points two, three, or four listed in the LOEC method for UF selection are incurred.

No median effects method is recommended for guidelines derivation for commercial and industrial land use. Because uncertainty factors are not applied at the point of departure from the effects distribution, the ECL would therefore be estimated at a level of median effects, which is contrary to the level of protection desired at the level of the ECL.

7.5.8 Insufficient Data for Soil Contact Guidelines Derivation

If minimum data requirements for the above methods cannot be met, then there is insufficient information to develop a final environmental soil quality guideline (SQG_E). Data gaps will be identified for further research.

7.5.9 Confidence Ranking for the Soil Contact Guideline

A confidence ranking is defined for the soil contact guideline as defined below:

Approach Used	Separate Plant & Invertebrate	Combined Plant & Invertebrate
Weight of Evidence – EC_{25}	A	B
Weight of Evidence – Effects/No Effects	C	D
LOEC	E	F
Median Effects	G	H
Provisional Guideline	I	J

The confidence ranking is presented in the scientific supporting document and fact sheet.

7.6 Derivation of Soil Quality Guidelines for Soil and Food Ingestion

The procedure for deriving soil quality guidelines for ingestion of soil and food (SQG_I) by grazing livestock and wildlife is described in this section. Current knowledge of effects to terrestrial receptors from the ingestion of contaminated soil and food is best understood, with some exceptions (e.g. some insectivores), for herbivores (livestock and wildlife) grazing on agricultural lands and is considered to be the most significant route of contaminant exposure to these receptors (Thorton and Abrahams, 1983; Fries, 1987; Paustenbach, 1989). This procedure also accounts for the consumption of contaminated forage via the accumulation of contaminants in the food chain. Because this procedure is limited to a herbivore food chain, chemicals that bioaccumulate in the tissues of plants and that can be transferred in the food chain are of primary importance.

In view of the data requirements and model parameters used to estimate generic guidelines for soil and food ingestion, it is only possible to derive ingestion guidelines where data are sufficient to keep model parameter uncertainty at a minimum and also reduce the need for large inter-species extrapolations. Therefore, until more data are available for other receptors, it is the opinion of SQGTG that guidelines for soil and food ingestion should only be derived for grazing herbivores on agricultural lands. However, if there is evidence that a particular type of wildlife (e.g., amphibians) is particularly sensitive to the contaminant being evaluated, then that type of wildlife may be evaluated on a contaminant-specific basis if there are sufficient data available.

An exception to the above is in the case of persistent substances that have a strong tendency to bioaccumulate and/or biomagnify, such as PCBs. For these substances, food chain pathways can lead to the exposure of ecological receptors at higher trophic levels, i.e. secondary and tertiary consumers. In the case of these substances, where data are sufficient, guidelines for soil and food ingestion should be developed for the protection of primary, secondary and tertiary consumers for both agricultural and residential/parkland land uses.

Figure 13 gives an overview of the derivation procedure for soil quality guidelines for soil and food ingestion.

7.6.1 Derivation of Soil Quality Guideline for Soil and Food Ingestion for Protection of Primary Consumers

7.6.1.1 Determining the Species Most at Threat from Soil and Food Ingestion

The first step requires the determination of the species considered to be most at threat from contaminated soil and food ingestion. Oral toxicological data from grazing and foraging species are used to determine which species are potentially at threat from the ingestion of contaminants. The species "most" threatened has the highest ratio of exposure (based on soil/food ingestion rates and body weight) to the Daily Threshold Effects Dose (DTED) (see below). A minimum of three studies must be considered. At least two of these must be oral mammalian studies and one should be an oral avian study. A maximum of one laboratory rodent study may be used to fulfil the data requirements for mammalian species if needed. A grazing herbivore (e.g. ungulates) with a high ingestion rate to body weight ratio should be considered in the minimum data set. Where possible, field data should be used in conjunction with laboratory data.

If minimum data requirements cannot be met when determining the DTED, then no soil quality guidelines for soil and food ingestion shall be set. Data gaps will be identified for further research.

7.6.1.2 Calculation of the Daily Threshold Effect Dose

The DTED is estimated using the lowest effects dose (ED_{1C}, where the subscript $_{1C}$ stands for primary consumer) from the species determined to be most threatened in Section 7.6.1.1, divided by an uncertainty factor (UF). The DTED is calculated according to the following equation.

$$DTED_{1C} = \text{lowest } ED_{1C} / UF \qquad [1]$$

where:

where:	$DTED_{1C}$	=	daily threshold effects dose of the primary consumer (mg/kg bw_{1C}-day)
	ED_{1C}	=	lowest effects dose (mg/kg bw_{1C}-day)
	UF	=	uncertainty factor (if needed)

If possible, the lowest effects dose should be based on an ECx value, where the preferred value of x, representative of a no effects range, is 25 (see discussion on soil contact in Section 7.5). Depending on data quality and availability, the lowest effects dose may be the lowest observed (adverse) effects level (LO(A)EL).

An uncertainty factor between one and five can be applied using expert judgement. The following criteria are a guide for UF application:

1. The effects dose is considered to be "biologically significant" and not just statistically different from controls and therefore extrapolation below the effects dose is required.

2. The effects dose is taken from an acute lethal or sublethal study.

3. Only the absolute minimum data requirements have been met.

4. Fewer than three taxonomic groups are available to select the lowest effects dose.

An uncertainty factor greater than five for other levels of uncertainty (e.g. intra-species variation, extrapolation to field conditions) is not recommended since a large measure of conservatism is already added by selecting the most threatened species.

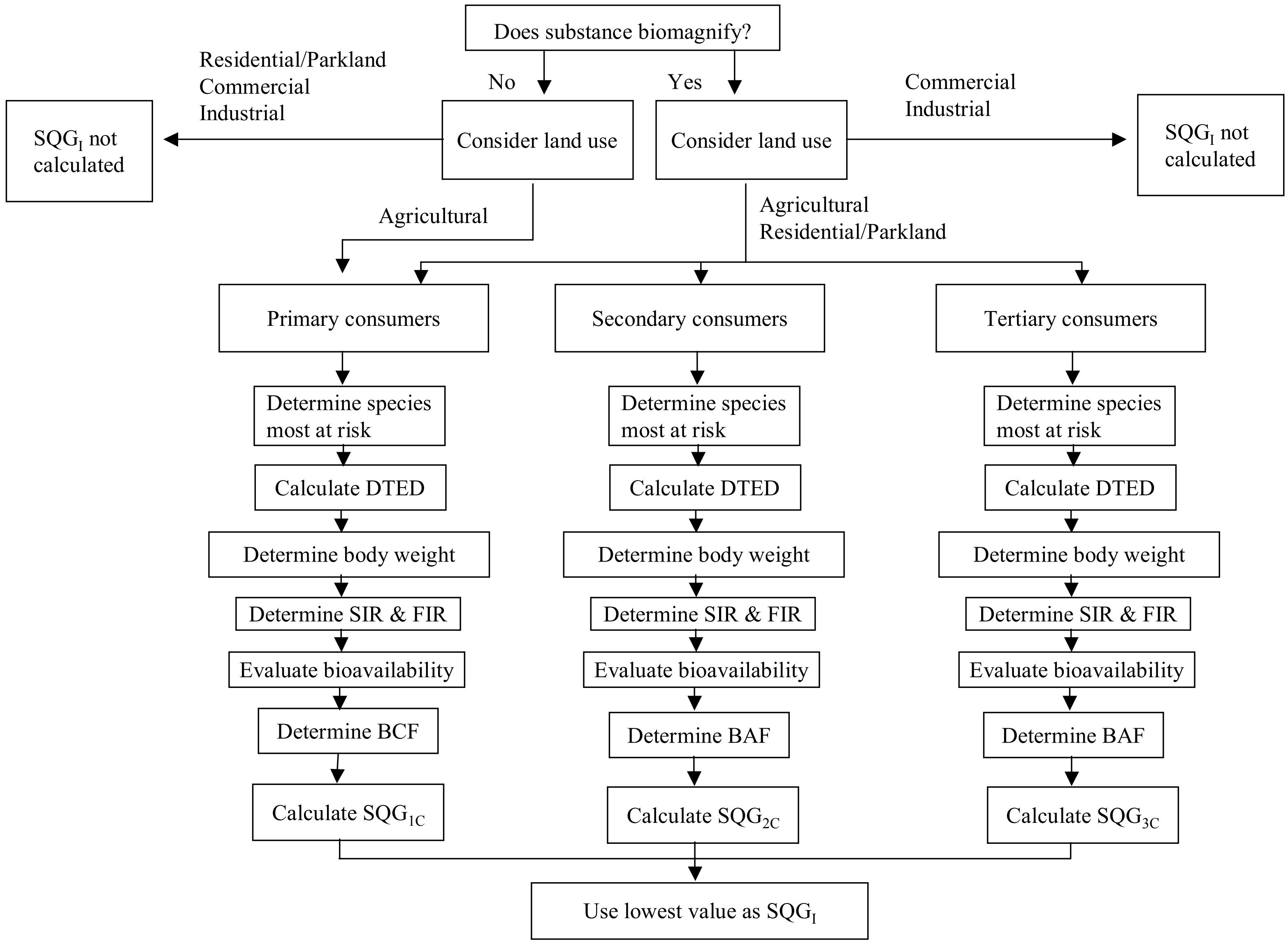


Figure 13: Procedure for Deriving Soil Quality Guidelines for Soil and Food Ingestion

7.6.1.3 Determining Body Weight

The mean body weight for the species used to calculate the $DTED_{1C}$ is estimated and is required with the $DTED_{1C}$, as well as soil and food ingestion rates (SIR and FIR), to evaluate the soil quality guideline for ingestion (Section 7.6.1.7).

7.6.1.4 Determining Exposure from Soil and Food Ingestion

Exposure from Soil Ingestion

Primarily, animals ingest soil as a result of soil adhering to forage. The ingestion rate of soil and forage together is referred to as the dry matter intake rate (DMIR). To estimate the rate of soil ingested directly, the percentage of the DMIR attributed to soil ingestion must be isolated. In most soil-based exposure studies, the proportion of soil ingested (PSI) is reported with the DMIR. Therefore, an animal's soil ingestion rate (SIR) is calculated as a proportion of the DMIR according to the following equation:

$$SIR_{1C} = (DMIR_{1C} \times PSI_{1C}) \qquad [2]$$

where:

SIR_{1C}	=	the soil ingestion rate of the primary consumer (kg dw soil/day)
$DMIR_{1C}$	=	geometric mean of the available dry matter intake rates for the primary consumer (kg dw/day)
PSI_{1C}	=	geometric mean of available soil ingestion proportions reported with the DMIR

It is assumed that the DMIR contains only dry matter as food or soil. The geometric mean of available DMIR values is used based on the assumption that the data are lognormally distributed; if there is evidence of a different distribution, then an alternate estimate (e.g., arithmetic mean) can be used based on professional judgement. If no DMIR data are available for the species selected to represent the primary consumer, the geometric mean DMIR for grazing herbivores can be employed in the preceding equation. A review of soil ingestion information comparing wildlife and livestock species (McMurter, 1993) reported that a small difference existed in the PSI among domestic grazing herbivores or wildlife. Therefore, if no information is available on the PSI for the species selected as the primary consumer, a default value of 0.083 shall be used for domestic livestock while a default value of 0.077 shall be used for wildlife species, in the preceding equation (McMurter, 1993).

Exposure from Food Ingestion

Similar to the SIR, the food ingestion rate (FIR) by livestock and wildlife is expressed as a proportion of the DMIR. Because the proportion of the DMIR for soil ingestion has already been calculated, the FIR is simply the remaining proportion of the DMIR attributed to food intake. The FIR should be calculated using data from the species used in the DTED. The FIR is calculated as follows:

$$FIR_{1C} = DMIR_{1C} - SIR_{1C} \quad [3]$$

where: FIR_{1C} = the food ingestion rate for the species selected as the primary consumer (kg dw food/day)

If no DMIR information is available, then allometric equations (Nagy, 1987) should be used to estimate the FIR according to the following equations. Note: equations have been converted from g dw food/day to kg dw food/day.

For mammalian species, the allometric equation is:

$$F_M = 0.0687 \text{ x } (BW_{1C})^{0.822} \quad [4]$$

where: F_M = feeding rate of mammalian species (kg dw food/day)
BW_{1C} = mean body weight in kilograms (kg) of the species selected as the primary consumer

For avian species, the allometric equation is:

$$F_A = 0.0582 \text{ x } (BW_{1C})^{0.651} \quad [5]$$

where: F_A = feeding rate of avian species (kg dw food/day)

7.6.1.5 Determining Bioavailability

The bioavailability of soil-adsorbed contaminants in the species selected as the primary consumer should be estimated. Due to lack of specific information on the bioavailability of contaminants from ingested soil for livestock and terrestrial wildlife, a bioavailability factor of one is assumed. However, if information exists to support an alternative bioavailability factor (BF), it should be incorporated into the calculation of the guideline.

7.6.1.6 Determining Bioconcentration Factors

The development of soil quality guidelines for herbivore food ingestion requires that a concentration of a substance in soil be established that will not lead to adverse effects on receptors via the ingestion of forage material. One method, using a generic equation, is to extrapolate to a concentration in soil using soil-to-plants bioconcentration factors (BCFs). Bioconcentration factors are used to estimate the concentration of contaminants in biota directly contributed from the media directly. In essence, these factors estimate the quantity of a substance that is bioavailable and can be concentrated in biota.

Therefore,

$$\text{BCF} = \frac{\text{concentration in plants}}{\text{concentration in soil}} \qquad [6]$$

Using professional judgment, BCF values from soil to plants can be estimated with available literature data. BCF values can be highly variable, may be specific to particular plant species and soil types, and may vary with chemical concentration in soil. In general, field data should be given preference over laboratory data, and BCFs calculated based on chemical properties such as K_{ow} should be avoided. If more than one value is available in the literature and deemed appropriate for use, the geometric mean of the literature values should be used. In the absence of literature values, it may be possible to estimate a BCF based on chemical properties and values measured for similar contaminants (see, for example, US EPA, 1999).

7.6.1.7 Calculation of the Soil Quality Guideline for Ingestion – Primary Consumers

An animal may be exposed to a contaminant by more than one route (i.e. through contaminated food, direct soil ingestion, dermal exposure, inhalation of air and dust, and drinking water). The effects from the sum of these exposures should not exceed the DTED. Contributions from each medium (also called apportionment factors) to the total exposure for livestock and wildlife are discussed below.

Water

The U.S. EPA and Health Canada agree that 20% of a human's estimated daily intake (EDI) is contributed from water. It is assumed that the remaining 80% originates from soil, air, and food. Due to the lack of sufficient data to estimate an inter-species apportionment factor for water, an apportionment factor of 20% shall be used for livestock and wildlife. If apportionment information is available for the species in question, it should be used in place of the human value.

If contaminant-specific properties (e.g. hydrophobicity) indicate that the water ingestion pathway is not likely to be applicable, part or all of the 20% apportionment factor may be re-allocated to soil and food ingestion.

Dermal Absorption and Inhalation

Insufficient information exists to account for dermal absorption and inhalation of contaminants in livestock and wildlife. It is assumed that these two routes of exposure together do not contribute significantly to the total contaminant exposure to livestock and wildlife (i.e. approximately 5%). However, if contaminant-specific and/or species-specific information is available for dermal absorption and inhalation (e.g. volatiles), it will be used in place of the default 5% apportionment factor. Dermal absorption can be important for certain wildlife species, including some amphibians or burrowing mammals with exposed skin surface areas.

If information is available that shows the dermal exposure and inhalation pathways are not applicable, the 5% apportionment may be allocated to soil and food ingestion.

Ingestion

The proportion of the total exposure attributed to ingestion is the product of the dry matter intake rate (DMIR) and contaminant concentration in that medium. Assuming that water and dermal absorption/inhalation account for 25% of the total exposure then the remaining 75% of the total exposure is from food and soil ingestion. Walker and MacDonald (1992) recommended an apportionment factor of 75% in the calculation of tissue residue guidelines for the protection of wildlife consumers of aquatic life to account for contributions of food to the total exposure.

Therefore, in order to provide protection to the animal, maximum exposure from ingestion of soil and food combined should not exceed 75% of the DTED. The soil quality guideline for ingestion (SQG_{1C}) should ensure that ingestion of the soil and ingestion of plants growing on that soil do not result in the animal receiving more than 75% of the DTED:

(exposure from soil + exposure from food) = 0.75 x $DTED_{1C}$ [7]

For soil ingestion, the exposure is equal to the concentration in soil multiplied by the soil ingestion rate (SIR) and a bioavailability factor, divided by the body weight of the animal exposed:

$$\text{Exposure from soil ingestion} = \frac{SQG_{1C} \times SIR_{1C} \times BF}{BW_{1C}} \qquad [8]$$

Where

SQG_{1C}	=	soil quality guideline for soil and food ingestion for the primary consumer (mg/kg)
SIR_{1C}	=	soil ingestion rate (kg dw soil/day)
BF	=	bioavailability factor (unitless)
BW_{1C}	=	body weight (kg)

Likewise, for food ingestion, the exposure is equal to the concentration in food multiplied by the food ingestion rate (FIR) and a bioconcentration factor and divided by the body weight of the animal exposed:

$$\text{Exposure from food ingestion} = \frac{SQG_{1C} \times FIR_{1C} \times BCF_1}{BW_{1C}} \qquad [9]$$

Where

BCF_1 = bioconcentration factor (unitless)

Equations (8) and (9) can be incorporated into equation (7) to provide the following equation:

$$\frac{SQG_{1C} \times SIR_{1C} \times BF}{BW_{1C}} + \frac{SQG_{1C} \times FIR_{1C} \times BCF_1}{BW_{1C}} = 0.75 \times DTED_{1C} \qquad [10]$$

Finally, equation (10) can be rearranged to give the final equation for deriving a soil quality guideline for ingestion (SQG_{1C}) that will prevent primary consumers from being exposed to more than 75% of the DTED resulting from the ingestion of soil and plants:

$$SQG_{1C} = \frac{0.75 \times DTED_{1C} \times BW_{1C}}{(SIR_{1C} \times BF) + (FIR_{1C} \times BCF_1)} \qquad [11]$$

7.6.2 Derivation of Soil Quality Guideline for Soil and Food Ingestion for Protection of Secondary Consumers

For substances that have a strong tendency to bioaccumulate and/or biomagnify, food chain pathways can lead to the exposure of ecological receptors at higher trophic levels, i.e. secondary and tertiary consumers. The food chain leading to secondary consumers is more complex and involves three trophic levels. It can be represented by either one of the following pathways:

Soil → Plant → Prey (primary consumer) → Predator (secondary consumer)

Soil → Prey (e.g. earthworms) → Predator (secondary consumer)

The model developed to represent this food chain and to derive SQG_{2C} (the subscript $_{2C}$ stands for secondary consumer) is similar to the one use in deriving SGC_{1C}. However, to account for biomagnification from contaminated food and soil to the prey (through the soil → plant → prey or soil → prey pathways), a bioaccumulation factor from soil to prey (BAF_2) is used instead of BCF_1. To be conservative and protective of all secondary consumers species, the food chain pathway involving the prey with the highest soil to prey bioaccumulation factor should be used for guideline derivation.

The most sensitive secondary consumer should be selected on the basis of the species with the lowest effects dose to exposure ratio, using the guidelines presented in Section 7.6.1.1. However, the data set should focus on predatory mammals and avian species rather than herbivores.

7.6.2.1 Calculation of the Daily Threshold Effect Dose

The DTED is estimated using the lowest effects dose (ED_{2C},) from the most sensitive secondary consumer, divided by an uncertainty factor. The DTED is calculated according to the following equation.

$$DTED_{2C} = \text{lowest } ED_{2C} / UF \qquad [1]$$

where:

$DTED_{2C}$	=	daily threshold effects dose of the secondary consumer (mg/kg bw_{2C}-day)
ED_{2C}	=	lowest effects dose (mg/kg bw_{2C}-day)
UF	=	uncertainty factor (if needed)

An uncertainty factor between one and five can be applied using expert judgement. The following criteria are a guide for UF application:

1. The effects dose is considered to be "biologically significant" and not just statistically different from controls and therefore extrapolation below the LOEC is required.

2. The effects dose is taken from an acute lethal or sublethal study.

3. Only the absolute minimum data requirements have been met.

4. Fewer than three taxonomic groups are available to select the lowest effects dose.

An uncertainty factor greater than five for other levels of uncertainty (e.g. intra-species variation, extrapolation to field conditions) is not recommended since a large measure of conservatism is already added by selecting the species with the lowest available effects dose.

7.6.2.2 Determining Body Weight

The mean body weight for the species used to calculate the $DTED_{2C}$ is estimated and is required with the $DTED_{2C}$ as well as soil and food ingestion rates (SIR and FIR) to evaluate the soil quality guideline for ingestion (Section 7.6.2.6).

7.6.2.3 Determining Exposure from Soil and Food Ingestion

Exposure from Soil Ingestion

Animals ingest soil principally as a result of soil adhering to forage. The ingestion rate of soil and forage together is referred to as the dry matter intake rate (DMIR). To estimate the rate of soil ingested directly, the percentage of the DMIR attributed to soil ingestion must be isolated. In most soil-based exposure studies, the proportion of soil ingested (PSI), is reported with the DMIR. Therefore, an animal's soil ingestion rate (SIR) is calculated as a proportion of the DMIR according to the following equation:

$$SIR_{2C} = (DMIR_{2C} \text{ x } PSI_{2C}) \qquad [2]$$

where: SIR_{2C} = the soil ingestion rate of the secondary consumer (kg dw soil/day)
$DMIR_{2C}$ = geometric mean of the available dry matter intake rates for the secondary consumer (kg dw/day)
PSI_{2C} = geometric mean of available soil ingestion proportions reported with the DMIR

It is assumed that the DMIR contains only dry matter as food or soil; food consumption rates should therefore be adjusted based on the water content of the food. The geometric mean of available DMIR values is used based on the assumption that the data are lognormally distributed; if there is evidence of a different distribution, then an alternate estimate (e.g., arithmetic mean) can be used based on professional judgement. If no DMIR data are available for the species used in the DTED, employ the allometric equations (4 and 5) below. If no data are available to determine the PSI for the secondary consumer, default values of 0.077 for wildlife species or 0.135 for soil probing insectivores may be used (McMurter, 1993).

Exposure from Food Ingestion

Similar to the SIR, the food ingestion rate (FIR) by livestock and wildlife is expressed as a proportion of the DMIR. Because the proportion of the DMIR for soil ingestion has already been calculated, the FIR is simply the remaining proportion of the DMIR attributed to food intake. The FIR should be calculated using data from the species used in the DTED. The FIR is calculated as follows:

$$FIR_{2C} = DMIR_{2C} - SIR_{2C} \qquad [3]$$

where:
- FIR_{2C} = the food ingestion rate for the species used in the $DTED_{2C}$ (kg dw food/day)
- $DMIR_{2C}$ = geometric mean of the dry matter intake rate for the species used in the $DTED_{2C}$ (kg dw/day)
- SIR_{2C} = the soil ingestion rate (kg dw soil/day)

If no DMIR information is available, then allometric equations (Nagy, 1987) should be used to estimate the FIR according to the following equations. Note: equations have been converted from g dw food/day to kg dw food/day.

For mammalian species, the allometric equation is:

$$F_M = 0.0687 \times (BW_{2C})^{0.822} \qquad [4]$$

where:
- F_M = feeding rate of mammalian species (kg dw food/day)
- BW_{2C} = mean body weight in kilograms (kg) of the species used to derive the $DTED_{2C}$

For avian species, the allometric equation is:

$$F_A = 0.0582 \times (BW_{2C})^{0.651} \qquad [5]$$

where:
- F_A = feeding rate of avian species (kg dw food/day)
- BW_{2C} = mean body weight in kilograms (kg) of the livestock or wildlife species used to derive the $DTED_{2C}$

7.6.2.4 Determining Bioavailability

The bioavailability of soil-adsorbed contaminants in the species used to calculate the DTED should be estimated. Due to lack of specific information on the bioavailability of contaminants from ingested soil for livestock and terrestrial wildlife, a bioavailability factor of one is assumed.

However, if information exists to support an alternative bioavailability factor (BF), it should be incorporated into the calculation of the guideline.

7.6.2.5 Determining Bioaccumulation Factors

The development of soil quality guidelines for food ingestion by secondary consumers requires that a concentration of a substance in soil be established that will not lead to adverse effects to receptors from the ingestion of prey. One way of doing this, in a generic equation, is to extrapolate to a concentration in soil using soil-to-plant-to-prey or soil-to-prey bioaccumulation factors (BAFs).

Therefore,

$$BAF = \frac{\text{concentration in prey}}{\text{concentration in soil}} \qquad [6]$$

Using professional judgment, BAF values from soil to prey can be estimated with available literature data. In general, field data should be given preference over laboratory data. If more than one value is available in the literature and deemed appropriate for use, the geometric mean of the literature values should be used. In the absence of literature values, it may be possible to estimate a BAF based on chemical properties and values measured for similar contaminants (see, for example, US EPA, 1999).

7.6.2.6 Calculation of the Soil Quality Guideline for Ingestion – Secondary Consumers

As noted above for primary consumers, it is assumed that, in the absence of contaminant-specific and/or species-specific information, the consumption of drinking water and exposure via dermal absorption and inhalation contribute 20% and 5% respectively to the intake of a contaminant and that the remaining 75% of the total exposure is from food and soil ingestion.

Therefore, in order to provide protection to the animal, maximum exposure from ingestion of soil and food combined should not exceed 75% of the DTED. The soil quality guideline for ingestion (SQG_{2C}) should ensure that ingestion of the soil and ingestion of plants growing on that soil does not result in the animal receiving more than 75% of the DTED:

$$(\text{exposure from soil} + \text{exposure from food}) = 0.75 \times DTED_{2C} \qquad [7]$$

If the literature review or chemical properties (e.g. hydrophobicity) indicate that the water ingestion, dermal contact, and/or inhalation pathways are not applicable for a contaminant, then some or all of the exposure allocated to these pathways may be re-allocated to soil and food ingestion.

For soil ingestion, the exposure is equal to the concentration in soil multiplied by the soil ingestion rate (SIR) and a bioavailability factor and divided by the body weight of the animal exposed:

$$\text{Exposure from soil ingestion} = \frac{SQG_{2C} \times SIR_{2C} \times BF}{BW_{2C}} \qquad [8]$$

Where SQG_{2C} = soil quality guideline for soil and food ingestion for the secondary consumer(mg/kg)
SIR_{2C} = soil ingestion rate (kg dw soil/day)
BF = bioavailability factor (unitless)
BW_{2C} = body weight (kg)

Likewise, for food ingestion, the exposure is equal to the concentration in food multiplied by the food ingestion rate (FIR) and a bioaccumulation factor and divided by the body weight of the animal exposed:

$$\text{Exposure from food ingestion} = \frac{SQG_{2C} \times FIR_{2C} \times BAF_2}{BW_{2C}} \qquad [9]$$

Where SQG_{2C} = soil quality guideline for soil and food ingestion for the secondary consumer(mg/kg)
FIR_{2C} = food ingestion rate (kg dw food/day)
BAF_2 = bioaccumulation factor (unitless)
BW_{2C} = body weight (kg)

Equations (8) and (9) can be incorporated into equation (7) to provide the following equation:

$$\frac{SQG_{2C} \times SIR_{2C} \times BF}{BW_{2C}} + \frac{SQG_{2C} \times FIR_{2C} \times BAF_2}{BW_{2C}} = 0.75 \times DTED_{2C} \qquad [10]$$

Finally, equation (10) can be rearranged to give the final equation for deriving a soil quality guideline for ingestion (SQG_{2C}) that will prevent secondary consumers from being exposed to more than 75% of the DTED resulting from the ingestion of soil and plants:

$$SQG_{2C} = \frac{0.75 \times DTED_{2C} \times BW_{2C}}{(SIR_{2C} \times BF) + (FIR_{2C} \times BAF_2)} \qquad [11]$$

where, SQG_{2C} = soil quality guideline for soil and food ingestion for the secondary consumer (mg/kg dw soil)
$DTED_{2C}$ = daily threshold effects dose for the secondary consumer (mg/kg bw-day)
BW_{2C} = body weight of the species used in the $DTED_{2C}$ (kg)
SIR_{2C} = the soil ingestion rate for the species used in the $DTED_{2C}$ (kg dw soil/day)
BF = bioavailability factor (unitless)
FIR_{2C} = the food ingestion rate for the species used in the $DTED_{2C}$ (kg dw food/day)

BAF_2 = bioaccumulation factor (unitless)

In order to take into consideration the variable behaviour of predator species, this equation must be modified by adding an apportionment factor accounting for the proportion of the foraging range represented by the contaminated site (AF_{FR}) and an apportionment factor accounting for the time spent by the predator on the site (AF_Y). If uncertainty exists regarding these two factors, a value of 1 is recommended. Therefore, the equation becomes:

$$SQG_{2C} = \frac{0.75 \times DTED_{2C} \times BW_{2C}}{[(SIR_{2C} \times BF) + (FIR_{2C} \times BAF_2)] \times AF_{FR} \times AF_Y} \qquad [12]$$

7.6.3 Derivation of Soil Quality Guideline for Soil and Food Ingestion for Protection of Tertiary Consumers

The food chain leading to tertiary consumers also involves three trophic levels and is based on the following pathway:

Soil → Invertebrate → Prey (secondary consumer) → Predator (tertiary consumer)

The model developed to represent this food chain and to derive SQG_{3C} (the subscript $_{3C}$ stands for tertiary consumer) is similar to the one use in deriving SQG_{1C}. In this case, the bioaccumulation factor from soil to secondary consumer (BAF_3) is used instead of BCF_1. To be conservative and protective of all secondary consumers species, the food chain pathway involving the prey with the highest soil to prey bioaccumulation factor should be used for guideline derivation.

The most sensitive tertiary consumer should be selected on the basis of the species with the lowest effects dose to exposure ratio, using the guidelines presented in Section 7.6.1.1. As for secondary consumers, the data set should focus on predatory mammals and avian species.

7.6.3.1 Calculation of the Daily Threshold Effect Dose

The DTED is estimated using the lowest effects dose (ED_{3C},) from the most sensitive tertiary consumer, divided by an uncertainty factor. The DTED is calculated according to the following equation.

$$DTED_{3C} = \text{lowest } ED_{3C} / UF \qquad [1]$$

where: $DTED_{3C}$ = daily threshold effects dose of the tertiary consumer (mg/kg bw_{3C}-day)
ED_{3C} = lowest effects dose (mg/kg bw_{3C}-day)
UF = uncertainty factor (if needed)

An uncertainty factor between one and five can be applied using expert judgement and the guidance given in Section 7.6.1.2.

7.6.3.2 Determining Body Weight

The mean body weight for the species used to calculate the $DTED_{3C}$ is estimated and is required with the $DTED_{3C}$ as well as soil and food ingestion rates (SIR and FIR) to evaluate the soil quality guideline for ingestion (Section 7.6.3.6).

7.6.3.3 Determining Exposure from Soil and Food Ingestion

Exposure from Soil Ingestion

Animals ingest soil principally as a result of soil adhering to forage. The ingestion rate of soil and forage together is referred to as the dry matter intake rate (DMIR). To estimate the rate of soil ingested directly, the percentage of the DMIR attributed to soil ingestion must be isolated. In most soil-based exposure studies, the proportion of soil ingested (PSI), is reported with the DMIR. Therefore, an animal's soil ingestion rate (SIR) is calculated as a proportion of the DMIR according to the following equation:

$$SIR_{3C} = (DMIR_{3C} \times PSI_{3C}) \qquad [2]$$

where:
- SIR_{3C} = the soil ingestion rate of the tertiary consumer (kg dw soil/day)
- $DMIR_{3C}$ = geometric mean of the available dry matter intake rates for the tertiary consumer (kg dw/day)
- PSI_{3C} = geometric mean of available soil ingestion proportions reported with the DMIR

It is assumed that the DMIR contains only dry matter as food or soil; food consumption rates should therefore be adjusted based on the water content of the food. The geometric mean of available DMIR values is used based on the assumption that the data are lognormally distributed; if there is evidence of a different distribution, then an alternate estimate (e.g., arithmetic mean) can be used based on professional judgement. If no DMIR data are available for the species used in the DTED, employ the allometric equations (4 and 5) below. If no data are available to determine the PSI for the tertiary consumer, default values of 0.077 for wildlife species or 0.135 for soil probing insectivores may be used (McMurter, 1993). For consumers preying solely on mammals or birds, the soil ingestion proportion may be negligible and thus PSI may be zero (Beyer et al., 1994).

Exposure from Food Ingestion

Similar to the SIR, the food ingestion rate (FIR) by livestock and wildlife is expressed as a proportion of the DMIR. Because the proportion of the DMIR for soil ingestion has already been calculated, the FIR is simply the remaining proportion of the DMIR attributed to food intake. The FIR should be calculated using data from the species used in the DTED. The FIR is calculated as follows:

$$FIR_{3C} = DMIR_{3C} - SIR_{3C} \qquad [3]$$

where: FIR_{3C} = the food ingestion rate for the species used in the $DTED_{3C}$ (kg dw food/day)
$DMIR_{3C}$ = geometric mean of the dry matter intake rate for the species used in the $DTED_{3C}$ (kg dw/day)
SIR_{3C} = the soil ingestion rate (kg dw soil/day)

If no DMIR information is available, then allometric equations (Nagy, 1987) should be used to estimate the FIR according to the following equations. Note: equations have been converted from g dw food/day to kg dw food/day.

For mammalian species, the allometric equation is:

$$F_M = 0.0687 \times (BW_{3C})^{0.822} \quad [4]$$

where: F_M = feeding rate of mammalian species (kg dw food/day)
BW_{3C} = mean body weight in kilograms (kg) of the species used to derive the $DTED_{3C}$

For avian species, the allometric equation is:

$$F_A = 0.0582 \times (BW_{3C})^{0.651} \quad [5]$$

where: F_A = feeding rate of avian species (kg dw food/day)
BW_{3C} = mean body weight in kilograms (kg) of the livestock or wildlife species used to derive the $DTED_{3C}$

7.6.3.4 Determining Bioavailability

The bioavailability of soil-adsorbed contaminants in the species used to calculate the DTED should be estimated. Due to lack of specific information on the bioavailability of contaminants from ingested soil for livestock and terrestrial wildlife, a bioavailability factor of one is assumed. However, if information exists to support an alternative bioavailability factor (BF), it should be incorporated into the calculation of the guideline.

7.6.3.5 Determining Bioaccumulation Factors

The development of soil quality guidelines for food ingestion by tertiary consumers requires that a concentration of a substance in soil be established that will not lead to adverse effects to receptors from the ingestion of prey. One way of doing this, in a generic equation, is to extrapolate to a concentration in soil using soil-to-prey or soil-to-invertebrate-to-prey bioaccumulation factors (BAFs).

Therefore,

$$BAF = \frac{\text{concentration in prey}}{\text{concentration in soil}} \quad [6]$$

Using professional judgment, BAF values from soil to prey can be estimated with available literature data. In general, field data should be given preference over laboratory data. If more than one value is available in the literature and deemed appropriate for use, the geometric mean of the literature values should be used. In the absence of literature values, it may be possible to estimate a BAF based on chemical properties and values measured for similar contaminants (see, for example, US EPA, 1999).

7.6.3.6 Calculation of the Soil Quality Guideline for Ingestion – Tertiary Consumers

As noted above for primary consumers, it is assumed that, in the absence of contaminant-specific and/or species-specific information, the consumption of drinking water and exposure via dermal absorption and inhalation contribute 20% and 5% respectively to the intake of a contaminant and that the remaining 75% of the total exposure is from food and soil ingestion.

Therefore, in order to provide protection to the animal, maximum exposure from ingestion of soil and food combined should not exceed 75% of the DTED. The soil quality guideline for ingestion (SQG_{3C}) should ensure that ingestion of the soil and ingestion of plants growing on that soil does not result in the animal receiving more than 75% of the DTED:

(exposure from soil + exposure from food) = 0.75 x $DTED_{3C}$ [7]

If the literature review or chemical properties (e.g. hydrophobicity) indicate that the water ingestion, dermal contact, and/or inhalation pathways are not applicable for a contaminant, then some or all of the exposure allocated to these pathways may be re-allocated to soil and food ingestion.

For soil ingestion, the exposure is equal to the concentration in soil multiplied by the soil ingestion rate (SIR) and a bioavailability factor and divided by the body weight of the animal exposed:

$$\text{Exposure from soil ingestion} = \frac{SQG_{3C} \times SIR_{3C} \times BF}{BW_{3C}} \qquad [8]$$

Where
SQG_{3C} = soil quality guideline for soil and food ingestion for the secondary consumer(mg kg^{-1})
SIR_{3C} = soil ingestion rate (kg dw soil/day)
BF = bioavailability factor (unitless)
BW_{3C} = body weight (kg)

Likewise, for food ingestion, the exposure is equal to the concentration in food multiplied by the food ingestion rate (FIR) and a bioaccumulation factor and divided by the body weight of the animal exposed:

Exposure from food ingestion = $\dfrac{SQG_{3C} \times FIR_{3C} \times BAF_3}{BW_{3C}}$ [9]

Where SQG_{3C} = soil quality guideline for soil and food ingestion for the tertiary consumer(mg kg^{-1})
FIR_{3C} = food ingestion rate (kg dw food/day)
BAF_3 = bioaccumulation factor (unitless)
BW_{3C} = body weight (kg)

Equations (8) and (9) can be incorporated into equation (7) to provide the following equation:

$$\frac{SQG_{3C} \times SIR_{3C} \times BF}{BW_{3C}} + \frac{SQG_{3C} \times FIR_{3C} \times BAF_3}{BW_{3C}} = 0.75 \times DTED_{3C} \quad [10]$$

Finally, equation (10) can be rearranged to give the final equation for deriving a soil quality guideline for ingestion (SQG_{3C}) that will prevent tertiary consumers from being exposed to more than 75% of the DTED resulting from the ingestion of soil and plants:

$$SQG_{3C} = \frac{0.75 \times DTED_{3C} \times BW_{3C}}{(SIR_{3C} \times BF) + (FIR_{3C} \times BAF_3)} \quad [11]$$

where, SQG_{3C} = soil quality guideline for soil and food ingestion for the tertiary consumer (mg/kg dw soil)
$DTED_{3C}$ = daily threshold effects dose for the tertiary consumer (mg/kg bw-day)
BW_{3C} = body weight of the species used in the $DTED_{3C}$ (kg)
SIR_{3C} = the soil ingestion rate for the species used in the $DTED_{3C}$ (kg dw soil/day)
BF = bioavailability factor (unitless)
FIR_{3C} = the food ingestion rate for the species used in the $DTED_{3C}$ (kg dw food/day)
BAF_3 = bioaccumulation factor (unitless)

In order to take into consideration the variable behaviour of predator species, this equation must be modified by adding an apportionment factor accounting for the proportion of the foraging range represented by the contaminated site (AF_{FR}) and an apportionment factor accounting for the time spent by the predator on the site (AF_Y). If uncertainty exists regarding these two factors, a value of 1 is recommended. Therefore, the equation becomes:

$$SQG_{3C} = \frac{0.75 \times DTED_{3C} \times BW_{3C}}{[(SIR_{3C} \times BF) + (FIR_{3C} \times BAF_3)] \times AF_{FR} \times AF_Y} \quad [12]$$

7.6.4 Calculation of Final Soil Guideline for Soil and Food Ingestion

For those contaminants for which only the primary consumer is considered, the final soil and food ingestion guideline, SQG_I, is the value calculated for the protection of the primary consumer. For substances where secondary and tertiary consumers are considered, the final soil and food ingestion guideline, SQG_I, is the lowest of the values calculated for the primary, secondary and tertiary consumers.

7.7 Using Microbial (Nutrient and Energy Cycling) Data for Derivation of Soil Contact Guidelines

Soil processes such as decomposition, respiration and organic nutrient cycles are important components of the ecological function of soil. These processes may be affected by the presence of contaminants, and therefore should be considered in the development of soil quality guidelines.

Appendix B outlines the procedures for determining the soil quality guideline for the protection of nutrient and energy cycling (SQG_{NEC}). Since data are expected to be limited for this pathway, the SQG_{NEC} is incorporated as a check mechanism; professional judgement should be used to decide whether the SQG_{NEC} is applied when determining the SQG_E.

7.8 Derivation of Soil Quality Guidelines for the Protection of Freshwater Life

Contamination present in soil can migrate to groundwater. If there are surface water bodies (streams, rivers, lakes, etc.) nearby, then aquatic life in these surface water bodies may be affected by the contamination, particularly if there is a permeable aquifer connecting the contamination with the surface water body. The soil quality guideline for the protection of freshwater life (SQG_{FL}) is calculated using a model adapted from that developed by the British Columbia Contaminated Sites Soil Task Group (CSST).

The model includes four components:

1. Partitioning of contamination from soil to pore water (see Appendix A).

2. Migration of contamination through the unsaturated zone to the groundwater surface (for generic guideline development, contamination is assumed to be in contact with the groundwater, so this component does not have an effect).

3. Dilution and mixing of the contamination in the groundwater aquifer.

4. Transport of the contamination through the saturated zone to the receptor.

The model is described in detail in Appendix C. This pathway is evaluated for soluble organic contaminants only on a generic basis. Inorganic contaminants may also affect nearby surface water quality; however, the partitioning and groundwater transport of inorganic compounds is complex and highly site-specific. Therefore, a generic SQG_{FL} is not calculated for inorganic compounds at this time. Potential impacts to freshwater life by inorganic compounds are evaluated on a site-specific basis.

The SQG_{FL} is calculated by setting the allowable receptor groundwater concentration in the model equal to the freshwater life (FL) guideline from the Canadian Water Quality Guidelines (CCME, 1999). If a FL guideline has not been published, then calculation of the SQG_{FL} is not required.

For purposes of developing generic guidelines, it is assumed that the surface water body is located 10 m away from the contaminated soil. While it is recognized that groundwater may be diluted within an initial mixing zone once it reaches a surface water body, a dilution factor for mixing in the surface water body is not applied for generic guideline development due to the site-specific nature of this process and the variance in policy decisions across Canada regarding dilution within the receiving environment.

The SQG_{FL} is independent of the land use classification, and may be excluded on a site-specific basis if there are no surface water bodies in the vicinity of the site.

7.9 Derivation of Soil Quality Guidelines for the Protection of Livestock Watering and Irrigation Water

Contamination that migrates to groundwater may affect the water quality in dugouts or water wells used for livestock watering or crop irrigation. These pathways apply only for the agricultural land use.

Determination of the soil quality guidelines for the protection of livestock watering (SQG_{LW}) and irrigation (SQG_{IR}) involves the application of the same groundwater model as for the SQG_{FL} (Section 7.7; model described in Appendix C); however, transport through the saturated zone is not considered (i.e. it is assumed that dugouts or wells could be installed within the contaminated area). As for the SQG_{FL} above, generic guidelines are not calculated for inorganic substances at this time; these are evaluated on a site-specific basis.

The guidelines are calculated by setting the allowable receptor groundwater concentration in the model equal to the livestock water (for the SQG_{LW}) and irrigation water (for the SQG_{IR}) guidelines from the Canadian Water Quality Guidelines (CCME, 1999). If a livestock water guideline is unavailable, a livestock water threshold value can be developed using the following equation:

$$LWT = \frac{DTED \times BW}{WIR}$$

where:

LWT = calculated livestock water threshold value
DTED = DTED for livestock (mg/kg-bw/d) – see Section 7.6.1.2
BW = livestock body weight (kg) = 550 kg for cattle (CCME,

2000)

WIR = livestock water ingestion rate (L/d) = 100 L/d for cattle (CCME, 2000)

If the calculated livestock water threshold value is higher than the pure-phase solubility of the contaminant, then calculation of the SQG_{LW} is not required. If an irrigation water guideline is not available, then calculation of the SQG_{IR} is not required.

7.10 Derivation of Environmental Soil Quality Guidelines for Offsite Migration

In deriving soil quality guidelines for commercial and industrial sites, SQGTG uses an exposure scenario which considers contact of ecological receptors with on-site soil only. However, wind and water erosion of soil and subsequent deposition can transfer contaminated soil from one site to another.

Therefore, SQGTG has developed a model to address the subsequent movement of soil from a commercial or industrial site to adjacent, more sensitive land (e.g., agricultural property). This procedure is briefly described below. The full details and assumptions of the model are provided in Appendix G. SQGTG acknowledges the imprecise nature of this model, the uncertainty surrounding the underlying assumptions, and use of scientific judgement in determining input parameters; therefore this pathway is considered to be a check mechanism, and professional judgement should be used to determine whether the SQG_E should be modified by this pathway. This check mechanism is not applied for volatile organic compounds, which are not expected to be associated with soil particles transported by wind and water.

Calculations using the Universal Soil Loss Equation and the Wind Erosion Equation are used to estimate the transfer of soil to an adjacent property. It is possible to calculate the concentration in eroded soil from the commercial or industrial site that will raise the contaminant concentration in the receiving soil to equal the agricultural guideline within a specified period of time; this concentration is applied as the soil quality guideline for off-site migration (SQG_{OM-E}). At specific commercial or industrial sites, management actions may be taken to prevent or limit erosive losses of surface soils. Accommodation for such situations is provided in the guidance for the development of site-specific objectives.

7.11 Consideration of Additional Exposure Pathways

It is anticipated that in most situations, the exposure pathways described above will sufficient for the development of environmental quality guidelines. However, other exposure pathways exist, such as dermal contact of wildlife with contaminated water. If the literature review indicates that another exposure pathway may be of particular concern, then this pathway should be evaluated. Specific guidance on the evaluation of additional exposure pathways is not provided herein at this time; where possible, methods published by regulatory agencies such as Environment Canada or US EPA should be applied.

For contaminants where the SQG_I is low relative to the other environmental quality guidelines, the soil and food ingestion pathway should be considered for commercial and industrial land uses. To reflect the lower amount of time wildlife would be expected to spend at commercial and industrial sites, the SQG_I can be multiplied by 5 for these land uses (i.e. wildlife are assumed to spend 20% of their time at the site). The SQG_I may also be applied for other chemicals at commercial and industrial sites on a site-specific or jurisdictional basis.

Professional judgement should be used to determine whether these exposure pathways are used to modify the final environmental soil quality guideline.

SECTION 8
DERIVATION OF THE FINAL ENVIRONMENTAL SOIL QUALITY GUIDELINES

The protocol defines three types of exposure pathway: required pathways, applicable pathways, and check mechanisms.

- Required pathways must be calculated, and are included in the derivation of the overall environmental soil quality guideline (SQG_E). If insufficient data exist to calculate a required pathway, then the SQG_E cannot be calculated.
- Applicable pathways must be calculated if sufficient data are available, and, if calculated, are included in the derivation of the overall SQG_E. However, even if insufficient data exist to calculate an applicable pathway, the SQG_E can still be calculated.
- Check values must be calculated if sufficient data are available, and, if calculated, may or may not be included in the calculation of the overall SQG_E, based on professional judgement. If insufficient data exist to calculate a check value, the SQG_E can still be calculated.

Exposure pathways to be evaluated for the SQG_E for each land use and chemical type are shown in Table 2.

Table 2. Exposure Pathways for Development of the SQG_E

Pathway	Agricultural	Residential/ Parkland	Commercial	Industrial
- Soil Contact (SQG_{SC})	All[a]	All[a]	All[a]	All[a]
- Soil Ingestion: 1° consumers (SQG_{1C})	All[c]	Biomagnifying[c]	None	None
- Soil Ingestion: 2° and 3° consumers (SQG_{2C}, SQG_{3C})	Biomagnifying[c]	Biomagnifying[c]	None	None
- Nutrient and Energy Cycling (SQG_{NEC})	All[b]	All[b]	All[b]	All[b]
- Groundwater: Freshwater Life (SQG_{FL})	Soluble	Soluble	Soluble	Soluble
- Groundwater: Agricultural (Irrigation - SQG_{IR}, Livestock Watering - SQG_{LW})	Soluble	None	None	None
- Offsite migration (SQG_{OM-E})	None	None	Non-volatile[b]	Non-volatile[b]

a – this pathway is required (i.e. a final guideline cannot be developed without it)
b – this pathway is considered to be a check mechanism
c – required pathway if the contaminant biomagnifies

8.1 Agricultural Land Use

The SQG_E for each soil type (coarse-grained and fine-grained) is the lowest of the values calculated for all exposure pathways applicable for the contaminant (i.e. the lowest of the SQG_{SC}, SQG_I, SQG_{NEC}, SQG_{FL}, SQG_{LW}, and SQG_{IR}). If there are insufficient data to calculate all of the applicable pathways, the SQG_E can still be determined so long as the SQG_{SC} has been calculated; if the substance is known to biomagnify, the SQG_I is also required.

If data are not available to derive an SQG_{SC} (or, for substances that biomagnify, the soil ingestion guidelines), then no SQG_E shall be set, since it is assumed that these guidelines do not represent exposures by the most likely critical pathways. In this situation, data gaps will be identified for further research. It may still be possible to develop a provisional SQG_E (see Part D, Section 1.4).

8.2 Residential/Parkland Land Use

For contaminants which do not bioaccumulate and/or biomagnify, the lowest of the SQG_{SC}, SQG_{NEC}, and SQG_{FL} for each soil type is used as the SQG_E for residential/parkland land use. For contaminants which bioaccumulate or biomagnify, the lowest of the SQG_{SC}, SQG_{NEC}, SQG_{FL} and SQG_I is used as the SQG_E. If there are insufficient data to calculate all of the applicable pathways, the SQG_E can still be determined so long as the SQG_{SC} has been calculated; if the substance is known to biomagnify, the SQG_I is also required. If no guideline can be set, then data gaps will be identified for further research. It may still be possible to develop a provisional SQG_E (see Part D, Section 1.4).

8.3 Commercial and Industrial Land Use

The lowest of the SQG_{SC}, SQG_{NEC} and the SQG_{FL} for each soil type is used as the SQG_E for commercial and industrial land use. The guideline may also be modified by the SQG_{OM-E}. If there are insufficient data to calculate all of the applicable pathways, the SQG_E can still be determined so long as the SQG_{SC} has been calculated. If no guideline can be set, then data gaps will be identified for further research. It may still be possible to develop a provisional SQG_E (see Part D, Section 1.4).

SECTION 1
DERIVATION OF HUMAN HEALTH SOIL QUALITY GUIDELINES

1.1 Introduction

The derivation of human health soil quality guidelines involves:

- assessing the toxicological hazard or risk posed by a chemical;

- determining estimated daily intake (EDI) of that chemical, unrelated to any specific contaminated site (i.e., "background" exposure);

- defining generic exposure scenarios for each land use;

- integrating exposure and toxicity information to set soil quality guidelines. These guidelines must ensure that total exposure to a contaminant (EDI and on-site exposure) will present no appreciable human health risk.

The CCME SQGTG submits that the steps employed to derive soil remediation guidelines are similar to those used for site-specific risk assessment, and recognizes that the process used to derive Environmental Quality Guidelines is subject to several sources of uncertainty (Section 1.3). Application of risk assessment methodology to establish numerical soil quality guidelines requires that several basic assumptions be made in lieu of site-specific information. Assumptions about the nature of chemical exposure for specified land uses are documented in Section 4.3. A defined exposure scenario was therefore chosen which details a sensitive receptor (toddler or adult) for a specified land use, the reference characteristics of that receptor (weight, amount of soil ingested/day, amount of water ingested/day, and other reference values, including those related to exposure duration) and specific pathways of exposure.

The derivation of human health soil quality guidelines includes the evaluation of several exposure pathways for each land use scenario, as well as the incorporation of “check mechanisms” to assess additional exposure pathways which may only be present at a limited number of sites or which may be subject to considerable uncertainty. Exposure pathways are evaluated using mathematical models. The input values for the models depend on the assumptions for each generic land use scenario, and include the choice of sensitive human receptor, exposure duration, frequency, and intensity. Evaluation of indirect pathways also requires input values representing physical characteristics of the site, which are affected by the soil type classification. Simplified models were deliberately chosen to represent the indirect pathways to limit the number of assumed input parameters; at a site-specific level, more complex models, along with detailed site-specific information, can provide more precise modelling results.

The potential exposure pathways considered in the development of human health soil quality guidelines, along with the contaminant classes for which they are evaluated (see Section 2.4 in Part A), are:

- Direct exposure (soil ingestion, dermal contact with soil and inhalation of soil particulates) – applies to all contaminant classes (though inhalation of soil particulate may be eliminated for volatile compounds).
- Migration of soil contaminants into groundwater used for potable water – applies to all soluble organic contaminants.
- Volatilization of soil contaminants into indoor air – applies to volatile contaminants.

In addition to these pathways, two "check mechanisms" are assessed:

- Exposure from ingestion of food grown on contaminated soils – applies to all contaminant classes, and is treated as a required or primary pathway for substances which biomagnify.
- The off-site migration via wind and water erosion of contaminants from commercial or industrial sites to more sensitive neighbouring properties – applies to all non-volatile contaminants.

Due to the necessarily imprecise nature of the models used to evaluate these mechanisms, the above check mechanisms are considered to be "Management Adjustment Factors" (MAFs), and may or may not adjust a generic guideline value, based on professional judgement.

Generic soil quality guidelines need to protect all normal activities associated with a particular land use. Normal activities entail exposure to all environmental media. Indirect pathways and MAFs record and respond to documented "secondary" exposures caused by redistribution of soil contamination to these interconnected media. The final generic soil quality guideline is the lowest of the values calculated for the direct and indirect soil exposure pathways. The check mechanisms are used to evaluate whether additional potential exposure routes may result in significant exposure to contaminants.

The indirect pathways and check mechanisms add a level of protectiveness to the generic guidelines which permits their use at a very broad range of sites within a land use category, but which may not be required or applicable to every site. The flexibility necessary to respond to site-specific conditions is available during the development of a site-specific objective, which may be based on guidelines developed under this protocol or through risk assessment. When site-specific objectives are developed from guidelines, most of the available flexibility is based on adjustment of indirect pathways and check mechanisms to reflect site conditions and to increase precision.

The development of site-specific objectives via limited modification of the generic guidelines, or the development of objectives using risk assessment, permits the flexibility required to remove or to add pathways or to use site-specific models to develop more accurate values.

1.2 Guiding Principles for Establishing Human Health Soil Quality Guidelines

The guiding principles (listed below) for the derivation of generic soil quality guidelines protective of human health reflect the principles adopted by CCME for contaminated sites.

1. There should be no appreciable risk to humans from a contaminated site. For each specified land use, there should be no restrictions as to the extent or nature of the interaction with the site. All activities normally associated with the intended land use should be free of any appreciable health risk.

2. Guidelines are based on defined, representative situations. Deriving numerical guidelines necessitates defining specific scenarios within which the exposure likely to arise on the site can be predicted with some degree of certainty.

3. Guidelines are derived by considering exposure through all relevant pathways. The total exposure from soil, air, water, food and consumer products is considered in the development of guidelines.

4. A critical human receptor is identified for each land use. To ensure that the guidelines do not limit the application of a site within the intended land use category, the defined exposure scenarios are usually based on the most sensitive receptor to the chemical, and the most critical health effect.

5. Guidelines should be reasonable, workable and usable. Guidelines are developed by applying scientifically derived information, backed by professional judgement where data gaps occur. Occasionally, defined exposure-based procedures produce numerical guidelines either far below background levels of contamination occurring naturally in the soil, or below practical quantitation limits. When this occurs, guidelines cannot be below background levels, and provisional guidelines should be established based on background soil concentrations.

1.3 Uncertainty in Guidelines Derivation

The uncertainties in assigning the relative exposures from different sources of a pollutant are numerous, but they may be considered under five broad headings:

- geographical,
- temporal,
- toxicokinetic,
- analytical, and
- philosophical or sociological (Park and Holliday, 1989).

Geographical uncertainties include:

- national and regional differences,
- the difference between urban and rural environments,
- local proximity to pollution sources, and
- individual lifestyles.

Temporal uncertainties include:

- the effects of changing measurement techniques, and
- uncertainties arising from using earlier data when pollution controls were less strict than today.

Toxicokinetic uncertainties include:

- possible differences in both intake and uptake,
- toxic effects associated with different routes of exposure, and
- different chemical forms existing in different media for some substances.

Analytical uncertainties include:

- the inevitable errors in measurement,
- the limitations of different techniques, and
- the representativeness of the samples analyzed.

Philosophical or sociological uncertainties include:

- questions about the nature and purpose of guidelines, and
- how far society might go to safeguard groups particularly at risk.

SECTION 2
INVESTIGATION OF CONTAMINANT TOXICOLOGY

Health Canada has determined the reference dose {Tolerable Daily Intake (TDI) for threshold substances and Risk Specific Doses (RSD) associated with risks of 10^{-4}, 10^{-5}, 10^{-6} and 10^{-7} for non-threshold substances} for a variety of contaminants. Health Canada has also established the reference concentration or Tolerable Concentration (TC) for volatile threshold substances and Risk-Specific Concentrations (RSC) for volatile non-threshold substances (e.g. Health Canada, 2003a). Some important aspects of developing TDIs and RSDs are described in Section 2.

Hazard assessment determines the health effect potentially attributable to a contaminant (e.g., carcinogenic, hepatotoxic, teratogenic) and estimates the reference dose believed to be associated with a defined level of incidence of that effect in the population. For a threshold substance, exposure less than the TDI should result in no adverse health effects in the population. For a non-threshold substance (i.e., a carcinogen or a germ cell mutagen), the critical risk specific dose may define a risk level of 1 in 1 million. Methods of chemical evaluation employed by Environmental Health Assessment Services, Safe Environments Program, Health Canada, are described in Richardson and Myers (1993), Armstrong and Newhook (1992), Meek (1989), Environmental Health Directorate (1989, 1990, 1991) and Health Canada (1994). The toxicological evaluation and exposure assessment can typically fit within a generalized framework for human health risk assessment (see Figure 14).

Toxic effects from exposure to environmental contaminants may be classified in the following broad categories:

- organ-specific,
- neurological/behavioural,
- reproductive/developmental,
- immunological, carcinogenic, and mutagenic.

These effects can be manifested at the biochemical, cellular, histopathological, and morphological levels. Effects vary, depending upon the dosage, route of exposure (e.g., ingestion, inhalation, or dermal contact), frequency and/or duration of exposure, species (and strain in the case of some animals), physiological state, sex, and age of the exposed population. Toxicological effects from exposure to chemical substances may be brief or prolonged, reversible or irreversible, immediate or delayed. The nature, number, severity, incidence and/or prevalence of specific toxicological effects in populations (of either humans or animal species) exposed to various chemicals generally increase with increasing dose or level of exposure; this is commonly referred to as the exposure- or dose-response relationship.

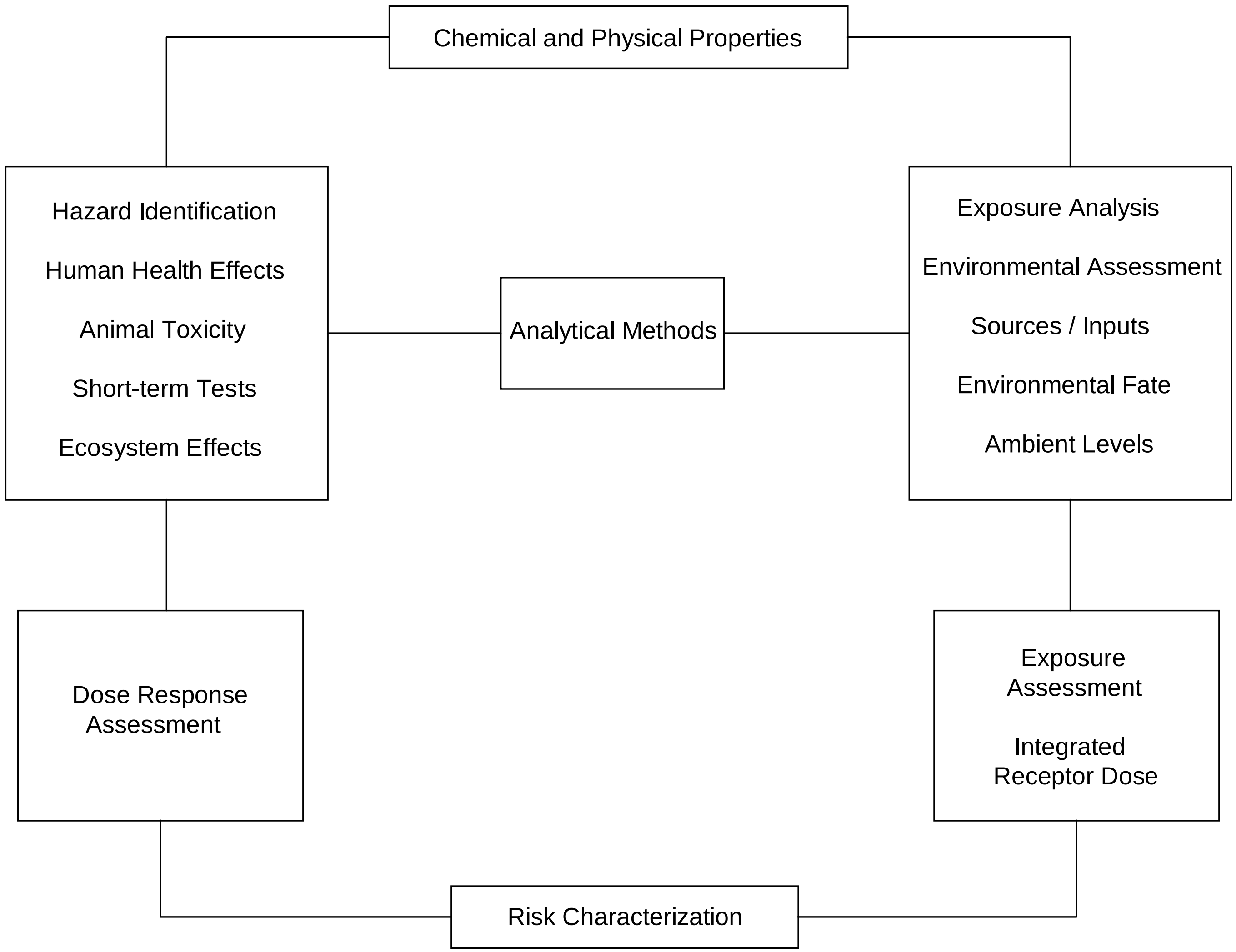


Figure 14: Generalized Framework for Human Health Risk Assessment

For most environmental contaminants, data on the toxicological effects resulting from exposure are restricted to information obtained from studies involving experimental animals. Occasionally, information derived from studies of human populations (principally epidemiological investigations) forms an integral part of the database upon which the assessment is based. Clearly, data on direct effects in humans are preferred for the toxicological assessment, but since data are limited or inadequate, extrapolation across species remains the rule.

Environmental contaminants are classified according to their potential carcinogenicity and mutagenicity to humans. This is based on the quantity, quality and nature of the available toxicological and epidemiological studies. The weight of evidence guidelines by which substances are classified by Health Canada for carcinogenicity and mutagenicity are outlined in the Sections 2.1 and 2.2.

2.1 Non-threshold Contaminants

For contaminants where the critical effect is assumed to have no threshold (i.e., currently restricted to mutagenesis and genotoxic carcinogenesis), it is assumed that there is some probability of harm to human health at any level of exposure. Consequently, it is not possible to determine a dose below which adverse effects do not occur.

For non-threshold substances, mathematical models are often used to extrapolate data on the exposure- or dose-response relationship derived from experimental studies in animal species or epidemiological studies (generally in workers) to estimate the cancer risk for concentrations to which the general population is exposed. There are numerous uncertainties in this approach, which generally involves linear extrapolation of results over several orders of magnitude, often in the absence of relevant data on mechanisms of tumour induction or differences in toxico--kinetics and toxico-dynamics between the relevant experimental animal species and humans.

Wherever possible, and if considered appropriate by Health Canada, information on pharmacokinetics, metabolism, and mechanisms of carcinogenicity and mutagenicity are incorporated into the quantitative estimates of potency derived, particularly from studies in animals (to provide relevant scaling of potency for human populations).

For the toxicological assessment of contaminants, SQGTG cannot specify a single concentration or dose considered to pose a "*de minimis*" level of risk (such as a lifetime cancer risk of 1 in 1 million). Such a judgement concerning negligible risk requires consideration of both social and scientific concerns about what level constitutes "de minimis" risk. There is no single "correct" value which adequately characterizes for all situations "de minimis" risk associated with a concentration or dose below which risks are acceptable and above which they are not. Rather, the risk at low doses or concentrations is assumed to be a continuum, with reduction of exposure leading to an incremental reduction of risk, and increases in exposure leading to incremental increases in risk.

It is recognised, however, that the incremental risks associated with exposure to low levels of such substances may be sufficiently small to be essentially negligible compared with other risks encountered in society. Because generic guidelines should attempt to accommodate a broad range of exposure scenarios, a large exposed population has been assumed.

CCME agrees in principle with the philosophy of Health Canada that human exposure to non-threshold toxicants should be reduced to the lowest levels deemed reasonably feasible.

This philosophy is consistent with the CCME philosophy to encourage remediation to the lowest reasonable levels. To derive numerical, generic environmental quality guidelines for soil, the SQGTG has adopted the position that contaminated site related risks arising from human exposure to non-threshold agents should be at least remediated to levels within the range of 10^{-4} to 10^{-7}. The guidelines derived based on this protocol reflect incremental risk levels from soil of both 10^{-5} and 10^{-6}, since these are the target incremental risk levels specified by most Canadian jurisdictions.

2.1.1 Guidelines for Classification of Carcinogenicity and of Mutagenicity in Germ Cells

Chemicals are classified into six categories on the basis of the following guidelines outlined by Health Canada (1994).

Group I: Carcinogenic to Humans/Human Germ Cell Mutagen

Group II: Probably Carcinogenic to Humans/Probable Human Germ Cell Mutagen

Group III: Possibly Carcinogenic to Human/Possible Human Germ Cell Mutagen

Group IV: Unlikely to be Carcinogenic to Humans/Unlikely to be Human Germ Cell Mutagen

Group V: Probably Not Carcinogenic to Humans/Probably Not a Human Germ Cell Mutagen

Group VI: Unclassifiable with Respect to Carcinogenicity in Humans/Unclassifiable with Respect to Germ Cell Mutagenicity in Humans

Health Canada considers those substances classified into Groups I and II as carcinogens having no threshold dose for effects. The risks from chemicals in groups III to VI are assessed on toxicological data relating to effects where some threshold is assumed to exist, and below which no risk exists.

It should be noted that for chemicals that are considered carcinogenic, there may also be toxicity data available for non-carcinogenic endpoints. In these cases, it may be necessary to evaluate both carcinogenic and non-carcinogenic endpoints separately, as their relative sensitivity may vary with different exposure pathways.

2.2 Threshold Contaminants

The approach to the assessment of substances classified in Groups III to VI, based on the above guidelines, is that adopted for "threshold toxicants" described in this section. However, for at least one of these categories (Group VI), adopting this approach is due to the lack of reliable data on carcinogenicity/mutagenicity, rather than certain knowledge that the substance is not carcinogenic. Though this may appear to be less than conservative, tolerable daily intakes for compounds in this group are developed using large uncertainty factors to account for inadequacies in the database.

Where possible, a dose (or concentration) of a chemical substance that does not produce any (adverse) effect [i.e., "no-observed-(adverse)-effect-level" (NO(A)EL)] for the critical endpoint is identified, usually from toxicological studies involving experimental animals, but sometimes from epidemiological studies of human populations. If a value for the NO(A)EL cannot be ascertained, a lowest-observed-(adverse)-effect-level (LO(A)EL) is used. The nature and severity of the critical effect (and to some extent, the steepness of the dose-response curve), are taken into account in the establishment of the NO(A)EL or LO(A)EL. For example, the concentration or dose which induces a transient increase in organ weight without accompanying biochemical or histopathological effects would generally be considered a NOEL (no observed effect-level). If there are accompanying adverse histopathological effects in the target organ, the concentration or dose at which these effects were observed would be considered a LO(A)EL.

Uncertainty factors are applied to the NO(A)EL or LO(A)EL to derive a Tolerable Daily Intake (TDI), the intake to which it is believed a person can be exposed daily over a lifetime without deleterious effect. Ideally, the NO(A)EL is derived from a lifetime (i.e., chronic) exposure study involving the most sensitive or relevant species or the most sensitive sub-population (e.g., developmental studies) in which the route of administration in animal studies is similar to that by which humans are principally exposed. Relevant species are determined, where possible, based on data on species differences in pharmacokinetic parameters or mechanism of action.

Tolerable Daily Intakes or Concentrations are not generally developed on data from acute or short-term studies unless observed effects in longer-term studies are expected to be similar. Occasionally, TDIs are based on data from sub-chronic studies, in the absence of available information from adequately designed and conducted chronic toxicity studies, and an additional factor of uncertainty is included in this case. Exceptionally, where toxicity studies using the route of exposure by which humans are principally exposed cannot be identified, a NO(A)EL or LO(A)EL from a bioassay by another route of exposure may be used where appropriate, incorporating relevant pharmacokinetic data.

An uncertainty factor accounts for uncertainty and variability in the toxicological data base of a substance. For instance, the uncertainty factor can account for:

- extrapolation of short-term experimental data to long-term human exposures;
- interspecific variability in the response to a contaminant;
- intraspecific variability (protection of sensitive individuals);
- use of LO(A)EL instead of NO(A)EL;

- other modifying factors.

The uncertainty factor is derived case-by-case, depending principally on the quality of the database. Generally, a factor of 1 to 10 accounts for intraspecies and interspecies variation. An additional factor of 1 to 100 accounts for inadequacies of the database, which include but are not necessarily limited to:

- lack of adequate data on developmental, chronic or reproductive toxicity,
- use of a LO(A)EL versus a NO(A)EL, and
- inadequacies of the critical study.

An additional uncertainty factor between one and five may be incorporated where there is sufficient information to indicate a potential for interaction with other chemical substances commonly present in the environment, particularly if these other chemicals are associated with the chemical being evaluated. If the chemical substance is essential or beneficial for human health, the dietary requirement is also taken into consideration in deriving the TDI. Numerical values of the uncertainty factor range from 10 to 10,000. Uncertainty factors greater than 10,000 are not applied since the limitations of such a database preclude the development of a reliable TDI. Where there are limitations in the protocol of the critical study a "tentative TDI" may be established.

The TDI can be defined as:

$$\text{TDI} = \frac{\text{NO(A)EL OR LO(A)EL}}{\text{Uncertainty Factors}}$$

Uncertainty factors are assigned by Health Canada based on professional judgement. Health Canada has accepted the responsibility for determining the Tolerable Daily Intake (TDI).

2.3 Toxicity Benchmarks in the Absence of Health Canada Evaluations

It is anticipated that the toxicity of most substances for which soil quality guidelines are being developed will have been evaluated by Health Canada (e.g. through the Canadian Environmental Protection Act, Priority Substances Lists I & II). However, in some cases, soil quality guidelines may be developed for substances for which Health Canada has not established a TDI/TC or RSD/RSC (e.g. ethylbenzene).

In these cases, it may be appropriate to adopt toxicity reference values developed by other regulatory agencies, particularly the United States Environmental Protection Agency (US EPA) or the World Health Organization (WHO). The supporting documentation for the toxicity benchmark should be carefully reviewed, and professional judgement used to evaluate the appropriateness of adopting the value for use in Canada.

A soil quality guideline developed using toxicity benchmarks specified by an agency other than Health Canada, or a guideline developed in the absence of any human health toxicity benchmarks, is considered to be a provisional guideline.

SECTION 3
EXPOSURE TO CONTAMINANTS

3.1 Exposure to Mixtures of Chemicals

No chemical substance exists alone in the environment; however, toxicological studies are usually carried out using single chemicals or simple mixtures. Hence, chemical interactions (additivity, antagonism, synergism, transformation, potentiation) are factors that may alter the risk posed by an individual chemical. The effects of chemical interactions on the toxicity of mixtures are not well understood; therefore, in most cases guidelines are developed for single chemicals which are treated as if they occur in isolation in the environment. However, there are instances where it is possible and desirable to evaluate groups of chemicals or mixtures.

In some cases, chemicals with similar structures will exhibit the same toxic effect but with different potency. For example, 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) generally occurs in a mixture with other congeners of the polychlorinated dibenzodioxin (PCDD) family. The overall toxicity of a PCDD mixture will be greater than just the toxicity of the 2,3,7,8-TCDD it contains. However, the toxicity will be much less than if all the PCDDs are considered to have the same toxicity as 2,3,7,8-TCDD. In this case, PCDD mixtures are assessed as toxic equivalents (TEQs) with 2,3,7,8-TCDD being the most toxic congener. Other congeners are ascribed a toxic equivalent relative to TCDD. This approach has also been applied for nonylphenol and its ethoxylates (Environment Canada, 2002). Use of TEQs assumes that all chemicals in the group have the same mode of toxicity and additive effects; if the review of toxicological data indicates that this is not the case, then application of TEQs is not appropriate.

An alternative approach to developing guidelines for a mixture of chemicals was used in the development of the Canada-Wide Standards for Petroleum Hydrocarbons in Soil (PHC CWS). The development of the PHC CWS involved dividing the range of petroleum hydrocarbons found in the environment into 4 “fractions”, then further dividing each fraction into several “sub-fractions” with similar physical-chemical properties. A single set of representative physical-chemical properties was used to represent each sub-fraction, and toxicity benchmarks representative of chemicals included in the sub-fraction were applied. With the PHC CWS approach, there is no need to measure the concentrations of individual compounds in soil; only the concentrations of the fractions are necessary. Further details of this approach are provided in the Scientific Rationale for the PHC CWS (CCME, 2000). Much like the TEQ approach, this method assumes a common mode of toxicity and additive effects within a fraction.

Evaluation of chemical mixtures should be performed on a case-by-case basis, with professional judgement used to determine the appropriate method. It is anticipated that only groups of chemically-related contaminants will be evaluated as mixtures in the foreseeable future. For more heterogeneous mixtures of contaminants, evaluation using a chemical by chemical approach is recommended.

3.2 Determining Estimated Daily Intake

Canadians are exposed to background contamination in the air, food, and water. This background exposure is quantified by the Estimated Daily Intake (EDI) for a particular contaminant. The EDI estimates the typical total concurrent background exposure from all known or suspected sources (air, water, food, soil, consumer products) via all known or suspected routes (inhalation, ingestion, dermal contact), often termed a multimedia exposure assessment (Appendix D), for the average Canadian. However, it does not include exposures which may occur from a contaminated or remediated site. This background exposure is with us at all times. Consequently, risks posed by a contaminated site must be determined in addition to this background exposure.

In general, mean concentrations in various environmental media used in estimating exposure are those interpreted as average or typical values by Health Canada, based on the original accounts and publications. Contaminant databases for environmental media may include non-detectable values, particularly for organic chemicals. Calculating a mean concentration requires that numeric values be substituted for non-detectable values. Traditional bounding approaches to this problem have either been to substitute by zero or to substitute by the detection limit, respectively under- or over-estimating the true value (Haas and Scheff, 1990; Slymen et al., 1994). A simple intermediate assumption is that non-detectable values can be estimated by half the detection limit, although this may lead to an overestimate of exposure. Where appropriate, information on concentrations of environmental contaminants in specific locales may also be used to estimate background exposure of some high exposure subgroups in the general population.

Information on the duration and frequency of exposure is also important in assessing the total daily intake of the environmental contaminant by the general population. Relevant data on behaviour and activity patterns are also considered in the development of estimates of background exposure of the general population.

SECTION 4
EXPOSURE SCENARIOS

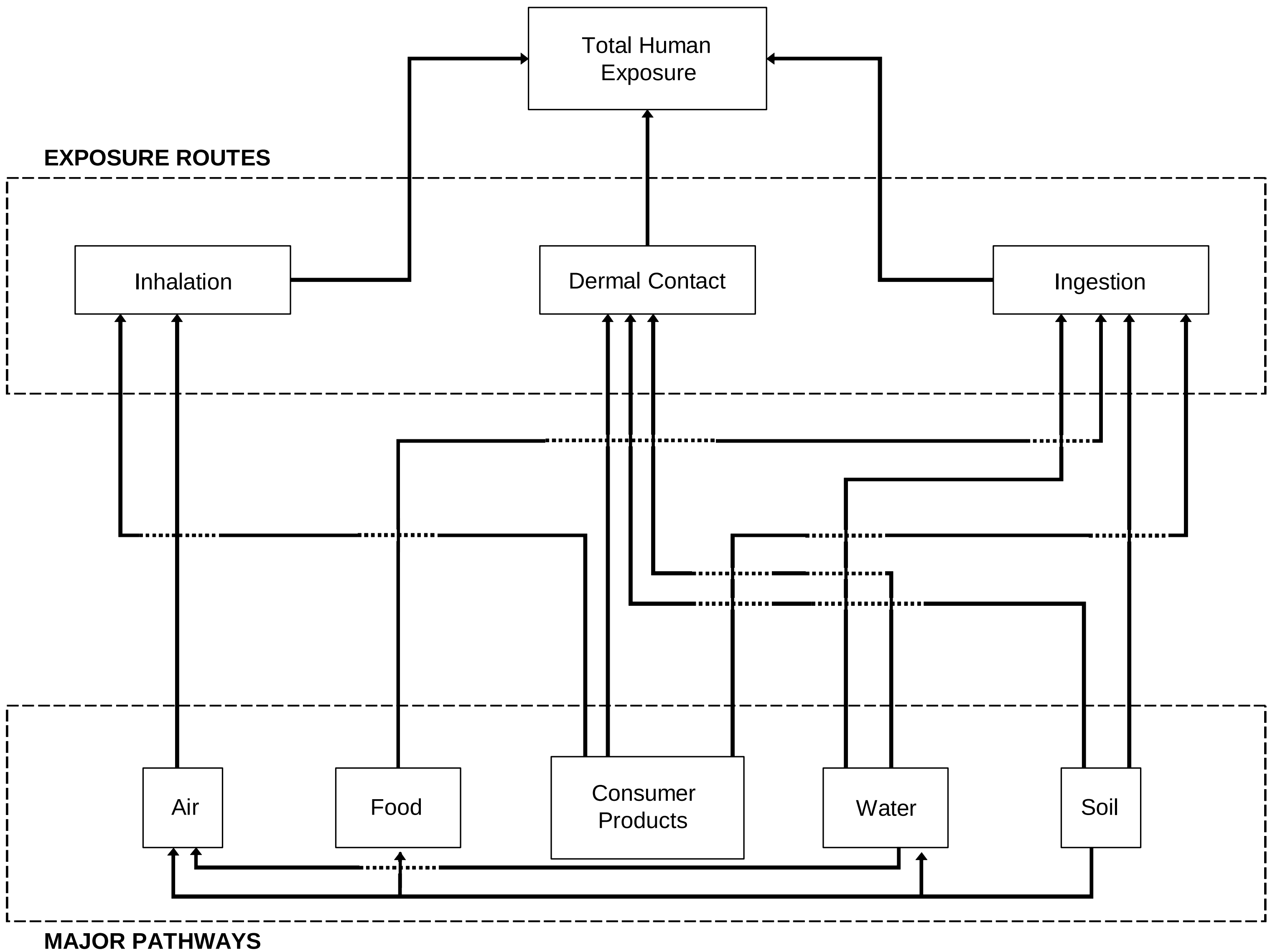
4.1 Assumptions about Exposure

In developing soil remediation guidelines to protect human health, one must ensure that exposure to contaminants at the guideline concentration will not result in adverse human health effects. For the purposes of guidelines development, CCME assumes a chronic exposure scenario (i.e., lifetime exposure to a remediated site). This is a conservative assumption, which helps ensure that no limitations will exist within the defined land use. The first step in setting soil guidelines is one of working backward from the Tolerable Daily Intake (TDI) or critical Risk Specific Dose (RSD) for a contaminant, through appropriate direct soil exposure pathways to a land use generic soil concentration. A second step considers indirect exposure pathways and protects against cross-media contamination.

A schematic diagram of potential human exposure pathways within a multimedia context is provided in Figure 15. Soil exposure pathways can result from direct or indirect exposure to soil, such as the direct ingestion of contaminated soil or cross-media contaminant transfer from soil to another medium (e.g., water, air, or food). Direct exposure pathways include ingestion of soil/dust, dermal uptake of contaminants in contact with the skin, and inhalation of soil particles into the lungs. Indirect exposure pathways include ingestion of food grown on contaminated soil, inhalation of vapours resulting from the volatilization of contaminants from soil into indoor air, and ingestion of groundwater contaminated by the leaching of contaminants from soil into groundwater. Development of guidelines is based on direct exposure pathways, with indirect exposures used to provide a check on the guidelines to ensure that generic guidelines are protective for a large majority of scenarios within a given land use.

The physical and chemical properties of a contaminant will determine its environmental fate. These properties will also focus the possible important exposure pathways to humans. For example, the dermal exposure pathway will be of prime importance for contaminants which are lipophilic and can readily cross the epidermal layer of the skin. Similarly, contaminants with a high vapour pressure likely to volatilize from soil to air are extremely important in the respiratory pathway.

Cross-media transfer of a chemical from contaminated soil to another medium, such as water, air, or food, can result in indirect exposure to contaminants from soil. Each of these cross-media transfers can be modelled using a number of assumptions about the exposure scenario. For example, the uptake of soil contaminants by plants and the subsequent ingestion of contaminated homegrown produce by people can be estimated for a given soil contaminant concentration.



PAI

Figure 15: Significant Pathways of Human Exposure to Environmental Contamination

In this document, a model for the movement of organic contaminants from soil to groundwater (see Appendix C), a model for the infiltration of volatile contaminants into indoor air via basements (Appendix E), a model for the movement of contaminants from soil into produce, meat and milk (Appendix F), and a model for the erosive movement of soil from an industrial site to an adjacent site (Appendix G) have been included. The two following examples illustrate use of these contaminant migration models. First, the guidelines derived for industrial sites should not allow for the cross-contamination of adjacent non-industrial sites by erosional forces. The model in Appendix G would call for a downward adjustment if the preliminary industrial guidelines would be projected to cause off-site contamination. Second, in an agricultural scenario, the model for bioconcentration of contaminants in food items (Appendix F) might also result in the preliminary soil guidelines being altered downward. The use of these models is described in Sections 5.5 and 5.6.

If the defined exposure scenario used in developing the generic guidelines is thought to be inappropriate for the particular site to be remediated, further guidance to allow modification of the generic guidelines within limits, through the setting of site-specific objectives has been developed (See Part A, Section 1.1). For example, this may involve the removal, addition, or calibration of exposure pathways to more accurately represent the exposure scenario present at a specific site.

4.1.1 Threshold Contaminants

To derive a guideline, it is necessary to ascribe some allowable proportion of total chemical exposure to the soil medium. One approach is to base allowable contaminant apportionment to the various media in proportion to the distribution expected if the various media were in equilibrium in terms of contaminant intermedia distribution (Travis and Hattemer-Frey, 1989). This assumes that the air, water, soil and food to which most Canadians are typically exposed exist in such equilibrium. However, much of the food consumed in Canada is imported or grown in areas distant from the residence (agricultural lands being an exception). Further, many Canadians consume water that has been treated and supplied by a municipal source. Canadians also spend a majority of time indoors where air is heated, cooled, and often contaminated by emissions from various consumer products, building materials, and lifestyle activities. Therefore environmental media are not likely to be in equilibrium. Apportionment of allowable exposure between media based on theoretical equilibria is therefore not appropriate.

SQGTG has proposed a simple and practical solution to this problem which recognizes that no single medium should deplete the entire tolerable daily intake or even the entire residual tolerable daily intake (RTDI). The RTDI is the difference between the tolerable daily intake and the estimated daily intake (RTDI = TDI - EDI). With up to five primary media to which people are potentially exposed (i.e., air, water, soil, food, and consumer products) SQGTG proposes that a default value of 20% of the residual acceptable daily intake be apportioned to each of these five media (Figure 16).

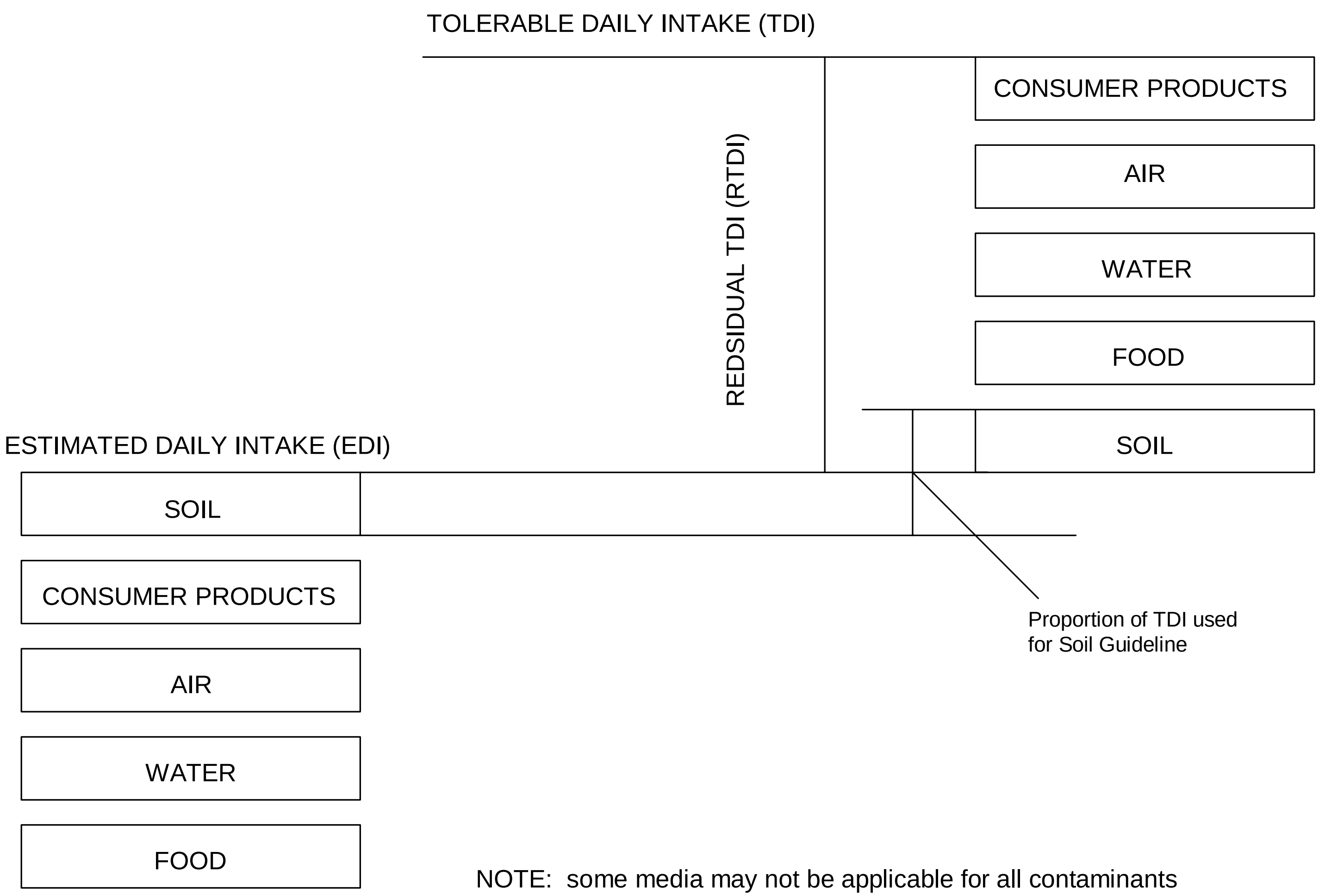


Figure 16: Conceptual Derivation of the Soil Guideline for Threshold Substances from the Multimedia Exposure Assessment and Assumed Soil Allocation Factor from the Residual Tolerable Daily Intake

In the default case, SQGTG proposes to apportion only 20% of the RTDI to soils for the purpose of deriving soil remediation guidelines. This "Soil Allocation Factor" (SAF) of 20% allows for 80% of the remaining tolerable incremental exposure to be reserved for other media (i.e., food, air, water, and consumer products). Although this soil allocation factor has been arbitrarily established, SQGTG believes that soils containing a substance at the guideline level will not cause the total exposure from all media (air, water, food, and contaminated soil), via all direct and indirect pathways (ingestion, inhalation, and dermal absorption), to exceed the TDI. The generic soil guidelines are calculated after considering the sum of the background soil exposure and the percentage (20%) of residual tolerable daily intake allocated to soil (Figure 16).

Depending on their physical and chemical properties, some soil contaminants may not normally be present in all four of the remaining media (air, water, food and consumer products). For example, high molecular weight hydrocarbons exhibit very low solubility and volatility and, as a result, the contribution of air and water to overall human exposure may be insignificant. If defensible contaminant-specific evidence exists demonstrating that the contaminant does not occur in a given medium, the RTDI may be distributed amongst fewer media and the soil allocation factor may be increased from 20% to a value given by:

$$SAF = 100\% / (\text{number of applicable exposure media})$$

When the EDI is greater than the TDI (RTDI = 0), theoretically the population cannot be safely subjected to any increased exposure. In these circumstances, the provisional soil quality guideline should be set at the background soil concentration or practical quantitation limit for that contaminant. Since this may result in a fairly restrictive criterion, data and any models used to develop the EDI should be checked to ensure their accuracy, and to assess any regional or site-specific factors.

4.1.2 Non-threshold Contaminants

In theory, Figure 15 would also be applicable to carcinogenic substances, as low levels of "background" exposure occur for many carcinogens. However, tolerable daily intake (TDI) and tolerable incremental exposure can not be determined for carcinogens as some level of risk is assumed to exist at any level of exposure other than zero.

For the purposes of this protocol, SQGTG derives human health soil quality guidelines representative of both a 1×10^{-5} and a 1×10^{-6} incremental risk above background from remediated soils at the guidelines concentration; individual jurisdictions may apply guidelines based on either one of these incremental risk levels. The uses of different incremental risk levels can be calculated and incorporated into the development of a site-specific objective, subject to the approval of the jurisdictional authority.

4.2 Absorption of Chemicals into the Body

The health risk posed by a particular inhalation, ingestion, or dermal exposure depends on the absorbed dose, which reflects properties of both the chemical and body surfaces involved. There is opportunity to apply information on absorption efficiency (i.e., absorption factors) during guideline development provided that the absorbed dose in the primary toxicological study(ies) is known. However, often only an administered dose or exposure is documented in the toxicological study. Where the critical toxicological study has used a different medium than that under investigation, sometimes an absorption factor is applied to account for the difference in absorption of the contaminant by the body in the two different media, when this information is available. For the purposes of guideline development, it is usually assumed that absorption efficiency in an environmental exposure is equal to that of the toxicological study. In such cases, the absorption rate has been accounted for in the development of the NO(A)EL or slope factor and no adjustment for absorption is necessary.

For each chemical, the data on its absorption into the body will be evaluated. When delivered doses are known, available scientific information will be consulted to assess the feasibility of assigning an absorption factor for the relevant exposure pathway(s).

Where there is sufficient information to evaluate the absorption into the body from both the environmental exposure route being evaluated and the study exposure route, the relative absorption factor can be calculated as:

$$AF = \frac{AE_{EM}}{AE_{SM}}$$

where:

AF	=	relative absorption factor for the environmental exposure route
AE_{EM}	=	absorption efficiency from the environmental exposure route/medium
AE_{SM}	=	absorption efficiency from the study exposure route/medium

If the primary toxicological study is based on an absorbed dose instead of an administered dose, then the absorption efficiency from the environmental exposure route may be applied directly as an absorption factor; however, this is not expected to be the case for most contaminants.

Since most toxicity studies are based on ingested or inhaled doses of contaminants, it is anticipated that the absorption factor will be 1 for ingestion- and inhalation-based pathways, unless there is sufficient information to evaluate the relative absorption efficiencies between the media involved (e.g. a toxicity study based on ingestion of the contaminant in solution vs. environmental exposure via ingestion of soil). Absorption factors are more likely to be applied for dermal contact exposure routes, since few TDIs are based on dermal exposure.

4.3 Receptors, Exposure Pathways, and Land Uses

4.3.1 General

The development of the soil quality guideline is a two step process. The first step considers all direct soil exposure pathways, including the ingestion of soil/dust, dermal contact, and inhalation of soil particles into lungs, as well as the primary indirect pathways. The primary indirect pathways that are included in the first step are dependent on the contaminant type, but may include inhalation of vapours migrating into indoor air (for volatile contaminants), ingestion of groundwater used as potable water (for soluble organic contaminants) and consumption of produce (for substances that biomagnify). The latter pathway applies principally to agricultural land use. The actual inclusion of each pathway in the guidelines derivation equation is based on the quality of the scientific evidence that a pathway is contributing to exposure. For cases where exposure pathways have been excluded, this decision will be reassessed as new scientific data becomes available.

Although ingestion of groundwater used as potable water is considered a primary pathway, for all defined land uses, the guideline calculated for this pathway may be excluded if deemed appropriate by the implementing authority.

The second step in the soil quality guideline derivation is the consideration of other indirect soil exposure pathways through the use of check mechanisms. In the check mechanisms, exposure is evaluated through the use of simplified models which utilize conservative generic input values for site-specific characteristics. Indirect pathways that are considered as check mechanisms are consumption of produce at agricultural and residential sites (for all substances that may bioaccumulate) and offsite migration of soil/dust under commercial or industrial land use.

The choice of sensitive receptors is linked to land use considerations. Guidelines will be developed for four defined primary land uses—agricultural, residential/parkland, commercial, and industrial. While the details of the nature and extent of exposure arising from these four defined land uses are different, the practical expression of these differences is dependent on the sensitive human receptor chosen to represent the occupant or user for the land use, and the exposure period (i.e., the frequency, duration, and intensity of the exposure assumed for the land use).

Studies indicate that toddlers ingest much greater amounts of soil and dust each day than adults, primarily due to normal mouthing activity and a greater time spent playing out of doors and on the floor. This greater intake of soil combined with a lower average body mass to "distribute" the dose places a child more at risk from contaminated soils than the adult. For dermal exposures, behavioural considerations are also likely to involve greater proportions of a toddler's versus an adult's skin (the toddler is more likely to expose the lower legs in addition to the face, neck, hands, and arms). Typical values for these and other receptor characteristics are in Appendix I.

In the case of non-threshold substances, hazard is normally assessed for an adult, as exposure is assumed to be continuous over 70 years; however, recent work indicates that potential child susceptibility to carcinogens should also be evaluated where such information is available, to ensure that the most sensitive life stage is evaluated. For threshold substances, exposure is averaged over, and TDIs measured against, the most sensitive life stage. Generally, this is the "toddler" stage (six months to four years). However, if a different age group is determined to be more sensitive (e.g. if there is a lower TDI for a different age group), then this more sensitive age group should be used as the receptor.

4.3.2 Defined Scenario for Agricultural Land Use

The generic agricultural scenario envisioned by SQGTG is a multi-functional farm with a family, including children (in particular, toddlers), resident on the property. This generic farm grows produce, raises livestock, and has a dairy herd, such that a large portion of the produce, meat, and milk being consumed by the family are produced on the farm. The family residence may either include a basement or be slab-on-grade, and groundwater may be used as potable water. The exposure assumptions for the agricultural site are shown in Figure 17. Other land use options commonly occur within the agricultural category, but SQGTG considers that these will not often represent greater environmental sensitivity.

4.3.3 Defined Scenario for Residential/Parkland Land Use

The generic residential/parkland scenario is a typical single family home with a basement and a backyard where the children (in particular, toddlers) play. Groundwater may be used as potable water. Differences in soil remediation guidelines between agricultural and residential/parkland land use will generally only arise when a contaminant bioaccumulates in the food chain and results in contamination of food produced on the agricultural site. The exposure assumptions for the residential/parkland site are shown in Figure 18.

4.3.4 Defined Scenario for Commercial Land Use

SQGTG recognises the existence of a "commercial" land use classification intermediate between residential/parkland and industrial. Although the commercial category is considered a discrete land use classification, exposure conditions and receptor characteristics at a particular site may substantially overlap those defined for residential/parkland or industrial land uses.

Commercial sites are generically defined as sites where commercial as opposed to residential or industrial activities predominate. An example of a commercial site as envisioned by the SQGTG is a typical urban shopping mall. As envisioned by SQGTG, individuals do not conduct manufacturing activities or reside at commercial sites.

All age groups generally have full access to commercial properties, and some commercial properties will include daycare facilities. Hence, the toddler was chosen as the critical receptor. In addition, the exposure period believed generally operative at commercial lands differs from those for other land uses. Relative to residential lands, exposure to soils on commercial land is expected to have less intensity, duration, and frequency. The exposure assumptions for the commercial site are summarized in Figure 19.

Since commercial sites include receptor and exposure characteristics which may be common to either residential/parkland or industrial sites, absolute definition of a commercial site is difficult. Consequently, SQGTG cautions that site managers should use discretion when classifying sites as commercial properties. In no case should a site which allows unrestricted 24-hour access by children (in particular, toddlers) or residential occupancy by any individual be considered a commercial site. If these conditions exist the appropriate classification is residential/parkland. Also, a site where children have extensive contact with soil, such as a playground, should be classified as residential/parkland. Similarly, sites where children are prohibited and located in a primary industrial zone could be considered industrial as opposed to commercial. If doubts exist regarding appropriate land use classification, the jurisdictional authority should be consulted.

4.3.5 Defined Scenario for Industrial Land Use

The generic industrial scenario envisioned by SQGTG is a factory site where goods are produced. For industrial land uses, public access is assumed to be both controlled and limited, and those typically spending the greatest time on-site would be working adults. The sensitive receptor for industrial land uses is therefore a working adult. Adults consume one quarter of the soil typically consumed by toddlers, and have about five times the body mass to distribute the chemical. Adults as critical receptors, coupled with assumed exposure period used for industrial lands, will result in industrial guidelines generally being greater than those for agricultural and residential/parkland land uses.

Derivation of high soil quality guidelines concentrations for industrial sites could result in contamination of local residential sites through the off-site migration of soils and dust. This potential off-site migration will be checked with the procedure contained in Appendix G. As with other land use scenarios, groundwater sources of potable water, and possible industrial water usage such as food processing, should be protected. Soil quality guidelines will be established that are likely to protect potable water quality for the groundwater below an industrial site remediated to generic guidelines. The exposure assumptions for industrial land uses are shown in Figure 20.

Agricultural Land Use
Sensitive Receptor: toddler *(threshold contaminants)* adult *(non-threshold contaminants)* Exposure Period: 24 hours/day 365 days/year Direct Soil Exposure Pathways: • direct soil ingestion • direct soil dermal contact • direct soil particulate inhalation Primary Indirect Soil Exposure Pathways • ingestion of groundwater • infiltration of volatile contaminants into indoor air via foundations • consumption of produce, meat and milk produced on-site (for substances that biomagnify) assumptions: • 100% of milk ingested produced on site • 50% of produce ingested grown on site • 50% of meat ingested produced on-site
Indirect Soil Exposure Pathway considered as a check mechanism: • consumption of produce, meat, and milk produced on-site (for non-biomagnifying substances) assumptions: • 100% of milk ingested produced on-site • 50% of produce ingested grown on-site • 50% of meat ingested produced on-site This check mechanism can result in the lowering of the final SQC_{HH} via a management adjustment factor.

Figure 17 Exposure Assumptions for Defined Agricultural Land Use Scenario

Residential/Parkland Land Use

Sensitive Receptor:	toddler	*(threshold contaminants)*
	adult	*(non-threshold contaminants)*
Exposure Period:	24 hours/day	
	365 days/year	

Direct Soil Exposure Pathways:
- direct soil ingestion
- direct soil dermal contact
- direct soil particulate inhalation

Primary Indirect Soil Exposure Pathways
- ingestion of groundwater
- infiltration of volatile contaminants into indoor air via foundations

Indirect Soil Exposure Pathway considered as a check mechanism for residential land with backyard garden:
- consumption of backyard garden produce

assumption:
- 10% of produce ingested grown on-site

This check mechanism can result in the lowering of the final SQC_{HH} via a management adjustment factor.

Figure 18 Exposure Assumptions for Defined Residential/Parkland Land Use Scenario

Commercial Land Use

Sensitive Receptor:	toddler	*(threshold contaminants)*
	adult	*(non-threshold contaminants)*
Exposure Period:	10 hours/day	
	5 days/week	
	48 weeks/year	

(Note: exposure term = 1 for non-threshold contaminants)

Direct Soil Exposure Pathways:
- direct soil ingestion
- direct soil dermal contact
- direct soil particulate inhalation

Primary Indirect Soil Exposure Pathways
- ingestion of groundwater
- infiltration of volatile contaminants into indoor air via foundations

Indirect Soil Exposure Pathway considered as check mechanism:
- off-site migration of soil contaminants via wind and water erosion

This check mechanism can result in the lowering of the final SQC_{HH} via a management adjustment factor.

Figure 19 Exposure Assumptions for Defined Commercial Land Use Scenario

Industrial Land Use

Sensitive Receptor: adult

Exposure Period: 10 hours/day
5 days/week
48 weeks/year

(Note: exposure term = 1 for non-threshold contaminants)

Direct Soil Exposure Pathways:

- direct soil ingestion
- direct soil dermal contact
- direct soil particulate inhalation

Primary Indirect Soil Exposure Pathways

- ingestion of groundwater
- infiltration of volatile contaminants into indoor air via foundations

Indirect Soil Exposure Pathway considered as check mechanism:

- off-site migration of soil contaminants via wind and water erosion

This check mechanism can result in the lowering of the final SQC_{HH} via a management adjustment factor.

Figure 20 Exposure Assumptions for Defined Industrial Land Use Scenario

SECTION 5
HUMAN HEALTH GUIDELINES DERIVATION

The exposure assumptions used in calculating the guidelines for direct contact with soil, identified as direct human health-based soil quality guidelines (SQG_{DH}), for each land use were described in Sections 4.3.2 to 4.3.5. Direct exposure includes three separate pathways, which are summed together: ingestion of soil, dermal contact with soil, and inhalation of suspended soil particulate. It is anticipated that for most contaminants, soil ingestion will be the dominant direct exposure pathway. Particulate inhalation is not expected to contribute significantly to direct exposure for most chemicals; however, particulate inhalation may become more important for non-volatile substances with high inhalation toxicity (e.g. cadmium) (US EPA, 1996).

Details of the equations and numerical procedures used to calculate the direct human health-based soil quality guidelines are provided in Sections 5.1 and 5.2. All equations and model input parameters are also summarized in Appendices H and I, respectively.

In principle, for threshold contaminants, the total exposure from direct soil pathways should not generally exceed typical background soil exposures by more than 20% of the residual tolerable daily intake (RTDI), although >20% may be allotted under certain circumstances. If the chemical is identified as a non-threshold substance by Health Canada, then SQGTG will develop a guideline representing an incremental risk from soil exposure of 10^{-6} above the background soil concentration.

While volatile compounds may be expected to volatilize rapidly from the surficial soils to which humans may be directly exposed, direct contact guidelines should still be developed for both volatile and non-volatile compounds. It is anticipated that direct contact will not be the governing exposure pathway for volatile compounds, and if it is then this pathway could be eliminated on a site-specific basis if analytical data show that the contaminant is not present in surficial soils (though elimination of direct exposure pathways may be accompanied by land use restrictions in some jurisdictions). Particulate inhalation does not need to be evaluated for volatile organic compounds, since they are not normally associated with particulate matter in air.

Direct contact soil quality guidelines may in some cases not be protective for acute exposure of children during pica events (Calabrese *et al*., 1997). Where adequate acute toxicity benchmarks are available, a guideline value for the protection of pica children should be calculated using the methods described by Calabrese *et al.* (1997) and presented in the supporting documents for the soil quality guideline.

5.1 Direct Human Health-Based Soil Guidelines Derivation for Threshold Substances

The direct human health-based soil guideline (SQG_{DH}) is calculated using the following equation:

$$SQG_{DH} = \frac{(TDI - EDI) \times SAF \times BW}{[(AF_G \times SIR) + (AF_S \times SR) + (AF_L \times IR_S) \times ET_2] \times ET_1} + BSC$$

where,

SQG_{DH}	=	direct human health-based soil quality guideline (mg/kg)
TDI	=	tolerable daily intake (mg/kg bw-day)
EDI	=	estimated daily intake (multimedia exposure assessment) (mg/kg-day)
SAF	=	soil allocation factor (unitless)
BW	=	body weight (kg)
BSC	=	background soil concentration (mg/kg)
AF_G	=	relative absorption factor for gut (unitless)
AF_L	=	relative absorption factor for lung (unitless)
AF_S	=	relative absorption factor for skin (unitless)
SIR	=	soil ingestion rate (kg/day)
IR_S	=	soil inhalation rate (kg/day)
SR	=	soil dermal contact rate (kg/day)
ET_1	=	exposure term 1 (unitless) – days per week/7 x weeks per year/52
ET_2	=	exposure term 2 (unitless) – hours per day/24

The soil inhalation rate is defined as the amount of respirable soil particles inhaled in a day. The soil dermal contact rate is the amount of soil contacting the skin in a day. The soil ingestion rate refers to the amount of soil ingested on a daily basis. Absorption factors may be required where the critical toxicity study used in developing the NO(A)EL employed an absorbed dose rather than an administered dose, or where the critical toxicity study has employed a different medium than that under investigation; further details are presented in Section 4.2. Then soil ingestion, dermal contact, and inhalation rates are multiplied by corresponding relative absorption factors (AF), when these data are available. The exposure term is the ratio of the defined exposure period for each land use to the maximum exposure period (24 hours/day x 365 days/year). Note that hours per day exposure is not considered for soil ingestion or dermal contact, consistent with Health Canada (2003b) recommendations, since soil ingestion and dermal contact are not expected to occur at a uniform rate throughout the day.

Soil type considerations do not affect the calculation of the SQG_{DH}, unless there are sufficient data available to determine separate relative absorption factors for coarse and fine soils.

In some cases, the mechanism of toxicity may be different for the different exposure routes, and separate TDIs may be used (e.g., in many cases there are separate oral and inhalation TDIs, and some chemicals are treated as a threshold chemical for some exposure routes and a non-threshold chemical for others). In these cases only, direct exposure pathways may be evaluated separately by eliminating terms related to the other pathways from the above equation. For example:

Soil ingestion only:

$$SQG_{DH-SI} = \frac{(TDI - EDI) \times SF \times BW}{(AF_G \times SIR) \times ET_1} + BSC$$

Dermal contact only:

$$SQG_{DH-DC} = \frac{(TDI - EDI) \times SF \times BW}{(AF_S \times SR) \times ET_1} + BSC$$

Particulate inhalation only:

$$SQG_{DH-PI} = \frac{(TDI - EDI) \times SF \times BW}{(AF_L \times IR_S) \times ET_1 \times ET_2} + BSC$$

The SQG_{DH} is then the lowest calculated value for any of the separate pathways.

5.2 Direct Human Health-Based Soil Guidelines Derivation for Non-threshold Substances

If the chemical is identified as a non-threshold substance by Health Canada, then SQGTG will develop guidelines representing an incremental risk from soil exposure of both 10^{-5} and 10^{-6} above the background soil concentration. The use of other critical risk levels can easily be accommodated by jurisdictional authorities or at a site-specific objective level. The direct human health-based soil guideline is established as follows:

$$SQG_{DH} = \frac{RSD \times BW}{[(AF_G \times SIR) + (AF_S \times SR) + (AF_L \times IR_S)] \times ET} + BSC$$

where,

SQG_{DH}	=	direct human health-based soil quality guideline (mg/kg)
RSD	=	risk specific dose (mg/kg-day)
BW	=	body weight (kg)
BSC	=	background soil concentration (mg/kg)
AF_G	=	relative absorption factor for gut (unitless)
AF_L	=	relative absorption factor for lung (unitless)
AF_S	=	relative absorption factor for skin (unitless)
SIR	=	soil ingestion rate (kg/day)
IR_S	=	soil inhalation rate (kg/day)
SR	=	soil dermal contact rate (kg/day)
ET	=	exposure term (unitless) = 1

The adult is the receptor when considering lifetime cancer risk. Absorption factors may be required when the critical toxicity study used in developing the cancer slope factor has used an absorbed dose rather than an administered dose. Absorption factors may also be required when the critical toxicity study employed a different medium in developing the cancer slope factor than that under investigation (see Section 4.2). Then soil ingestion, dermal contact, and inhalation rates are multiplied by corresponding relative absorption factors (AF), when these data are available. For non-threshold substances, the exposure term for all land uses (including commercial and industrial) is one since the exposure period (i.e., 10 hours/day, 5 days/week, 48 weeks/year for 30 to 40 years over a lifetime) exceeds the likely latency period for most carcinogens.

Soil type considerations do not affect the calculation of the SQG_{DH}, unless there are sufficient data available to determine separate relative absorption factors for coarse and fine soils.

As for threshold chemicals, if the mechanism of toxicity is different for different exposure routes, then one or all of the direct contact pathways may be evaluated separately, with the lowest calculated value becoming the SQG_{DH}. The separated equations are:

Soil ingestion only:

$$PSQG_{HH-SI} = \frac{RSD \times BW}{(AF_G \times SIR) \times ET} + BSC$$

Dermal contact only:

$$PSQG_{HH-DC} = \frac{RSD \times BW}{(AF_S \times SR) \times ET} + BSC$$

Particulate inhalation only:

$$PSQG_{HH-PI} = \frac{RSD \times BW}{(AF_L \times IR_S) \times ET} + BSC$$

5.3 Guidelines for the Protection of Potable Groundwater

5.3.1 General

Soils are hydrologically linked to groundwater systems. Therefore, soil contamination can and does lead to groundwater contamination. In some cases, soils may be hydrologically linked to groundwater that is being used as a source of potable water, or conceivably could be used as a potable water source in the future. Therefore, soil quality guidelines must be protective of potential use of groundwater as a potable water source (i.e. predicted contaminant concentrations in groundwater at a remediated site cannot exceed the Guidelines for Canadian Drinking Water Quality).

In some locations, use of contaminated groundwater as a potable water source may not be a concern, because of:

- municipal bylaws prohibiting water wells for potable water use;
- naturally non-potable shallow groundwater; or
- lack of hydrological connection between contaminated soils and groundwater aquifers with sufficient recharge for potable water use.

For these reasons, the soil quality guideline for the protection of potable groundwater (SQG_{PW}) may be excluded on a site-specific basis if it can be demonstrated that existing or potential potable water sources are not likely to be affected.

5.3.2 Development of Source Guidance Values for Groundwater

If there is no guideline for the contaminant being evaluated in the Guidelines for Canadian Drinking Water Quality, then an allowable concentration in potable water (Source Guidance Value for Groundwater) can be derived using the following equation:

$$SGVG = \frac{TDI \times BW \times WF}{WIR} \quad or \quad SGVG = \frac{RSD \times BW}{WIR}$$

where:

SGVG	=	source guidance value for groundwater (mg/L)
TDI	=	tolerable daily intake (mg/kg/d)
WF	=	water allocation factor (unitless)
RSD	=	risk-specific dose (mg/kg/d)
BW	=	body weight (kg)
WIR	=	water ingestion rate (L/d)

Generally, body weight and water ingestion rate should be based on an average daily intake of 1.5 L of drinking water by a 70.7 kg adult (Health Canada 2003b). However, where appropriate, the SGVG may be derived based on intake in the most sensitive subpopulation (e.g., pregnant women, children). The risk-specific dose, for non-threshold substances, should be based on an incremental lifetime cancer risk of 10^{-6}.

If the derived potable water threshold value is higher than the pure-phase solubility of the contaminant, then calculation of the SQG_{PW} is not required for the contaminant.

The water allocation factor (WF) is analogous, and normally equal to, the SAF (see Section 4.1.1). As for the SAF, the default value for the WF is 0.2.

5.3.3 Derivation of the SQG_{PW}

The SQG_{PW} is determined using the groundwater model detailed in Appendix C, with the drinking water guideline or Source Guidance Value for Groundwater applied as the allowable concentration in groundwater at the receptor. For purposes of generic guideline development, it is assumed that the contaminated soil is in contact with the groundwater, and that a potable water well could be installed at the edge of the remediated area (i.e. the lateral distance from the source to the receptor is 0 and transport through the saturated zone is not considered).

5.4 Guidelines for the Protection of Indoor Air Quality

Volatile organic compounds have migrated into basements of homes from underground storage tanks leaking petroleum-based fuels, and from hazardous waste landfills where vinyl chloride was improperly disposed (Stephans *et al*., 1986). Contamination of indoor air by volatilization from contaminated soil is a critical pathway of exposure for volatile organic chemicals. Therefore, human health soil guidelines for volatile organic chemicals must be protective of indoor air quality.

Volatile contaminants in soil are found adsorbed to soil particles, dissolved in soil porewater, and in vapour phase within the soil pores. The relative proportions of the contaminant in each phase are a function of various chemical properties, including the organic carbon partitioning coefficient (K_{oc}), solubility, and vapour pressure. Further details on this relationship are provided in Appendix A.

Once in the vapour phase, volatile organic compounds migrate into buildings via diffusion and barometric pressure differentials between the soil gas and the indoor air. The migration into indoor air is also a function of a variety of factors including soil type, depth or distance of contamination from the building foundation, type of building foundation, the building air exchange rate, and building dimensions. The processes are evaluated using an analytical model developed by Johnson and Ettinger (1991); further details are provided in Appendix E.

The soil quality guideline for the protection of indoor air quality (SQG_{IAQ}) is calculated using the equations presented in Appendix H and model input parameters presented in Appendix I. The allowable indoor air concentration originating from soil contamination is the reference concentration (RfC) or tolerable concentration (TC) minus the background indoor air concentration for threshold substances, or the risk-specific concentration (RSC) for non-threshold substances. If only a tolerable daily intake (TDI) or risk-specific dose (RSD) is available, then a toxicity benchmark for inhalation can be calculated as:

$$TC = \frac{TDI \times BW}{IR} \quad \text{or} \quad RSC = \frac{RSD \times BW}{IR}$$

where:

TDI	=	tolerable daily intake (mg/kg/d)
RSD	=	risk-specific dose (mg/kg/d)
BW	=	body weight (kg) of the critical receptor (Appendix I)
IR	=	air inhalation rate (m^3/d) of the critical receptor (Appendix I)

It should be noted that there may be considerable uncertainty in a TC or RSC extrapolated from an oral toxicity benchmark, since the target organs and toxicity mechanisms may be different for inhalation exposure than for oral exposure. However, in the absence of inhalation-specific toxicity benchmarks, use of an extrapolated toxicity benchmark is considered to be more appropriate than excluding the pathway for protection of indoor air quality for a volatile organic compound.

5.5 Evaluation for Contamination of Produce, Milk, and Meat

Humans can be indirectly exposed to contaminants in soil through food-chain contamination of produce, meat, and milk. For agricultural land use, it is likely that some meat, produce, and milk will be produced and consumed on-site. Fruits and vegetables grown in residential gardens may also be a source of human exposure to contaminants. To ensure that soil remediation guidelines do not result in an unacceptable contribution to total daily intake of contaminants via home-grown produce, meat, and milk, it is necessary to compare the expected intake of contaminants from these sources with the total daily intake.

The procedure and assumptions for estimating the daily intake of contaminants in food grown on a contaminated site are described in Appendix F. The procedure relies on the use of bioconcentration factors. The identification of foods of concern with regard to bioconcentration will depend on the contaminant's physio-chemical properties. For the agricultural site, SQGTG assumed that for agricultural lands, 50% of meat and produce, and 100% of milk consumed by residents was produced on site (Appendix F). This approach reflects the variations in growing seasons and dependence on other food sources. For residential lands, it is assumed that 10% of produce (no meat or milk) is grown in a backyard garden. The SQG_{FI} is calculated using the TDI and EDI with the soil allocation factor, or the risk-specific dose, as specified in Appendix F; equations and model input parameters are included in Appendices H and I. As a further check, calculated concentrations in food are compared to the Maximum Residue Limits (MRLs) found in the Food and Drug Act (1985). Where MRLs are exceeded, the SQG_{FI} will be lowered to ensure that unacceptable contamination of meat, milk or produce does not occur.

SQGTG acknowledges the imprecise nature of this model, the uncertainty surrounding the underlying assumptions, and the use of scientific judgement in determining input parameters. Therefore, it is recommended that the meat, produce and milk ingestion pathways be evaluated as a check mechanism, for substances which are not known to biomagnify. If the substance does not biomagnify, professional judgement should be applied before using the result as a generic guideline (it is a required pathway for biomagnifying substances), based on the level of confidence in the calculated guideline. Of particular note, low guidelines may be calculated for some organic compounds which in reality would be likely to be metabolized, especially if the bioconcentration factors are estimated from the K_{ow} (see Appendix F).

Acknowledging potential variations in lifestyle, geographical considerations, and frequency of garden produce consumption, SQGTG has some reservations about considering local food consumption in generating generic guidelines for a residential setting. SQGTG therefore recommends that generic guidelines for residential sites not take into account exposure from local produce consumption. However, residential guidelines values calculated on a 10% consumption of homegrown produce will be available in each contaminant assessment document. For residential sites with homegrown produce, SQGTG recommends that the modification of generic residential guidelines to include this exposure pathway be considered in determining a site-specific remediation objective.

5.6 Off-site Migration of Soil/Dust from Commercial or Industrial Sites

In deriving soil quality guidelines for commercial industrial sites, SQGTG uses an exposure scenario which considers on-site exposure only, (i.e., during a normal work week). However, wind and water erosion of soil and subsequent deposition can transfer contaminated soil from one site to another.

Therefore, SQGTG has developed a model to address the subsequent movement of soil from a commercial or industrial site to adjacent, more sensitive land (e.g., agricultural or residential property). This procedure is briefly described below. The full details and assumptions of the model are provided in Appendix G. This check mechanism is not applied for volatile organic compounds, which are not expected to be associated with soil particles transported by wind and water.

Calculations using the Universal Soil Loss Equation and the Wind Erosion Equation are used to estimate the transfer of soil to an adjacent property. It is possible to calculate the concentration in eroded soil from the industrial site that will raise the contaminant concentration in the receiving soil to equal the agricultural or residential/parkland guideline within a specified period of time; this concentration is applied as the human health soil quality guideline for off-site migration (SQG_{OM-HH}). SQGTG acknowledges the imprecise nature of this model, the uncertainty surrounding the underlying assumptions, and use of scientific judgement in determining input parameters; therefore this pathway is considered to be a check mechanism, and professional judgement should be used to determine whether the SQG_{HH} should be modified by this pathway. At specific industrial sites, management actions may be taken to prevent or limit erosive losses of surface soils. Accommodation for such situations is provided in the guidance for the development of site-specific objectives.

5.7 Consideration of Additional Exposure Pathways

It is anticipated that in most situations, the exposure pathways described above will be sufficient for the development of human health quality guidelines. However, other exposure pathways exist, such as volatilization of chemicals from a domestic water supply. If the literature review indicates that another exposure pathway may be of particular concern, then this pathway should be evaluated. Specific guidance on the evaluation of additional exposure pathways is not provided herein at this time; where possible, methods published by regulatory agencies such as Health Canada, Environment Canada, or US EPA should be applied. Professional judgement should be used to determine whether the final human health soil quality guideline is modified by these additional pathways.

SECTION 6
DERIVATION OF THE FINAL HUMAN HEALTH SOIL QUALITY GUIDELINES

The protocol defines three types of exposure pathway: required pathways, applicable pathways, and check mechanisms.

- Required pathways must be calculated, and are included in the derivation of the overall human health soil quality guideline (SQG_{HH}). If insufficient data exist to calculate a required pathway, then the SQG_{HH} cannot be calculated.
- Applicable pathways must be calculated if sufficient data are available, and, if calculated, are included in the derivation of the overall SQG_{HH}. However, even if insufficient data exist to calculate an applicable pathway, the SQG_{HH} can still be calculated.
- Check values must be calculated if sufficient data are available, and, if calculated, may or may not be included in the calculation of the overall SQG_{HH}, based on professional judgement. If insufficient data exist to calculate a check value, the SQG_{HH} can still be calculated.

The pathways which must be evaluated for each land use and chemical type to determine the SQG_{HH} are shown in Table 3.

Table 3 Exposure Pathways for Development of the SQG_{HH}

Pathway	Agriculture	Residential/ Parkland	Commercial	Industrial
- Direct Contact (SQG_{DH})	All[a]	All[a]	All[a]	All[a]
- Potable groundwater (SQG_{PW})	Soluble	Soluble	Soluble	Soluble
- Indoor air quality (SQG_{IAQ})	Volatile (basement and slab-on-grade)	Volatile (basement and slab-on-grade)	Volatile (slab-on-grade)	Volatile (slab-on-grade)
- Consumption of produce, meat, and milk (SQG_{FI})	Required for biomagnifying; recommended for all[b,c]	Produce only[b,c]	None	None
- Offsite migration[a] (SQG_{OM-HH})	None	None	Non-volatile[b]	Non-volatile[b]

a – pathway is required (i.e. final guideline cannot be developed without it)
b – pathway is considered to be a check mechanism
c – check mechanism may not be relevant if substance does not bioaccumulate

6.1 Agricultural Land Use

The direct human health-based soil quality guideline (SQG_{DH}) is calculated using the equations in Sections 5.1 or 5.2, depending on whether the contaminant is a threshold or non-threshold contaminant. For agricultural land use, the soil quality guidelines for indirect exposure to soil contaminants via infiltration of volatile compounds into indoor air (SQG_{IAQ}), protection of potable groundwater (SQG_{PW}) and ingestion of produce, meat, and milk produced on-site (SQG_{FI}) are all calculated. The final SQG_{HH} is set at the lowest value of the applicable soil quality guidelines, though SQG_{FI} is considered to be a check mechanism if the chemical does not biomagnify. This ensures that the final SQG_{HH} is protective of all these potential contaminant media transfer pathways. If there are insufficient data to calculate all of the applicable pathways, the SQG_{HH} can still be determined so long as the SQG_{DH} has been calculated; if the substance is known to biomagnify, the SQG_{FI} is also required. If data requirements for the SQG_{DH} are not met, then only a provisional SQG_{HH} can be developed (see Part D, Section 1.4).

6.2 Residential/Parkland Land Uses

The direct human health-based soil quality guideline (SQG_{DH}) is calculated using equations in Section 5.1 or 5.2, depending on whether the contaminant is a threshold or non-threshold contaminant. For residential/parkland land uses, the SQG_{IAQ} and SQG_{PW} are calculated as well and the final SQG_{HH} is set at the lower of the values generated.

For residential properties with backyard gardens, the check mechanism for contamination of produce grown on-site is calculated and presented in the contaminant assessment document for possible use as a site-specific objective.

If there are insufficient data to calculate all of the applicable pathways, the SQG_{HH} can still be determined so long as the SQG_{DH} has been calculated. If data requirements are not met for the SQG_{DH}, then only a provisional SQG_{HH} can be developed (see Part D, Section 1.4).

6.3 Commercial and Industrial Land Uses

The direct human health-based soil quality guideline (SQG_{DH}) is calculated using equations in Section 5.1 or 5.2, depending on whether the contaminant is a threshold or non-threshold contaminant. As with residential land use, the SQG_{IAQ} and SQG_{PW} are also calculated, and the SQG_{HH} is set as the lower of the values. Professional judgement should be used to determine whether the $SQG_{OM\text{-}HH}$ should be used to modify the SQG_{HH}.

If there are insufficient data to calculate all of the applicable pathways, the SQG_{HH} can still be determined so long as the SQG_{DH} has been calculated. If data requirements are not met for the SQG_{DH}, then only a provisional SQG_{HH} can be developed (see Part D, Section 1.4).

SECTION 1
DERIVATION OF THE FINAL SOIL QUALITY GUIDELINE

1.1 Final Guideline Derivation

The goal of the final recommended soil quality guideline (SQG_F) is to protect both ecological and human health.

The lower of the two guidelines obtained through the environmental procedure (SQG_E) (Part B) and the human health based procedure (SQG_{HH}) (Part C), will be recommended as the final soil quality guideline (SQG_F), for each land use subject to restrictions discussed in Section 1.2 below. A general overview of the entire guidelines derivation process outlining the major steps leading to derivation of the final soil quality guideline is illustrated in Figure 21.

If either the SQG_{HH} or the SQG_E cannot be calculated, and the SQG_F is higher than the interim (1991) criterion, then the SQG_F will be set at the 1991 criterion.

Development of Canadian soil quality guidelines is complex and involves many parameters. While some parameters are known with great precision, most of them are estimates with considerable variability. In consideration of this and other uncertainties in the guideline development process (see Section 6, Part B), SQG_F are rounded to not more than two significant figures for presentation in assessment documents.

In some cases, a pathway-specific guideline exceeding 1,000,000 mg/kg (i.e. a concentration exceeding 100% by weight) may be calculated; in this case, the guideline for that pathway is reported as “NA”.

1.2 Considerations Other than Toxicity

Contaminants may have adverse effects in addition to producing toxic responses in human and ecological receptors. These may include aesthetic concerns (e.g. odours), explosive hazards, free-phase liquid formation, or damage to utilities and infrastructure.

If there is evidence that a contaminant may cause significant environmental effects beyond toxicity to human and ecological receptors, then this evidence should be evaluated. A soil quality guideline for management considerations (SQG_M) is developed to reflect any additional concerns associated with the contaminant.

There may be considerable uncertainty in the development of the SQG_M, and for some concerns associated with contaminants only a qualitative evaluation may be possible. Therefore, professional judgement should be used as to whether the SQG_F should be adjusted based on the SQG_M.

Certain contaminants may potentially degrade into more toxic or more mobile chemicals (e.g., degradation of trichloroethylene to vinyl chloride). Since degradation rates are affected by several site-specific factors, at this time soil quality guidelines are not adjusted to reflect degradation into more toxic compounds. However, major degradation products should be highlighted in the scientific supporting document.

1.3 Evaluation Against Plant Nutritional Requirement, Geochemical Background and Practical Quantitation Limits

SQGTG believes that guidelines should be reasonable, workable and usable. Guidelines are developed by applying scientifically derived information, backed by professional judgement where data gaps occur. Occasionally, defined exposure-based procedures produce numerical guidelines that conflict with one or more of the following:

- plant nutritional requirements;
- geochemical background;
- practical quantitation limits;

When a conflict of this type occurs, guidelines must be adjusted as described below:

Some chemicals (e.g., copper and zinc) considered hazardous at high levels also provide minimum nutritional requirements for the maintenance of plant growth at lower levels. SQGTG acknowledges that the SQG_F determined for these chemicals may fall below the nutritional requirements. For agricultural and residential/parkland land uses maintenance of nutritional requirements is critical to sustaining the primary activity on these lands (i.e., growing crops, grass, trees). Accordingly, SQGTG recommends that the SQG_F for these land use categories be compared to minimum plant nutritional requirements. If the SQG_F is below acceptable minimum plant nutritional requirement levels, then insufficient nutritional requirements for plant growth may result at the value of the SQG_F. The SQG_E should therefore default to the soil concentration required for minimum plant nutrition. This value does not apply to the commercial or industrial land use categories, because it is anticipated that the resulting SQG_F will be above plant nutritional requirements.

Where applicable, the SQG_F should also be compared to an acceptable geological (non-anthropogenic) background soil concentration to ensure the final value is not below background levels. The natural background concentration should represent a concentration that is typical of most unimpacted soils in Canada. Where the SQG_F is below the accepted geological background soil concentration, SQGTG recommends that the accepted background concentration replace the SQG_F generated using this protocol. It should be noted that although the SQG_F may be above natural background soil concentrations that are typical of most soils in Canada, there may be specific locations with unusually high natural background concentrations that still exceed the guidelines. In these cases, jurisdictions have the option to set site-specific guidelines that consider the unique geological characteristics of the particular locations.

Finally, a candidate SQG_F for a given substance should be checked against the current practical quantitation limit achievable in Canada. Where the candidate SQG_F is below the limit of practical quantitation (generally 5 times the analytical detection limit), a footnote should be added to the SQG_F stating “laboratories may not be able to reliably measure concentrations of this magnitude”. The SQG_F should not be adjusted based on the practical quantitation limit, however.

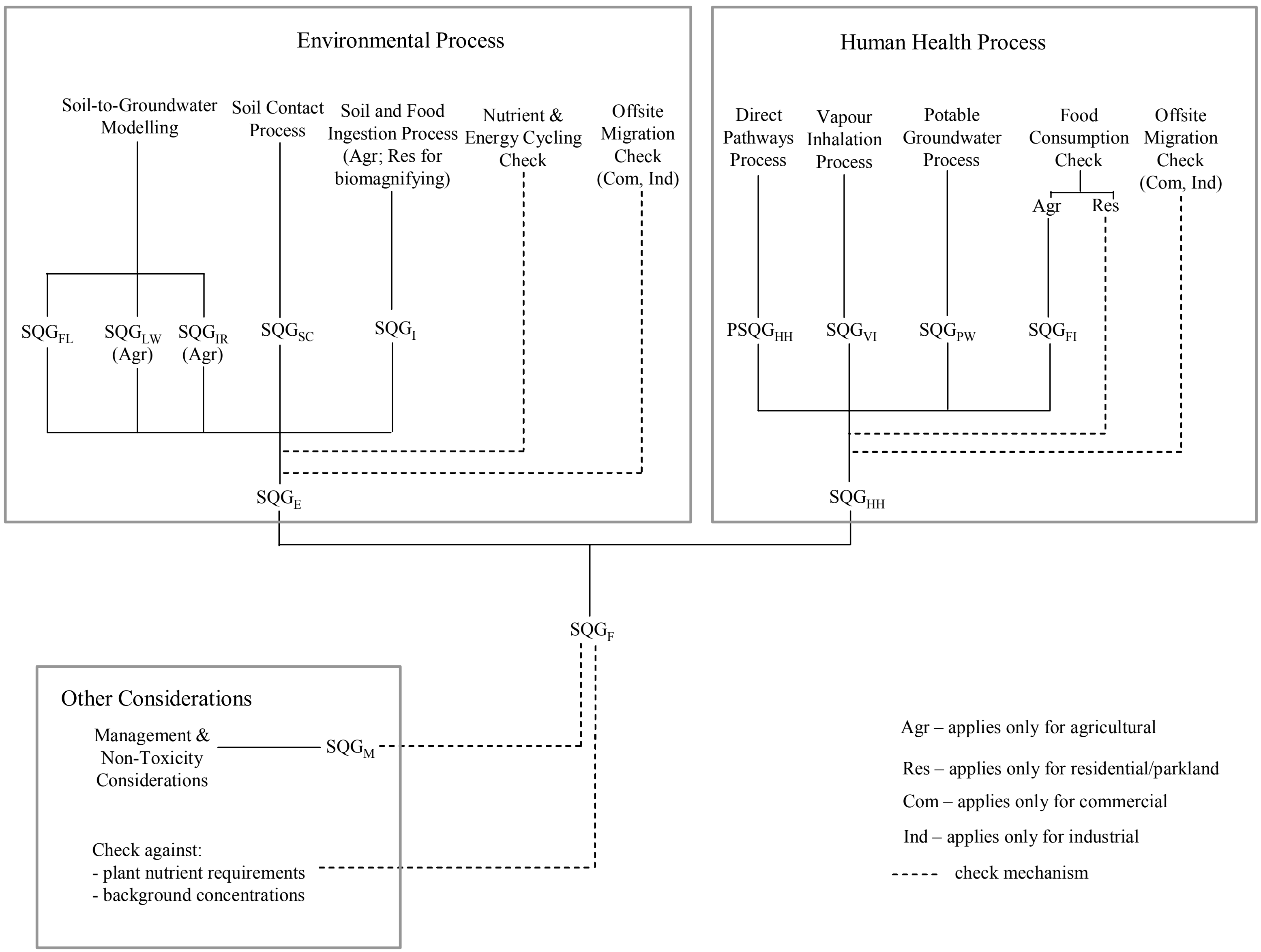


Figure 21: Overview of Steps Leading to Derivation of a Final Soil Quality Guideline

Because guidelines are based primarily on biological effects and background exposures are, wherever possible, incorporated into the procedures, it is anticipated that very few candidate SQG_F will require adjustment. Where any of the evaluation procedures described above does result in modification of a candidate SQG_F, this condition will be noted in the assessment document for the substance.

1.4 Provisional Guidelines

In order for a final soil quality guideline to be developed, guidelines for certain required pathways (see Section 8 in Part B and Section 6 in Part C) must be calculated. In some cases, though, it may not be possible to calculate a guideline for a particular required pathway, or it may not be possible to completely meet the data requirements for the calculation of a required pathway. However, literature searches often yield data that do not meet the requirements of the soil protocol, but still provide useful toxicity information. Also, toxicity tests using standard methodologies may produce data that do not meet the regular quality standards defined by toxicologists, due to difficulties in handling and evaluating certain substances such as volatile organic chemicals in the context of a soil contact test, for example.

While acknowledging the need for toxicity and exposure data of the highest quality, it is considered to be better to establish a guideline based on incomplete data than to not establish a risk-based guideline. In these cases, the SQG_E and SQG_{HH} are determined, but are designated as a "Provisional Guidelines" to reflect the uncertainty and data gaps in the guideline development. A guideline will also be designated as a Provisional Guideline if the EDI exceeds the TDI. If either the SQG_E or SQG_{HH} are provisional, then the SQG_F is also considered to be a provisional guideline.

The guiding principles for calculating the soil quality guidelines are still followed when developing provisional guidelines. However, since data requirements are relaxed, the following principles are followed:

- be precautionary; use higher safety factors where degree of uncertainty is high;
- keep in mind that provisional environmental soil quality guidelines for agricultural and residential/parkland land uses are intended to approximate no appreciable effect levels while those for commercial and industrial land uses allow for a low level of effects;
- provisional human health soil quality guidelines are intended to result in no appreciable risk to humans for all activities associated with the intended land use;
- be consistent with the spirit of the protocol.

If the provisional SQG_F is higher than an existing guideline, such as a 1991 interim soil criterion (if applicable), the previously existing guideline is retained as the SQG_F.

1.5 Presentation of Soil Quality Guidelines

The soil quality guidelines will be presented in tabular format, showing the guidelines developed for each pathway and the final soil quality guideline. An example is shown below (separate tables will be prepared for coarse and fine-grained soils, and for non-threshold substances for both 10^{-5} and 10^{-6} incremental risk levels):

Table 4 Example of Soil Quality Guideline Presentation

	Land Use			
	Agricultural	**Residential/ Parkland**	**Commercial**	**Industrial**
Guideline (SQG_F)	**##**	**##**	**##**	**##**
Human health guidelines/check values				
SQG_{HH} (or provisional SQG_{HH})	##	##	##	##
Direct contact (SQG_{DH})	##	##	##	##
Protection of indoor air quality – basement (SQG_{IAQ})	##	##	-	-
Protection of indoor air quality – slab-on-grade (SQG_{IAQ})	##	##	##	##
Protection of potable water (SQG_{PW})	##	##	##	##
Off-site migration check (SQG_{OM-HH})	-	-	##	##
Produce, meat and milk check (SQG_{FI})	##	##	-	-
Environmental health guidelines/check values				
SQG_E (or provisional SQG_E)	##	##	##	##
Soil contact (SQG_{SC})	##	##	##	##
Soil contact confidence rank	rank	rank	rank	rank
Soil and food ingestion (SQG_I)	##	## (or -)	-	-
Protection of freshwater life (SQG_{FL})	##	##	##	##
Livestock Watering (SQG_{LW})	##	-	-	-
Irrigation Water (SQG_{IR})	##	-	-	-
Nutrient and energy cycling check (SQG_{NEC})	##	##	##	##
Off-site migration check (SQG_{OM-E})	-	-	##	##
SQG_M (non-toxicity considerations)	##	##	##	##
Interim soil quality criterion (CCME 1991)	##	##	##	##

1.6 Scientific Supporting Documents

Scientific supporting documents are prepared in support of all soil quality guidelines. Separate supporting documents may be prepared for the environmental and human health guidelines, or a single combined document may be prepared.

The supporting documents should generally include the following sections:

- Background Information (physical and chemical properties, analytical methods, production and uses in Canada, sources and concentrations in various environmental media, and a summary of existing guidelines);
- Environmental Fate and Behaviour;
- Behaviour and Effects in Biota (environmental guidelines);
- Behaviour and Effects in Humans and Mammalian Species (human health guidelines);
- Derivation of Environmental Soil Quality Guidelines;
- Derivation of Human Health Soil Quality Guidelines; and
- Data Gaps and Uncertainties

The supporting documents are summarized in fact sheets which are included with the published Canadian Environmental Quality Guidelines.

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APPENDIX A
PARTITIONING OF CONTAMINANTS BETWEEN SOIL, PORE WATER, AND SOIL VAPOUR

1.0 Development of Partitioning Equations

Contamination in soil includes contaminants adsorbed to soil particles, dissolved in soil pore water, and in vapour phase in soil gas. In the absence of free-phase contaminants, the total concentration of a contaminant in a soil sample is defined by the equation (US EPA, 1996):

$$C_t = \frac{C_s\rho_b + C_w\theta_w + C_a\theta_a}{\rho_b} \qquad [1]$$

where
C_t = total concentration of contaminant in soil (mg/kg)
C_s = concentration of contaminant adsorbed to soil particles (mg/kg)
C_w = concentration of contaminant in aqueous phase (mg/L)
C_a = concentration of contaminant in vapour phase (mg/L)
ρ_b = soil dry bulk density (kg/L)
θ_w = water-filled porosity (L-water/L-soil)
θ_a = air-filled porosity (L-air/L-soil)

It should be noted that the above equation is based on the assumption that soil, pore water and soil gas are included in the soil sample. If soil gas is not preserved during sampling, θ_a can be assumed to be zero (US EPA, 1996). It should also be noted that metals may be present in a mineral phase, which is not normally extracted and measured in environmental analyses.

Numerous studies have shown that sorption of organics by soils is highly correlated with the organic matter content (e.g., Chiou et al. 1979, Hassett et al. 1980). Chiou (1989) presents evidence that the linearity of sorption with organic contaminant concentration and correlation with soil organic matter content reflect dissolution of the organic contaminant into the soil organic matter phase–as opposed to sorption to organic matter surfaces. Normally a Freundlich isotherm is fitted to the sorption data:

$$C_s = K_d \times C_w^{1/n} \qquad [2]$$

where
K_d = distribution coefficient
n = empirical constant

For most non-ionic organics n=1 and sorption is a linear function of equilibrium solution concentration up to 60% to 80% of its water solubility (Hassett and Banwart 1989). The relationship between solution concentrations and the *sorbed* concentration is described in Equation 1.

Likewise, C_a can be determined from C_w and the unitless Henry's Law constant (H') (US EPA, 1996):

$$C_a = C_w H'$$ [3]

For most inorganic compounds (excluding mercury), vapour pressure is negligible, and H' can be assumed to be 0.

By substituting equations 2 and 3 into equation 1 and re-arranging, an equation can be derived for the relationship between the total concentration of the contaminant in soil and C_w:

$$C_t = C_w \left(K_d + \frac{\theta_w + H'\theta_a}{\rho_b} \right) \qquad [4]$$

Equation 4 is used as the partitioning relationship between soil and soil pore water for developing groundwater protection guidelines.

A similar relationship can be derived to describe partitioning from soil to soil vapour for the protection of indoor air quality pathway, by substituting the relationship in Equation 3 for C_w in Equation 4 and re-arranging:

$$C_a = \frac{C_t H' \rho_b}{\theta_w + K_d \rho_b + H'\theta_a} \qquad [5]$$

2.0 K_d for Non-Dissociating Organic Contaminants

For non-dissociating organic contaminants, K_d has been shown to be related to the soil organic carbon content:

$$K_d = K_{oc} f_{oc} \qquad [6]$$

where K_{oc} = organic carbon partitioning coefficient (L/kg)
f_{oc} = organic carbon fraction of soil (g/g)

This relationship does not hold for soil with very low organic carbon, where adsorption to mineral surfaces may become significant. The organic carbon content at which this happens is dependent on both the chemical and soil properties (US EPA, 1996); for organic carbon contents less than 0.001, site-specific evaluation may be necessary.

3.0 K_d for Dissociating Organic Contaminants

Equilibrium partitioning isotherms effectively describe the behaviour of non-dissociating organic contaminants in soils. This description may be extended to dissociating organic contaminants provided sorption of both the dissociated and non-dissociated forms is understood and easily treated.

These conditions are met for some weak organic acids, such as chlorophenols, because only the non-dissociated form is appreciably sorbed. Like many other anions, the phenate generated by the dissociation of the parent chlorophenol is mobile in soils. Because of this difference, chlorophenol partitioning can be predicted from the concentration of the non-ionized form, which is a function of pH. The pH-dependent distribution coefficient can then be calculated (Schellenberg et al. 1984) as the product of the partitioning coefficient for the chlorophenol and the proportion of the non-ionized form:

$$K_d = K_{oc} \times F_{oc} \times Q \qquad [7]$$

where
K_{oc} = organic carbon-normalized coefficient for non-ionized chlorophenol
F_{oc} = fraction of organic carbon in soil
Q = proportion of chlorophenol in non-ionized form

It is important to note that experimental data that nominate K_{oc} have been referenced to the concentration of non-dissociated chlorophenol.

Q is derived from the equilibrium acidity expression for the chlorophenol:

$$Q = 1/(1 + K_a/[H^+]) \qquad [8]$$

where K_a = acidity constant

Substituting [7] and [8] into [4]:

$$C_t = C_w \left(\frac{K_d}{1 + K_a/[H^+]} + \frac{\theta_w + H'\theta_a}{\rho_b} \right)$$

The additional information required to calculate the total soil concentration of a weak acid contaminant in equilibrium with the desired water quality is therefore:

- soil pH;
- acidity constant of the contaminant, and
- partition coefficient for the non-ionized acid.

Organic contaminants that protonate to cationic forms (e.g., amines) cannot be accommodated by the above treatment because, in soils, cations are competitively sorbed on colloids, which vary with soil type.

4.0 K_d for Inorganic Chemicals

Partitioning of metals between soil and water is dependent on several factors, including pH, cation exchange capacity, iron oxide content, and oxidation-reduction conditions. As a result, the K_d for metals is difficult to determine; reported values for individual metals can vary over 5 orders of magnitude. Therefore, it is not considered appropriate at this time to develop generic nation-wide guidelines for metals based on partitioning relationships.

Where groundwater pathways may be of concern for sites contaminated by metals, these pathways should be addressed on a site-specific basis; this would likely include measurement of metals in groundwater at the source and/or at the point of exposure.

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APPENDIX B
NUTRIENT AND ENERGY CYCLING CHECK

1.0 Using Soil Nutrient and Energy Cycling Data to Derive Effects-based Guidelines

Soils are dynamic, open, systems characterized by fluxes of energy and nutrients. The ability of soils to support plant life depends on the coordinated activities of a myriad of invertebrates and microorganisms that mediate nutrient and energy cycles. Decomposition, respiration, and organic nutrient cycles are examples of measurable soil processes whose rates may be adversely affected by contaminants.

In theory, these processes should be good indicators of soil quality. In practice, however, it is difficult to integrate currently available literature into the derivation process because of:

- uncertainties in interpretation of results, including variability seen in dose-response relationships (Dennemen and van Gestel 1990),
- frequent lack of a reference toxicant, and
- uncertainty over appropriate controls ("acceptability criteria") (Environment Canada 1992).

On the other hand, soil process data have the desirable property of ecological relevance -- the use of these measures as indicators or predictors of ecosystem performance is well established (see Paul and Clark 1989). Furthermore, data on effects of common contaminants on some soil microbial processes are abundant (see, for example, Bååth 1989). A balance is struck among these considerations by using qualifying nutrient and energy cycling data as a check against preliminary soil quality guidelines derived from single-species bioassay.

The microbial ecology relevant to the cycling of organic nutrients indicates that contaminant data from nitrogen fixation, nitrification, nitrogen mineralization, decomposition, and respiration studies are all potentially acceptable for use in a checking role against guidelines derived from single species bioassay (See Section 2.0). Of these, N-fixation and nitrification data are preferred, but carbon cycling and nitrogen mineralization measures may be used when the former are unavailable or insufficient for guideline derivation.

2.0 Nutrient and Energy Cycling Processes as Indicators of Soil Quality

Because biological activity in soils is dominated by the detrital food chain, energy flow is closely linked to carbon cycling, which is normally observed through measurements of decomposition and respiration. Other elements whose cycling rates in soils are mainly functions of biological activity include the plant macroelements nitrogen, sulphur and phosphorus. Activities of microorganisms are primarily responsible for liberating these elements from organic forms (mineralization), and making them available for uptake by plants.

McGill and Cole (1981) reviewed patterns of cycling of carbon, nitrogen, sulphur, and phosphorus in soils. They concluded that nitrogen and to a lesser extent, sulphur transformations in soil were closely linked to the energy needs of heterotrophs searching for organic carbon. Organic phosphorus and a portion of organic sulphur, on the other hand, is mineralized by extracellular enzymes (phosphatases and sulphatases) partly in response to biological demand for this element. As a result of these different mechanisms, phosphatase and sulphatase activity varies with the phosphorus and sulphur status of the soil (Spiers and McGill 1979, Maynard et al. 1984), and is a source of variability in any potential bioassay. Moreover, some enzymatic activity may be stabilized in soil outside the microbial cell (Stewart and Tiessen 1987), adding further uncertainty to the interpretation of sulphur and phosphorus mineralization rates or estimates of phosphatase and sulphatase activity. Mineralization rates of phosphorus and sulphur are therefore not good candidates to report on soil quality in relation to contaminants. Carbon and nitrogen transformations, however, are well suited to report on soil quality in relation to contaminants.

Rates of decomposition (mass loss of organic substrates) and respiration (CO_2 evolution) are common measures of carbon cycling and are potentially useful for assessing contaminant effects. These measures integrate the activities of soil heterotrophs and therefore report on community-level performance. This integration indicates a high degree of ecological relevance. However, respiration also reflects functional redundancy since many heterotrophic organisms carry out the same process during catabolic energy generation (Paul and Clark 1989). Biological redundancy in respiration and decomposition is responsible for relatively high levels of resistance and resilience to stressors such as contaminants. Therefore, at contaminant levels sufficient to inhibit respiration or decomposition, it is likely that significant impacts have occurred in at least a segment of the heterotrophic community. Conversely, such impacts are likely to be mitigated by compensating actions by resistant, re-colonizing or physically-protected organisms. Respiration and decomposition are therefore expected to have limitations as indicator processes for contaminant effects.

Nitrogen cycling in soils is complex and involves a broad range of soil organisms. Some nitrogen cycling processes such as mineralization, immobilization and denitrification (Figure A.1) are carried out incidentally by generalists during the catabolism of carbon-rich substrates (McGill and Cole 1981). These processes share the same limitations for respiration and decomposition. Other nitrogen cycling processes are carried out by more specialized organisms with narrower ecological amplitudes.

Nitrogen fixation, the process by which nitrogen is added to soils from atmospheric N_2, is carried out only by a very limited range of specialized bacteria. The root nodule symbionts *Rhizobium species* and *Frankia species* are particularly important. Nitrification, the oxidation of ammonium to nitrate, is performed by only a few genera of chemoautotrophic bacteria. It is important in soil because the mobility of nitrate allows plants to acquire large amounts of nitrogen — poorly mobile in other chemical forms — in the mass flow of water to roots. In a given soil, often only two species of nitrifiers are involved.

The nature of the energy metabolism of both nitrogen-fixers and nitrifiers make them susceptible to stressors. Nitrogen-fixers require large amounts of energy, in a micro-anaerobic environment, to chemically reduce N_2 to ammonium. Cellular apparatus required to maintain these conditions places a heavy energy demand on the cell that can be met only under favourable conditions — including minimal contaminant stress. Conversely, nitrifiers maintain life on a very low energy budget dictated by the small amount of energy available from oxidation of ammonium to nitrate. This low energy yield limits growth rates and makes nitrifiers sensitive to stress conditions (Schmidt 1982).

For these reasons, nitrogen-fixers and nitrifiers are ecologically relevant, sensitive to a broad range of stressors, and are functionally unique (i.e., when lost or damaged their functions cannot be overtaken by related species). These properties, in turn, make them good microbial indicators of soil quality.

Based on these findings, contaminant data from nitrification, nitrogen fixation, nitrogen mineralization, decomposition and respiration studies are all potentially acceptable for checking against a single species bioassay. Of these, nitrogen-fixation and nitrification data are preferred, but carbon cycling and nitrogen mineralization measures may be used when the former are unavailable or insufficient.

3.0 Evaluation of Laboratory and Field Toxicological Data

In general, the guiding principles for selecting acceptable bioassay studies (described in Part B, Section 7.2) also apply to the evaluation of laboratory and field toxicological data. While controlled, laboratory data are preferred, field data may be used provided influential variables are measured and within acceptable ranges. Acceptable data sources will include:

- a replicated and controlled statistical design,
- a known and reported exposure period, and
- analytical assessment of contaminant test concentrations.

Minimum toxicological data requirements also apply to soil nutrient and energy cycling process data (see Part B, Section 7.3). Minimum requirements vary within the hierarchy of derivation methods described in Section 4.0 below.

4.0 Deriving Soil Quality Guidelines

A tiered, or hierarchical approach to guideline derivation is presented that considers: land use, data sources and derivation method. Four derivation methods {the weight of evidence, LOEC extrapolation, and median effects methods (described in Part B, Section 7.5); and a modified LOEC method applied to nitrogen and carbon cycling data} are used to calculate the soil quality guideline for the protection of nutrient and energy cycling (SQG_{NEC}).

4.1 Agricultural and Residential/Parkland Land Uses

4.1.1 Nitrification and Nitrogen-fixation Data

Using nitrification and N-fixation data, the weight of evidence, LOEC and median effects methods are applied (see Part B, Sections 7.5.4 to 7.5.6) with the exception that quantity requirements for invertebrate and plant studies are replaced by required balance between N-fixation and nitrification studies. In addition, single dose studies (control plus one level of contaminant) are acceptable as a source of "pseudo-LOEC" data if the effects dose does not exceed a 40% response for nitrogen fixation and nitrification processes.

4.1.2 Decomposition, Respiration and Nitrogen Mineralization Data

If data are insufficient to generate a guideline check value on the basis of N-fixation and nitrification, C cycling and N mineralization data can be used to supplement or, less desirably, replace these data in modified LOEC or median effects methods.

In the modified LOEC method, a guideline is derived from the geometric mean of LOECs from a minimum of 3 studies (as before) but with the following conditions. First, because C cycling and N mineralization measurements are expected to be less sensitive than N-fixation and nitrification data (see Section 2.0 in this appendix), unrestricted use of LOEC data is not recommended. In particular, LOEC values for C cycling and N mineralization should not exceed a response. Second, "single dose" studies (control plus one level of contaminant) are acceptable as a source of "pseudo-LOEC" data if the effects dose does not exceed a 40% response for N-fixation and nitrification or a 25% response for C cycling and N mineralization data. Although data of the latter type do not meet all formal requirements for reporting a LOEC, they are nevertheless considered scientifically and technically defensible in consideration of the restrictions imposed, peer-reviewed status, and importance of the biological processes involved.

If only the minimum number of LOEC studies are available, professional judgement should be used to assess whether the resulting microbial value represents an accurate measure of potential process-level effects, otherwise, this method may be discarded and the median effects method (using microbial EC_{50} values, see Part B, Section 7.5.6) applied with an application factor of 5. The median effects method is applied in the same way if the requirements for the modified LOEC method cannot be met. If data are insufficient for any of the methods above, no SQG_{NEC} will be generated. Data gaps and areas for further research will be noted.

4.2 Commercial and Industrial Land Use

4.2.1 Nitrification and Nitrogen-fixation Data

The weight of evidence method or LOEC method are applied (see Part B, Section 7.5) with the exception that quantity requirements for invertebrate and plant studies, are replaced by a required balance between nitrogen-fixation and nitrification studies. In addition, single dose studies (control plus one level of contaminant involving nitrogen fixation and nitrification) are acceptable as a source of "pseudo-LOEC" data if the effect dose does not exceed a 50% response.

4.2.2 Decomposition, and Respiration and Nitrogen Mineralization Data

If data are insufficient to generate a guideline check value on the basis of N-fixation and nitrification, C cycling and N mineralization data can be used to supplement or, less desirably, replace these data in modified LOEC or median effects methods.

In the modified LOEC method, a guideline is derived from the geometric mean of LOECs from a minimum of 3 studies (as before) but with the following conditions. First, because C cycling and N mineralization measurements are expected to be less sensitive than N-fixation and nitrification data (see Section 2.0 in this appendix), unrestricted use of LOEC data is not recommended. In particular, LOEC values for C cycling and N mineralization should not exceed a response. Second, "single dose" studies (control plus one level of contaminant) are acceptable as a source of "pseudo-LOEC" data if the effects dose does not exceed a 50% response for N-fixation and nitrification or a 35% response for C cycling and N mineralization data. Although data of the latter type do not meet all formal requirements for reporting a LOEC, they are nevertheless considered scientifically and technically defensible in consideration of the restrictions imposed, peer-reviewed status, and importance of the biological processes involved.

If only the minimum number of LOEC studies are available, professional judgement should be used to assess whether the resulting microbial value represents an accurate measure of potential process-level effects, otherwise, this method may be discarded and the median effects method (using microbial EC_{50} values, see Part B, Section 7.5.6) applied with an application factor of 5. The median effects method is applied in the same way if the requirements for the modified LOEC method cannot be met. If data are insufficient for any of the methods above, no SQG_{NEC} will be generated. Data gaps and areas for further research will be noted.

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APPENDIX C
MODEL FOR THE PROTECTION OF GROUNDWATER FROM SOIL CONTAMINATION

1.0 Background

Soil contamination may migrate into groundwater. The groundwater may in turn be intercepted by water wells or dugouts, or be transported to nearby surface water bodies. In order to ensure that potential use of groundwater at a remediated site and the health of nearby aquatic environments are not affected, protection of groundwater is included in the guideline development process.

Specifically, guidelines are developed for the following pathways:

- protection of potable (drinking) water;

- protection of groundwater for agricultural uses (livestock watering and irrigation);

- protection of freshwater life in nearby surface water.

Many models exist that describe the transport dynamics of contaminants in soils and groundwater (e.g. Rao and Jessup, 1983; Jury and Godhrati, 1989; Korfiatis et al., 1991; Piver and Lindstrom, 1991 and Toride et al., 1993 among others). Most of these models are theoretically rigorous, mechanism-based descriptions that focus on advective and dispersive processes but sometimes also incorporate biodegradation and fate of derived products. Mathematically, they generally involve complex partial differential equations that can be solved only under restrictive boundary and continuity conditions. Numerical solutions supported by sophisticated computer codes are usually needed. These models have been developed for research purposes or for application to specific field sites and are generally considered to be scientifically sound.

The above approaches could form a basis for the estimation of a generic dilution factor. However, mechanistic models are extremely parameter intensive, and generally require parameters that are not readily available or can be measured or estimated only with difficulty. Furthermore, these models are generally abstract and complex, making them poorly accessible to many contaminated sites stakeholders.

For generic guideline development, an approach that was simple, practical, effective and transparent was desired. Specifically, an appropriate model for generic guideline development should:

- be clearly documented and easily understood;
- be scientifically defensible;
- require parameters that can be derived from readily available sources;
- include only the most influential parameters;
- apply to Canadian conditions; and
- be tuned to the appropriate temporal and spatial scales.

The groundwater pathways are addressed using a groundwater model developed by the British Columbia Contaminated Sites Soil Taskgroup (CSST), based on the US EPA (1996) draft Soil Screening Guidance and using saturated groundwater transport equations developed by Domenico and Robbins (1985). The model is based on one-dimensional groundwater flow, and incorporates a variety of mechanisms including dispersion, biodegradation, adsorption-desorption, and dilution of leachate into groundwater. All equations used in the model are included in Appendix H.

The model has four components:

1. Partitioning of the contaminant between soil, soil vapour and soil pore water.

2. Leaching of the contaminant through the unsaturated zone to the groundwater table.

3. Mixing and dilution of the leachate into groundwater.

4. Saturated-zone transport of the contaminant to a downgradient receptor.

It should be noted that not all of these components will apply in every scenario. Specifically, the unsaturated zone transport (component 2) only applies if the contamination is not in contact with groundwater, and is therefore not applied in generic guideline development. Also, the saturated-zone transport (component 4) only applies if there is a lateral separation between the remediated site and the groundwater receptor; for the development of generic guidelines it is assumed that a water well or livestock dugout could be installed at the edge of (or even within) the boundaries of the remediated area.

2.0 Assumptions

Several assumptions are incorporated into the model (adapted from CCME, 2000):

- the soil is physically and chemically homogeneous;
- the groundwater aquifer is present in unconsolidated mineral soils (not fractured bedrock);
- the moisture content is uniform throughout the unsaturated zone;
- the infiltration rate is uniform throughout the unsaturated zone;
- decay of the contaminant source is not considered (i.e. infinite source mass);
- flow in the unsaturated zone is assumed to be one-dimensional and downward only (vertical recharge) with dispersion, sorption-desorption, and biological degradation;
- the contaminant is not present as a free product phase (non-aqueous phase liquid);
- the groundwater aquifer is unconfined;
- groundwater flow is uniform and steady;
- co-solubility and oxidation/reduction effects are not considered;
- attenuation of the contaminant in the saturated zone is assumed to be one-dimensional with respect to sorption-desorption, dispersion, and biological degradation;
- dispersion is assumed to occur in the longitudinal and transverse directions only (no vertical dispersion) and diffusion is not considered;
- mixing of the leachate with the groundwater is assumed to occur through mixing of leachate and groundwater mass fluxes; and

- dilution of the plume by groundwater recharge down-gradient of the source is not included.

3.0 Calculation of Soil Quality Guidelines for the Protection of Groundwater

Calculation of the soil quality guidelines for the protection of potable groundwater (SQG_{PW}), freshwater life (SQG_{FL}), livestock watering (SQG_{LW}) and irrigation water (SQG_{IR}) is performed using the equations detailed in Appendix H. The allowable concentration of the chemical in groundwater at the receptor is the appropriate water quality guideline for the pathway. These calculations only apply for organic compounds due to the highly site-specific nature of partitioning for inorganic chemicals and the lack of generalized modelling techniques appropriate for inorganic substances. Generalized techniques for evaluating the partitioning and transport of inorganic appropriate for generic guidelines are not expected to be developed in the foreseeable future.

Soil/Leachate Partitioning

Partitioning of chemicals between soil and soil pore water (leachate) is detailed in Appendix A. The equation below is used to describe the partitioning relationship:

$$SQG_{GW} = C_L \left\{ K_d + \left(\frac{\theta_w + H'\theta_a}{\rho_b} \right) \right\}$$

SQG_{GW}	=	soil quality guideline for the protection of groundwater (mg/kg) (i.e. SQG_{PW}, SQG_{FL}, SQG_{IR}, SQG_{LW})
C_L	=	allowable leachate concentration at source (mg/L) – calculated below
K_d	=	distribution coefficient (cm^3/g) – see Appendix A
θ_w	=	water filled porosity (unitless)
H'	=	dimensionless Henry's Law constant = H x 42.32
H	=	Henry's Law constant (atm-m^3/mol)
θ_a	=	air-filled porosity (unitless)
ρ_b	=	soil bulk density in contaminant partitioning zone (g/cm^3)

Unsaturated Groundwater Zone

The equation describing transport of the chemical through the unsaturated groundwater zone is included below for completeness. However, generic guidelines are developed based on the assumption that the contamination could be in direct contact with groundwater. Therefore, this process is not normally considered in the development of generic guidelines (i.e. the concentration of the chemical in leachate at the water table is equal to the concentration of the chemical in leachate at the source).

$$C_L = \frac{C_z}{\exp\left[\dfrac{b}{2\partial_u} - \dfrac{b}{2\partial_u}\left(1 + \dfrac{4\partial_u L_{US}}{v_u}\right)^{1/2}\right]}$$

$$v_u = \frac{I}{\theta_w R_u}; \quad R_u = 1 + \frac{\rho_b}{\theta_w} K_d$$

C_L	=	allowable chemical concentration in leachate at the source (mg/L)
C_z	=	allowable chemical concentration in leachate at the water table (mg/L) calculated below
b	=	thickness of unsaturated zone below the source (m) = d – Z
d	=	depth from surface to groundwater surface (m)
Z	=	depth to bottom of contaminated soil (m)
∂_u	=	dispersivity in the unsaturated zone (m) = 0.1b
L_{US}	=	decay constant for chemical (y^{-1}) in unsaturated zone:

$$L_{US} = \frac{0.693}{t_{1/2US}}\left(e^{-0.07d}\right)\left(1 - \frac{D_{1/2US}}{365}\right)$$

$t_{1/2US}$	=	chemical half-life in unsaturated zone (years)
$D_{1/2US}$	=	days with temperature <0^{o}C
v_u	=	average linear leachate velocity (m/y)
I	=	infiltration rate (m/y) = precipitation minus runoff and evapotranspiration
θ_w	=	water-filled porosity (unitless)
R_u	=	retardation factor in unsaturated zone (unitless)
ρ_b	=	soil bulk density in unsaturated zone (g/cm^3)
K_d	=	distribution coefficient (cm^3/g) – see Appendix A

Groundwater Mixing Zone

The mixing zone unsaturated/saturated equation (below), used to represent dilution of the leachate into groundwater, is based on a mass-balance approach considering movement of the chemical into the groundwater beneath the source (via infiltration of leachate) and away from the source area (via aquifer flow).

$$C_z = C_{gw}\left\{1 + \left(\frac{Z_d K_H i}{IX}\right)\right\}$$

C_z	=	allowable chemical concentration in leachate at the water table (mg/L)
C_{gw}	=	allowable chemical concentration in groundwater at the source (mg/L) – calculated below
Z_d	=	average thickness of mixing zone (m) – calculated below

K_H	=	hydraulic conductivity in the saturated zone (m/y)
i	=	hydraulic gradient (unitless)
I	=	infiltration rate (m/y) = precipitation minus runoff and evapotranspiration
X	=	length of source parallel to groundwater flow (m)

The equation is based on the assumption that the chemical is distributed evenly throughout a "mixing zone". While in reality the concentration of the chemical would not be constant throughout this zone, further vertical mixing would be expected to occur at the point of exposure (water well, dugout or surface water body). Therefore, the mixing zone approach is considered to be a reasonable approximation for purposes of generic guideline development.

The mixing depth is calculated using the equation in below, which considers both vertical dispersion of the contamination along the length and the source area and mixing due to the downward velocity of infiltrating leachate. It should be noted that the equation can, under certain circumstances, calculate a mixing zone thickness greater than the aquifer thickness (assumed to be 5 m for generic guideline development); if this occurs, the mixing depth should be set at the aquifer thickness.

Calculation of average thickness of mixing zone:

$$Z_d = r + s$$

r	=	mixing depth available due to dispersion and diffusion (m)
	=	0.01 X
X	=	length of source parallel to groundwater flow (m)
s	=	mixing depth available due to infiltration rate and groundwater flow rate (m)

$$s = d_a \left\{ 1 - e^{-\frac{2.178 XI}{K_H i d_a}} \right\}$$

d_a	=	depth of unconfined aquifer (m)
I	=	infiltration rate (m/y) = precipitation minus runoff and evapotranspiration
K_H	=	hydraulic conductivity in the saturated zone (m/y)
i	=	hydraulic gradient (unitless)

Saturated Zone Transport

The groundwater model includes the Domenico and Robbins analytical equation to evaluate lateral transport to a downgradient receptor. The implementation of this model presented below assumes no vertical dispersion downgradient of the source area. This assumption is "realistic" (doesn't significantly affect model results) for situations where the contaminant has mixed through the entire thickness of the aquifer or where there is a relatively large mixing depth and relatively short distance to the receptor, such as the default fine-grained soil scenario, and is conservative in other situations.

$$C_w(x,y,z,t)=\left(\frac{C_{gw}}{4}\right)\exp\left\{\left(\frac{x}{2\partial_x}\right)\left[1-\left(1+\frac{4L_s\partial_x}{v}\right)^{1/2}\right]\right\}erfc\left[\frac{x-vt\left(1+\frac{4L_s\partial_x}{v}\right)^{1/2}}{2(\partial_x vt)^{1/2}}\right]$$

$$\left\{erf\left[\frac{(y+Y/2)}{2(\partial_y x)^{1/2}}\right]-erf\left[\frac{y-Y/2}{2(\partial_y x)^{1/2}}\right]\right\}$$

$$V=Ki;\quad v=\frac{Ki}{n_e R_f};\quad R_f=1+\frac{\rho_b}{n}K_d$$

C_w	=	allowable chemical concentration in water at receptor (mg/L) (i.e. drinking water guideline or source guidance value for groundwater, FL guideline, irrigation water guideline, livestock watering guideline as appropriate)
x	=	distance from source to receptor (m)
x,y,z	=	Cartesian coordinates relating source and receptor (m); y, z assumed to be 0
t	=	time since contaminant release (years)
C_{gw}	=	allowable chemical concentration in groundwater at source (mg/L)
∂_x	=	longitudinal dispersivity tensor = 0.1x
∂_y	=	lateral dispersivity tensor = $0.1\partial_x$
L_s	=	decay constant (y^{-1}) in saturated zone:

$$L_s=\frac{0.693}{t_{1/2S}}\left(e^{-0.07d}\right)$$

d	=	depth from surface to groundwater surface (m)
$t_{1/2S}$	=	biodegradation half-life (y)
v	=	velocity of contaminant (m/y)
K_H	=	hydraulic conductivity in the saturated zone (m/y)
i	=	hydraulic gradient (unitless)
n	=	total porosity of soil = 1 - ρ_b/2.65 (unitless)
n_e	=	effective soil porosity (unitless)
Y	=	source width (m) perpendicular to groundwater flow
R_f	=	retardation factor (unitless)
ρ_b	=	soil bulk density in saturated zone (g/cm^3)
K_d	=	distribution coefficient (cm^3/g) – see Appendix A

Generic guidelines developed for the protection of potable water and the protection of water for agricultural uses are based on the assumption that a water well or dugout could be installed at the downgradient edge of the remediated area, in order to ensure that remediation to generic guidelines does not result in land or water use restrictions. Under this scenario, saturated zone transport is not considered (i.e. the concentration of the chemical in groundwater at the receptor is equal to the concentration in groundwater at the source). Jurisdictions may develop guidelines for these pathways incorporating offset distances if appropriate.

Guidelines developed for the protection of freshwater life in nearby surface water bodies include saturated zone transport. It is assumed for generic guidelines that the nearest surface water body is at least 10 m away from the remediated soils; if surface water bodies are located closer to the remediated soils than 10 m, then a site-specific evaluation may be necessary. The saturated zone transport model is not considered to be appropriate for use at distances less than 10 m.

The saturated zone transport model requires that a saturated zone biodegradation rate be defined. A conservative (high) value for the half-life in the saturated zone should be established based on a review of the literature. If the literature review indicates that biodegradation may not occur to significant extents under reasonably likely conditions, then a very large value (e.g. 100,000,000 days) should be applied in the model.

It should be noted that the saturated zone transport equation is time-dependent, and the maximum concentration of the chemical at the receptor would be expected to occur at some time in the future, depending on the groundwater velocity and the retardation of the chemical. Since source-depletion is not considered in the development of generic guidelines, the predicted concentration of the chemical at the receptor will eventually become stable. The default time was therefore set at a relatively large number (100 years) to ensure that the guidelines would be protective for most soluble chemicals.

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APPENDIX D
MULTIMEDIA EXPOSURE ASSESSMENT

Multimedia exposure assessment can be defined as the quantification of total concurrent exposure from all known or suspected sources via all known or suspected routes. Sources of exposure can be air, water, food, soil and consumer products. Routes of exposure can be inhalation, ingestion and dermal absorption.

The estimated daily intake (EDI) of a particular contaminant by an average Canadian can be obtained from a multimedia exposure assessment of background concentrations. The concept of estimated daily intake of a chemical for an individual can be defined as the sum of all exposures from various pathways and is represented by the following equation:

$$EDI = \sum_{i=1}^{n} ED_i \qquad [1]$$

Pathway-specific equations are used in exposure estimates. In comparison to hazard identification and dose-response assessment, exposure assessment has components that can be applied to all media:

ED_i is the exposure from pathway i,

$$ED_i = \frac{C \times CR \times BF \times EF}{BW} \qquad [2]$$

where,

ED	=	exposure dose (mg/kg-day)
C	=	contaminant concentration in medium (e.g., mg/L)
CR	=	contact rate (e.g., L/day)
BF	=	bioavailability factor (unitless)
EF	=	exposure factor which is the product of the exposure frequency (events/year) and exposure duration (years/lifetime) and is unitless
BW	=	body weight (kg)

The contact rate (CR) is medium-specific:

- Inhalation: CR = air inhalation rate (m^3/d)
- Water ingestion: CR = water ingestion rate (L/d)
- Soil ingestion: CR = soil ingestion rate (kg/d)
- Food ingestion: CR = food ingestion rate (kg/d); exposure are calculated for each type of food and summed

- Dermal contact: CR = soil dermal contact rate (kg/d) – see Appendix H

Total multimedia exposure is simply the sum of the dose estimated from air, water, soil, food (and consumer products where appropriate).

The four phases to quantifying exposure are:

A. **Data Collection**: Data on concentrations in air, water, soil, food, consumer products are combined with information on the rate of contact with air, water, and food to estimate exposure. Alternately, data on the concentrations in human tissues (i.e., hair, blood, urine, fat) can be combined with pharmacokinetic data to predict exposure rates. When no residue data are available because concentrations in the medium are below detection limit, the current detection limit will estimate the concentration in that medium. This would constitute a maximum value.

B. **Determination of Intake**: Receptors are classified into five groups defined by Health Canada based on age (i.e., newborn to 6 months, 7 months to 4 years, 5 to 11 years, 12 to 19 years, 20 years plus). In this document, the term child describes any individual five to 11 years of age. For each particular receptor group, quantities of air breathed, or water, food, and soils ingested, and surface area of exposed skin are required to convert a concentration in a contaminated medium (such as air or food) to a dose. Some basic assumptions routinely made about Canadians in risk assessments can be found in the CCME working document "Review and Evaluation of Receptor Characteristics for Multimedia Assessments". Figures for daily food consumption patterns in Canada by age group are available within the Health Protection Branch. Once data on the level of contamination are known for these foods, it is then possible to estimate the daily dose that these age classes might experience.

C. **Determination of Retention and Absorption**: It is often assumed that 100% of the dose ingested, inhaled or applied to the skin is bioavailable. However, where possible, more precise data on actual retention and absorption relative to retention and absorption from the exposure media for the primary toxicity study for the chemical are used to make the assessments as accurate as possible.

D. **Identification of Sensitive Receptors**: It is important to attempt to identify individuals (receptors) who may be at greater risk from exposure to a substance or at greater risk due to increased sensitivity to toxic effects. For example, the fetus and newborn are at greater risk to mercury and lead due to the sensitivity of the developing neurological system.

Often, data do not exist to permit the precise quantification of exposure. However, it may be known or suspected that the substance is present in the environment and some indication of exposure and risk is needed. In these cases, it is common to use predictive models to estimate levels in environmental media. Environmental fate and partitioning models have been developed to predict relative concentrations in water, soil, air and foods. These models rely on regression equations or other mathematical relationships which can predict environmental partitioning from simple chemical characteristics such as water solubility, octanol/water partition coefficients, Henry's Law Constant, and organic carbon partition coefficient. Rates of physical, chemical and biological degradability can also be predicted by some models.

Different models have been developed for different purposes. Environmental partitioning models are based on the assumption of chemical equilibrium between all media. However, deterministic models are required if one wants to predict the concentration of a substance at some specified distance from a point source of contamination. These models exist to assess chemicals released from incinerator stacks and from spills into soil or groundwater.

In foods, simple bioaccumulation or biomagnification factors were established for many chemicals. These will give a simplistic indication of the likelihood that a chemical will build up in fish, game or domestic animals and the approximate ratio of concentrations in the food to that in the environment. For example, fish BCFs are the ratio of the concentration in the fish (usually the flesh) to that in the water in which it was reared. Plant BCFs are the ratio of the concentration in the plant to that in the soil in which it was grown.

APPENDIX E
MIGRATION OF CONTAMINANT VAPOURS INTO BUILDINGS

The following section is adapted from CCME (2000).

1.0 Modelling Vapour Intrusion

Volatile organic compounds have migrated into homes from underground storage tanks leaking petroleum-based fuels, and from hazardous waste landfills where vinyl chloride was improperly disposed (Stephans *et al*., 1986). For volatile organic compounds, inhalation of vapours is often a dominant exposure pathway.

Vapours may be released from soil to the outside air at the ground surface, in addition to entering buildings. However, since buildings are enclosed spaces (and therefore have less air circulation than the outdoors), and buildings are often under-pressured due to heating (resulting in pressure-driven movement of soil gas into the building), migration of vapours into buildings poses a much greater health risk than migration of vapours to the outdoors.

Processes involved in the migration of vapours into buildings include:

- Partitioning of contaminants into soil gas (see Appendix A).
- Diffusion of vapour-phase contaminants through soil to the building slab.
- Advective flow of soil gas into a building due to pressure differences between the building and the external atmosphere.
- Diffusion of contaminants through soil-filled cracks in the building foundation.

Johnson and Ettinger (1991) provided one of the first screening level models to assess potential risks posed by the indoor infiltration of volatile contaminants emanating from soil and/or groundwater, and it has become a widely accepted work in this area. The model is described by the following equation:

$$\alpha = \frac{\left(\frac{D_T^{eff} A_B}{Q_B L_T}\right) \exp\left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}}\right)}{\exp\left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}}\right) + \left(\frac{D_T^{eff} A_B}{Q_B L_T}\right) + \left(\frac{D_T^{eff} A_B}{Q_{soil} L_T}\right)\left[\exp\left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}}\right) - 1\right]}$$

where:

α	=	contaminant concentration in building ÷ contaminant concentration in soil vapours (unitless)
D_T^{eff}	=	effective porous media diffusion coefficient (cm^2/s)
A_B	=	building area – floor and subgrade walls (cm^2)
Q_B	=	building ventilation rate (cm^3/s)

L_T = distance from contaminant source to foundation (cm)
Q_{soil} = volumetric flow rate of soil gas into the building (cm^3/s)
L_{crack} = thickness of the foundation (cm)
D^{crack} = effective vapour-pressure diffusion coefficient through the crack (cm^2/s)
A_{crack} = area of cracks through which contaminant vapours enter the building (cm^2)

A risk assessment modelling tool based on Johnson and Ettinger (1991) has been published and adopted by the US EPA (2003, 2002). A modified version of the Johnson and Ettinger model has been adopted within ASTM Standard PS 104-98 (Standard Provisional Guide for Risk-Based Corrective Action, RBCA) (ASTM, 1998) and ASTM Standard E 1739-95 (ASTM, 1995), and subsequently by the Atlantic Provinces PIRI initiative. Such models are routinely used in Canada and elsewhere for assessment of soil-borne volatile contaminants, particularly petroleum hydrocarbons.

Johnson and Ettinger (1991) demonstrated the mathematical rigour of their model by solving for a number of hypothetical, limiting situations. This work demonstrated that the solutions to these limiting cases agreed with what was anticipated theoretically. As yet there are insufficient data from field trials or controlled experimentation on full-scale buildings to ‘field validate’ the model. However, laboratory research has demonstrated the validity of various components, at least at bench scale.

The complete calculations for soil quality guidelines for the protection of indoor air quality are presented in Appendix H; default model parameters are summarized in Appendix I. For purposes of generic guideline development, steady state conditions are assumed, and depletion of the contaminant source is not considered.

2.0 Mass Transfer Phenomena Controlling Vapour Migration Through Soil

As mentioned, a modified version of the Johnson and Ettinger model has been adopted by ASTM (1998, 1995). The primary modification within RBCA is the omission of advective (also termed convective) vapour transport through cracks and spaces in the building envelope at Tier 1. Although all the Johnson and Ettinger equations (and quantification of the necessary variables) are provided within RBCA (ASTM, 1998), the standard assigns the critical variable for advective flow (Q_{soil}) a value of zero for the default case. This effectively restricts the model to diffusion-driven infiltration only. No explanation is provided within the RBCA documentation to rationalize or justify this modification. The earlier (ASTM, 1995) standard for petroleum release sites presented only a modified version of the Johnson and Ettinger equations excluding parameters related to advective flow. However, Nazaroff et al. (1985, 1987; cited in Johnson and Ettinger, 1991) report Q_{soil} values ranging from 280 cm^3/s to 2800 cm^3/s for indoor to outdoor barometric pressure differentials of 5 to 30 Pa (lower pressure indoors). Given that such pressure differentials are routinely observed in the range of 4 to 10 Pa (CMHC, 1997), then the default assumption of $Q_{soil} = 0$ is inappropriate in all default cases.

Numerous authors indicate that advective (pressure-driven) flow, which moves volatile contaminants from the soil-foundation interface into the living space of the building under a net negative barometric pressure differential (possibly due to wind effects, temperature differentials, appliance fans, stack effect, etc.), must be considered when quantifying the indoor infiltration and potential health risks of soil-borne volatile hydrocarbons (Johnson and Ettinger, 1991; CMHC, 1997; Williams et al., 1996; U.S.EPA, 2003; Hers and Zapf-Gilje, 1998; Little et al., 1992; and references therein).

Q_{soil} is dependent on the indoor to outdoor pressure differential and the soil permeability to vapour flow, as well as the depth below grade of the foundation and the length and radius of foundation cracks. Existing field data for coarse textured soils indicates that the value for Q_{soil} falls in the range of from 1 to 10 L/minute, and other jurisdictions are targeting the mid-point of this range (5 L/min) as an approximate minimum target value for Q_{soil} (USEPA, 2003). For the purposes of this document, model parameterization should result in a Q_{soil} value of 5 L/min or greater for coarse textured soils.

3.0 Indoor to outdoor pressure differential (ΔP)

One of the over-riding factors contributing to advective flow of volatile contaminants to the indoor environment is a net negative pressure differential in indoors, relative to out of doors. Indoor to outdoor barometric pressure differences have been investigated by a variety of researchers (reviewed by U.S.EPA, 2003; CMHC, 1997; Johnson and Ettinger, 1991). In general, a net negative pressure difference on the order of 1 to 12 Pa has been observed, with this pressure difference being influenced by factors such as house height, presence/absence of chimney, presence/absence of appliance fans, below grade versus slab on grade construction (CMHC, 1997). CMHC (1997) indicates that pressure differentials between the indoor and outdoor environment during the winter heating season for 1 or 2 storey dwellings span from 2 Pa (no chimney, mild winter) to 12 Pa (severe winter, chimney, no fresh air intake for combustion air supply, frequently used exhaust fan and/or fireplace). The expected modal or average condition during winter would be a 7 Pa negative pressure differential. Assuming that the heating season lasts 6 months, and that a zero pressure difference exists for the remainder of the year, then the annual average or typical pressure differential would be 4 Pa (rounded to one significant digit from a value of 3.5 Pa).

For commercial and industrial buildings, a lower default negative pressure differential of 2 Pa was selected. Commercial and industrial buildings are expected to maintain a lower overall pressure differential, compared to residential buildings, because of forced, calibrated air exchange designed into heating systems, and due to the more regular and routine movement of building occupants into and out of the structure.

4.0 Soil Permeability to Vapour Flow

The permeability of soil beneath a building foundation to vapours is one of the most sensitive parameters in the Johnson and Ettinger (1991) model. It is affected by the size and shape of soil pore openings as well as the water content of the soil.

US EPA (2003) suggests that typical soil vapour permeabilities are within the following ranges:

Soil Type	Vapour Permeability (cm^2)
Medium sand	1.0×10^{-7} to 1.0×10^{-6}

Fine sand	$1.0x10^{-8}$ to $1.0x10^{-7}$
Silty sand	$1.0x10^{-9}$ to $1.0x10^{-8}$
Clayey silt	$1.0x10^{-10}$ to $1.0x10^{-9}$

The Johnson and Ettinger (1991) model indicates that advective flow is the dominant process by which contaminants enter a building when the soil vapour permeability is high; as the soil vapour permeability becomes lower, diffusion begins to affect transport into the building. However, advection can still have a noticeable effect even at a soil vapour permeability of $1.0x10^{-10}$ cm^2.

5.0 Building Air Exchange Rate

Information on air exchange rate (or air changes per hour; ACH) is required to estimate the degree of dilution of infiltrating PHC vapours in fresh (uncontaminated) indoor air. A large variety of studies have been published documenting measurements of ACH in homes. Most of those studies suggest an average ACH of between 0.3 and 0.5 for homes in Canada or homes from northern regions of the United States. However, these ACH measurements are routinely collected with conditions that simulate Canadian winter conditions: all windows and doors tightly closed. Also, these measurements are often taken in unoccupied homes. As a result, average ACH values from reported data generally do not reflect typical 'lived-in' house conditions, nor do they reflect annual average conditions. Pandian et al. (1993) reported data collected on air change rate for more than 4000 U.S. homes. Their data include measurements collected during all four seasons. Average summer measurements were between 2.8 times greater, 13.5 times greater, and 10.8 times greater than measurements collected in spring, fall and winter, respectively. The fact that ACH increases significantly with open doors and/or windows is corroborated by Otson et al. (1998) and Lamb et al. (1985).

CMHC (1997) indicates that more recently built residences have a lower ACH than older homes. CMHC suggests that ACH values for homes built pre-1960 may range from 2 to 10 times greater than recently constructed 'airtight' homes. This is generally supported by data from Pandian et al. (1993), Grimsrud et al. (1983), Gerry et al. (1986) and King et al. (1986) and likely reflects building practices which increase energy efficiency in more recent construction. Based on data presented by Grimsrud et al. (1983) the geometric mean ACH for homes built prior to 1970 was 0.69, whereas homes built during or after 1970 had a geometric mean ACH of 0.46. This difference was statistically significant.

ACH values for multi-level homes tend to be greater than ACH values for single storey residences. Pandian et al. (1993) report ACH values of 0.6 and 2.8 for one-level and two-level homes, respectively. Data from Grimsrud et al. (1983) indicate geometric mean ACH values of 0.47 and 0.52 for one-level and two-level homes, respectively. Again, these latter values are statistically significantly different.

Data comparing natural air exchange rates in commercial properties are limited compared to residential homes. Greater door traffic is anticipated to result in greater natural air exchange in commercial versus residential buildings. Data reported by Kailing (1984) on natural air exchange rates indicate ACH values ranging from 0.09 to 1.54 for commercial structures compared to 0.01 to 0.85 for residences. Many commercial properties (especially malls and other large facilities) will have mechanical ventilation systems to maintain adequate ventilation to ensure indoor air quality (see ASHRAE Standard 62-1989, for example). Sherman et al. (1994) and Weschler et al. (1996) report ACH values of 1.5 to 1.8 ACH for small commercial buildings under mechanical ventilation.

6.0 Diffusional path length for volatile chemicals

For purposes of generic guideline development, it has been assumed that the soil-borne contamination is a minimum of 30 cm (0.3 m) from the building foundation. The contaminated vapours must migrate through this 30 cm of clean fill before reaching and penetrating the building foundation.

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APPENDIX F
CHECKING PROTOCOL FOR CONTAMINANT INGESTION RESULTING FROM PRODUCE, MEAT, AND MILK PRODUCED AT REMEDIATED RESIDENTIAL AND AGRICULTURAL SITES

1.0 Background and Application

Humans can be indirectly exposed to contaminants in soils through food-chain contamination of produce, meat, and milk. For agricultural land use, it is likely that some meat, produce (i.e., vegetables), and milk will be produced and consumed on-site. In the residential setting, it is also possible that a backyard garden may provide a significant portion of the produce consumed by a family. To ensure that soil remediation guidelines do not result in an unacceptable contribution to total daily intake of contaminants via home-grown produce, meat, and milk, it is necessary to compare the expected intake of contaminants from these sources with the total intake.

The concentration estimated to occur in food from soils contaminated at the preliminary soil guideline concentration must be less than the maximum residue limit (MRL) published under the Food and Drug Act. In addition, the total daily intake estimated by the procedure outlined in this appendix must not exceed total background exposure from food (i.e., estimated daily intake) by more than 20% of the difference between the TDI and the EDI, for non-carcinogens. For carcinogens, total contaminant intake must not exceed the risk specific dose (RSD) for a cancer risk of 10^{-6}.

The procedure and assumptions for estimating the concentration of a contaminant in food and daily intake is described in Section 1.1. The protocol provides an estimate of bioconcentration of contaminants into, and consumption of, these foods, in relation to the assumptions outlined in Section 1.1.

No detailed data about the proportion of food Canadians consumed from local origin are available. However, it is believed to be a substantial contribution to total daily intake, especially in an agricultural setting. For example, a 1985 survey in Quebec indicated that 42% of urban residents, and 58% of rural residents consume vegetables from their own gardens (MAPAQ, 1985). The proportion may vary considerably from one location to another. Generally however, it should reflect the variations relating to the type of environment considered (i.e., urban, rural or suburban) and it is also unlikely that these foods will be consumed over the entire year.

SQGTG considers that exposure from consumption of local garden produce, meat and milk has to be calculated as part of the generic guidelines for every contaminant for this land use. Generic guidelines for an agricultural setting will be protective of exposure from local produce consumption.

Because of the potential variation in lifestyle (i.e., geographical considerations, frequency of consumption), SQGTG decided that generic guidelines for a residential scenario would not take into account exposure from local produce consumption. However, guideline values which calculate the contribution of local produce to total intake will still be available in each assessment document. Therefore, SQGTG does not recommend using generic guidelines for a residential setting with a garden. When there is a garden, the calculated value presented in each assessment document would have to be considered as remediation objectives.

1.1 Assumptions

Based on the results of a study (USDA, 1983 in Versar, 1989), the homegrown fraction of total vegetables (leafy, head, and root/tuber) consumed in rural, suburban and urban areas have been calculated. The values presented in Table F.1 were estimated from the available data distributions based on climatic and lifestyle considerations (MSSSQ, 2002).

As shown in Table F.1, 50% of all produce (i.e., vegetables) consumed on an agricultural site is grown on site, and 50% of the meat and 100% of the milk consumed is from local origin. For residential land, the contribution of home-grown produce (i.e., from local gardens) to daily contaminant intake is considerably less (10%) and is not considered at all for milk or meat. Based on a health risk assessment prepared by the Ontario Ministry of Environment and Energy (1994), the percentage of total fruits and vegetables consumed originating from a backyard garden is 7%. The value recommended by SQGTG is 10% (Table F.1).

2.0 Bioconcentration of Soil Contaminants into Produce

The concentration of a chemical in produce resulting from contaminated soil may be estimated as:

$$C_p \text{ (mg/kg)} = B_v \times C_S \text{ (mg/kg)} \qquad [1]$$

Where C_p is the concentration in produce, and B_v is the chemical-specific (and possibly plant-specific) bioconcentration factor.

Bioconcentration factors should be based on measured values where possible. For organic chemicals, a model was developed by Travis and Arms (1988) using the octanol/water partition coefficient (log K_{ow}) to estimate the bioconcentration factor. This model was recently updated by US EPA (2003) reflecting new data and a critical evaluation of the existing data:

$$\log B_v = 2.53 - 0.4965 \log K_{ow} \qquad [2]$$

However, given the high level of uncertainty in the application of a bioconcentration factor estimated based on chemical properties, this approach should be used as a last resort, and only if there is evidence that the chemical does bioaccumulate. Estimated bioconcentration factors should not normally be used for volatile organic chemicals, which are often metabolized in plants, unless there is clear evidence of measured bioaccumulation.

Table F.1 Proposed Values for Local Garden Produce, Meat, and Dairy Product Consumption

	Proportion of Produce from Local Origin (% food intake)			
	Residential land use	Agricultural land use	Commercial land use	Industrial land use
Leafy Vegetables	10	50	NA	NA
Vegetables with Fruits (tomatoes...)	10	50	NA	NA
Root and Tuber	10	50	NA	NA
Meat	NA	50	NA	NA
Milk	NA	100	NA	NA

Note: Values are estimated by professional judgement from data distributions reported by VERSAR (1989), on the basis of climatic and lifestyle considerations. These data are used by Ministère de la Santé et des Services sociaux du Québec (2002).

2.1 Human Daily Intake of Contaminants from Produce

Intake resulting from contaminated produce (i.e. vegetables only) can be defined as:

$$I_p = \frac{(P_h \times P_c \times B_v \times C_s) + (P_l \times P_c \times P_r)}{BW} \qquad [3]$$

where:

I_p = total intake of contaminants from produce (mg/kg-day)
P_h = percent produce homegrown
P_c = produce consumption rate (kg/day)
B_v = bioconcentration factor for produce
C_S = concentration of contaminant in soil (mg/kg)
P_l = percent produce purchased

P_r = average chemical concentration in retail produce (mg/kg)
BW = body weight (kg)

Note that concentrations, consumption rates and bioconcentration factors are based on wet weights; if concentrations or bioconcentration factors are available on a dry weight basis they must be corrected to reflect wet weight.

For non-carcinogens, the receptor is assumed to be a toddler; for carcinogens, the receptor is assumed to be an adult. Receptor characteristics, including body weight and produce ingestion rates, are summarized in Appendix I. For residential land use, 10% of garden produce consumed is assumed to be locally grown. For agricultural land use, a 50% value is recommended (Table F.1).

3.0 Bioaccumulation of Soil Contaminants in Meat and Milk and Estimated Daily Intakes

For the purposes of this general procedure, only direct ingestion of soils by beef and dairy cattle will be considered. It is assumed that beef is the major type grazing animal consumed by humans. Grazing animals directly ingest anywhere from 0.4 to 0.9 kg of soil per day (McKone and Ryan, 1989; Fries and Paustenbach, 1990). Studies demonstrate that the uptake of lipophilic substances such as PCBs deposited in or on grazed crops is much less than that taken up through the direct ingestion of soil by cattle (Fries and Jacobs, 1986). Therefore indirect contamination via ingestion of vegetation is not considered.

Most dairy and beef cattle are fed from harvested forage in enclosed feedlots or barns (Paustenbach, 1989). Therefore, the opportunity to ingest soils will be somewhat restricted. However, increasing consumer demand for "organically grown" meat and milk from free-range animals increases the likelihood that animals will spend more of their lifetime grazing. The procedure recommended by SQGTG assumes that an animal is free range for the majority of the year.

3.1 Bioaccumulation of Contaminants in Meat

Where possible, the concentration of the chemical in meat should be evaluated using measured bioaccumulation or biotransfer factors.

Travis and Arms (1988) have studied the bioaccumulation potential of organic contaminants into beef and developed the following model:

$$\log B_p = -7.6 + \log K_{ow},\ n = 36,\ r = 0.81 \qquad [4]$$

Where the biotransfer factor for beef (B_p) is defined as:

$$B_p = \frac{\text{concentration in beef (fresh weight: mg/kg)}}{\text{daily intake of chemical (mg/day)}}$$

Assuming that beef cattle ingest, on average, 0.9 kg of soil per day (Fries and Paustenbach, 1990), and that chemical intake with vegetation is negligible compared to that with direct soil intake, then the daily intake of chemical can be defined as:

daily chemical intake by beef cattle = C_s x 0.9 kg/day [5]
Where C_s is the concentration (mg/kg) of the chemical in the soil.

Substituting equation 5 into the definition for B_p and the latter into equation 4, and rearranging the equation, then the potential concentration of the organic chemical in beef (C_p) can be defined as:

C_p = antilog(-7.6 + log K_{ow})(day/kg) x C_S (mg/kg) x 0.9(kg/day)(mg/kg) [6]

3.2 Human Daily Intake of Contaminants from Meat

Intake resulting from contaminated meat can be defined as:

$$I_h = \frac{\left(M_h \times M_c \times B_p \times C_s \times SIR_c\right) + \left(B_c \times M_c \times M_r\right)}{BW} \qquad [7]$$

Where:

I_b	=	total intake of contaminants from beef (mg/kg-day)
M_h	=	percentage of meat home produced
M_c	=	meat consumption rate (kg/day)
B_p	=	biotransfer factor for beef (d/kg)
C_s	=	chemical concentration in soil (mg/kg)
SIR_c	=	soil ingestion rate for cattle (0.9 kg/d)
B_c	=	percentage beef purchased
M_r	=	average chemical concentration in retail beef (mg/kg)

Note that concentrations, consumption rates and biotransfer factors are based on wet weights; if concentrations or biotransfer factors are available on a dry weight basis they must be corrected to reflect wet weight.

For non-carcinogens, the receptor is a toddler; for carcinogens, the receptor is an adult. Receptor characteristics, including body weight and produce ingestion rates, are summarized in Appendix I. For agricultural land uses, 50% of all meat is assumed to be produced on-site and that most meat consumed is beef. The average chemical concentration in retail beef is obtained from the multimedia exposure assessment.

3.3 Bioaccumulation of Contaminants in Milk

Concentrations of chemicals in milk should be estimated from measured bioaccumulation or biotransfer factors where possible. In the absence of measured values, Travis and Arms (1988) have also studied the bioaccumulation potential of organic contaminants into milk and developed the following model:

$$\log B_m = -8.1 + \log K_{ow},\ n= 28,\ r = 0.74 \qquad [8]$$

Where the biotransfer factor for milk B_m is defined as:

$$B_m = \frac{\text{concentration in whole milk (mg/kg)}}{\text{daily intake of chemical (mg/day)}}$$

Assuming that dairy cattle ingest, on average, 0.9 kg of soil per day (Fries and Paustenbach. 1990), and that chemical intake via vegetation is negligible compared to that with direct soil intake, then the daily intake of chemical can be defined as:

$$\text{daily chemical intake by dairy cattle} = C_s \times 0.9 \text{ kg/day} \qquad [9]$$

Where C_s is the concentration (mg/kg) of the chemical in the soil.

Substituting equation 9 into the definition of B_m and the latter into equation 8, and rearranging the equation, then the potential concentration of the organic chemical in milk (C_m) can be defined as:

$$C_m = \text{antilog}(-8.1 + \log K_{ow})(\text{day/kg}) \times C_S \text{ (mg/kg)} \times 0.9(\text{kg/day}) \qquad [10]$$

3.4 Human Daily Intake of Contaminants from Milk

Intake resulting from contaminated milk can be defined as:

$$I_m = \frac{(MK_h \times MK_c \times B_m \times C_s \times SIR_c) + (MK_s \times MK_c \times MK_r)}{BW} \qquad [11]$$

Where:

I_m	=	total intake of contaminants from milk (mg/kg day)
MK_h	=	percentage milk home produced
MK_c	=	milk consumption rate (kg/day)
B_m	=	biotransfer factor for milk (d/kg)
C_s	=	chemical concentration in soil (mg/kg)
SIR_c	=	soil ingestion rate for cattle (0.9 kg/d)
MK_s	=	percentage milk purchased
MK_r	=	average chemical concentration in retail milk (mg/kg)

For non-carcinogens, the receptor is assumed to be a toddler; for carcinogens, the receptor is assumed to be an adult. Receptor characteristics, including body weight and produce ingestion rates, are summarized in Appendix I. For agricultural land use, 100% of all milk is assumed to be produced on site (Table F.1).

4.0 *Soil Guidelines Checking Procedure*

The soil quality guideline for produce, meat and milk ingestion is calculated using the following equations:

Threshold Chemicals

$$SQG_{FI} = \frac{(TDI - EDI) \times BW \times SAF}{(P_h \times P_c \times B_v) + (M_h \times M_c \times B_p \times SIR_c) + (MK_h \times MK_c \times B_m \times SIR_c)} + BSC \qquad [12]$$

Non-Threshold Chemicals

$$SQG_{FI} = \frac{RSD \times BW}{(P_h \times P_c \times B_v) + (M_h \times M_c \times B_p \times SIR_c) + (MK_h \times MK_c \times B_m \times SIR_c)} + BSC \qquad [13]$$

where:

SQG_{FI} = soil quality guideline for food (produce, meat, milk) ingestion (mg/kg)
TDI = tolerable daily intake (mg/kg/d)
EDI = estimated daily intake (mg/kg/d)
RSD = risk-specific dose (mg/kg/d)
BW = receptor body weight (kg)
SAF = soil allocation factor (unitless)
P_h = proportion of produce homegrown (0.5 for agricultural; 0.1 for residential)
P_c = produce consumption rate (kg/d)
B_v = bioconcentration factor for produce
M_h = proportion of meat home produced (0.5 for agricultural; 0 for residential)
M_c = meat consumption rate (kg/d)
B_p = meat biotransfer factor (d/kg)
SIR_c = soil ingestion rate for cattle (= 0.9 kg/d)
MK_h = proportion of milk home produced (1.0 for agricultural; 0 for residential)
MK_c = milk consumption rate (kg/d)
B_m = milk biotransfer factor (d/kg)
BSC = background soil concentration (mg/kg)

Note: the above equations assume the biotransfer factors for meat and milk are in units of d/kg (mg/kg in meat or milk per mg/day intake) such as those determined using equations 4 and 8. If a bioaccumulation factor in units of mg/kg in meat or milk per mg/kg in soil is used, then the SIR_c terms should be omitted from the equations.

Additionally, the calculated contaminant concentration in produce, meat and milk must be less than the required Maximum Residue Limit (MRL) specified in the Food and Drug Regulations, when available. This can be checked by solving equations 1, 6 and 10 with C_S set equal to the preliminary SQG_{FI}. If the calculated concentration in any food type exceeds the MRL, then the SQG_{FI} must be lowered such that the MRL is not exceeded.

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APPENDIX G
EVALUATION OF INDUSTRIAL AND COMMERCIAL LAND USE HUMAN HEALTH GUIDELINES RELATIVE TO ADJACENT LAND USE

1.0 General

Soil erosion and subsequent deposition can transfer contaminated soil from one property to another. Where adjacent properties are uncontaminated or used for a more sensitive land use, this transfer of contaminants may result in unacceptable degradation.

Soil contamination can also be transported to more sensitive properties via groundwater and subsurface vapours; however, evaluation of these mechanisms requires that a minimum offset distance be established; therefore these mechanisms are not evaluated on a generic basis at this time. Groundwater or subsurface vapour migration may be considered on a jurisdictional or site-specific basis where appropriate.

2.0 Erosion

Soil susceptibility to wind and water erosion is related to soil and climate characteristics and soil management. Although similar factors help determine sensitivity to erosion, wind and water erosion processes are sufficiently different to require separate modelling efforts.

2.1 Water erosion

Water erosion can be modeled by the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978), shown below:

$$A = R \times K \times L \times S \times C \times P$$

where

A is the rate of soil loss.
R is the rainfall and runoff factor.
K is the soil erodibility factor.
L is the slope-length factor.
S is the slope steepness factor.
C is the cover and management factor.
P is the support practice factor.

2.2 Wind erosion

Wind erosion is modeled by the Wind Erosion Equation (WEQ) (Woodruff and Siddoway, 1965). The WEQ is described by the functional relationship:

$$E = ü(l, C, K, L, V)$$

where

E is the rate of soil loss.
I is the soil erodibility index.
C is the climate factor.
K is the surface roughness.
L is the maximum unsheltered distance across a site along the direction of the prevailing wind.
V is the vegetative cover factor.

2.3 Modelling

Both the above models are empirical, based on a number of regression equations between soil loss and the various soil, climate, and management factors that affect it. Derivation of the input parameters is difficult and in the case of the WEQ, calculation is also difficult because of complex mathematical relationships between the input variables. However, computer models based upon these two equations are available to calculate erosion losses from basic soil, climate, and management data. One of the most commonly used is the Erosion/Productivity Impact Calculator (EPIC) (Williams et al., 1990). Although EPIC is primarily used to evaluate the effect of agricultural practices on soil productivity, its ability to estimate soil erosion rates (t/ha/y) from basic soil and climate data make it a valuable tool for evaluating erosion under other land use scenarios.

3.0 Deposition

To estimate the impact of eroded soil on off-site areas, soil deposition on the area of concern must be calculated. Although the above models are available for estimating soil loss by erosion, very little work has been done on subsequent deposition. Process based models of erosion and subsequent deposition have been developed, but their data input requirements are generally extensive, and as such they are not considered appropriate for generic guideline development at this time.

Deposition of the eroded material will depend on the landscape. In water erosion, most industrial sites are designed to contain runoff within site boundaries and off-site erosion will be minimal. Soil eroded from those sites without runoff controls will move with runoff until reaching the toe of the slope where it will be deposited. The area covered by the deposited material will depend on local topography. Wind eroded material will move until encountering a barrier acting as a windbreak or, in the case of fine particles, until removed from suspension by rain.

One can assume that the eroded soil leaving a contaminated site will be deposited over an equivalent area off-site. This soil will be deposited as a surface layer and will therefore be immediately available for contact. The depth of this layer can be calculated from the quantity of soil deposited over a given area and an assumed bulk density. Some mixing of the deposited material with native soil due to traffic, runoff, or gardening will likely occur, diluting the contaminated soil. A reasonable surface microrelief is 2 cm and the mixing of deposited soil with uncontaminated native soil takes place within this zone. Soil erosion and deposition are on-going processes. However gradual removal of the contaminated soil and ongoing mixing with uncontaminated soil through erosion occurring in other parts of the landscape should result in contaminant concentrations reaching an equilibrium over time. In the absence of any procedure for accounting for these mitigating processes, a period of five years was chosen as an appropriate timeframe for contaminants to build up in the receiving soil.

4.0 Calculations

4.1 General

The concentration of a contaminant in erosional soil that will raise the receiving soil concentration above a given level can be calculated by assuming a background concentration in the receiving soil and estimating the quantity of soil originating from a hypothetical commercial or industrial site. With appropriate input parameters for soil, climate, and site characteristics, EPIC can estimate the quantity of soil eroded from the site. Agricultural land use is the receiving soil.

4.2 Erosion/Productivity Impact Calculator input parameters

A soil with 3% organic carbon and a sandy loam texture (73% sand, 19% silt and 8% clay) was chosen as representative soil susceptible to erosion. A 1 hectare site with 1% slope and 650 kg/ha of vegetative surface cover was modeled.

Two climate scenarios were run. Climate data from Lethbridge, Alberta were used to estimate potential erosion where wind is the dominant erosive force. Climatic data from Halifax, Nova Scotia, were used to simulate erosion where rainfall was the dominant force. EPIC estimated the following losses over a five-year period:

Soil Lost by Erosion (t/ha)

Site	Wind	Water	Total
Lethbridge	13.2	3.3	16.5
Halifax	0.0	11.3	11.3

The two values were averaged to produce an estimated loss by erosion of 13.9 t/ha.

Assuming a bulk density for the eroded material of 1 t/m^3 and a depositional area equal to the source area, the depth of the deposited material (D^d) can be evaluated to 0.14 cm using:

$$D_d = E/(\rho_b \times 10^2) \qquad [1]$$

where

E   is the mass of deposited material = 13.9 t/ha
ρ_b is the bulk density = 1 t/m^3

Assuming a bulk density of 1 t/m^3 for receiving soil and a mixing depth of 2 cm, the final concentration of the receiving soil after mixing can be calculated by:

$$Cm = \{(2 - D_d)\ BSC) + (D_d \times C_i)\}/2 \qquad [2]$$

where

Cm   is the concentration of contaminant in the receiving soil after mixing (μg/g).
BSC is the background concentration of the contaminant in the receiving soil (μg/g).
C_i is the concentration of contaminant in the eroded soil (μg/g).

Substituting the value calculated for D_d in equation [2] and replacing C_m with SQG_A (the soil quality guideline for the agricultural land use), that equation can be rearranged to calculate the concentration in soil eroded from the commercial or industrial site that will raise the contaminant concentration in the receiving soil (assumed to be a background concentration initially) to the agricultural use guideline, which is used to calculate the soil quality guidelines for offsite migration:

$$SQG_{OM} = 14.3 \times SQG_A - 13.3 \times BSC \qquad [3]$$

The environmental soil quality guideline for offsite migration ($SQG_{OM\text{-}E}$) is calculated using the SQG_E for the agricultural land use; the human health soil quality guideline for offsite migration ($SQG_{OM\text{-}HH}$) is calculated using the SQG_{HH} for the agricultural land use.

A review of nine metal contaminants showed that SQG_{OM} calculated by equation (3) averaged 12 times the value of the soil quality guidelines for the more sensitive land use.

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APPENDIX H
SUMMARY OF MODELS AND EQUATIONS USED IN SOIL QUALITY GUIDELINE DEVELOPMENT

Soil Quality Guideline for Human Health (Direct Contact Pathways)

Threshold chemicals:

$$SQG_{DH} = \frac{(TDI - EDI) \times SAF \times BW}{[(AF_G \times SIR) + (AF_L \times IR_S \times ET_2) + (AF_S \times SR)] \times ET_1} + BSC$$

Soil ingestion only:

$$SQG_{DH-SI} = \frac{(TDI - EDI) \times SAF \times BW}{(AF_G \times SIR) \times ET_1} + BSC$$

Dermal contact only:

$$SQG_{DH-DC} = \frac{(TDI - EDI) \times SAF \times BW}{(AF_S \times SR) \times ET_1} + BSC$$

Particulate inhalation only:

$$SQG_{DH-PI} = \frac{(TDI - EDI) \times SAF \times BW}{(AF_L \times IR_S) \times ET_1 \times ET_2} + BSC$$

Non-threshold chemicals:

$$SQG_{DH} = \frac{RSD \times BW}{[(AF_G \times SIR) + (AF_L \times IR_S) + (AF_S \times SR)] \times ET} + BSC$$

Soil ingestion only:

$$SQG_{DH-SI} = \frac{RSD \times BW}{(AF_G \times SIR) \times ET} + BSC$$

Dermal contact only:

$$SQG_{DH-DC} = \frac{RSD \times BW}{(AF_S \times SR) \times ET} + BSC$$

Particulate inhalation only:

$$SQG_{DH-PI} = \frac{RSD \times BW}{\left(AF_L \times IR_S\right) \times ET} + BSC$$

where,

SQG_{DH}	=	direct human health-based soil quality guideline (mg/kg)
TDI	=	tolerable daily intake (mg/kg bw·day)
EDI	=	estimated daily intake (multimedia exposure assessment) (mg/kg·day)
RSD	=	risk specific dose (mg/kg·day)
SAF	=	soil allocation factor (unitless)
BW	=	body weight (kg)
BSC	=	background soil concentration (mg/kg)
AF_G	=	relative absorption factor for gut (unitless)
AF_L	=	relative absorption factor for lung (unitless)
AF_S	=	relative absorption factor for skin (unitless)
SIR	=	soil ingestion rate (kg/day)
IR_S	=	soil inhalation rate (kg/day)
SR	=	soil dermal contact rate (kg/day) – ***see below***
ET	=	exposure term (unitless) = 1 for non-threshold chemicals
ET1	=	exposure term 1 (unitless) – days per week/7 x weeks per year/52
ET2	=	exposure term 2 (unitless) – hours per day/24

Soil Dermal Contact Rate:

$$SR = \left(SA_H\, DL_H + SA_O\, DL_O\right)EF$$

where,

SA_H	=	exposed surface area of hands (m^2)
SA_O	=	area of exposed body surfaces other than hands (m^2)
DL_H	=	dermal loading of soil to hands (kg/m^2-event)
DL_O	=	dermal loading of soil to other surfaces (kg/m^2-event)
EF	=	exposure frequency (events/d)

Soil Quality Guideline for the Protection of Indoor Air Quality

Threshold chemicals:

$$SQG_{IAQ} = [(TC - C_a)\{\theta_w + (K_{OC})(f_{OC})(\rho_b) + (H')(\theta_a)\}(SAF)(DFi)(10^3\, g/kg)]/[(H')(\rho_b)(ET)(10^6\, cm^3/m^3)] + BSC$$

Non-threshold chemicals:

$$SQG_{IAQ} = [(RSC)\{\theta_w + (K_{OC})(f_{OC})(\rho_b) + (H')(\theta_a)\}(DFi)(10^3 g/kg)]/[(H')(\rho_b)(ET)(10^6 cm^3/m^3)] + BSC$$

Where: SQG_{IAQ} = soil quality guideline for the protection of indoor air quality
TC = tolerable concentration or reference concentration (mg/m^3)
RSC = risk-specific concentration
C_a = background indoor/outdoor air concentration (mg/m^3)
SAF = soil allocation factor (unitless)
BW = body weight (kg)
θ_a = vapour-filled porosity (unitless) = effective porosity (n) – moisture-filled porosity
θ_w = moisture-filled porosity (unitless)
n = soil porosity (unitless)
K_{OC} = organic carbon partition coefficient (mL/g)
f_{OC} = soil organic carbon fraction in contaminant partitioning zone (g/g)
ρ_b = soil dry bulk density in contaminant partitioning zone (g/cm^3)
H' = unitless Henry's Law Constant = H/RT
H = Henry's Law Constant ($atm\text{-}m^3/mol$)
R = gas constant (8.2×10^{-5} $atm\text{-}m^3/mol\text{-}K$)
T = annual average soil temperature (K)
DF_i = dilution factor from soil gas to indoor air (unitless): ***see derivation below***
ET = exposure term (unitless)
BSC = background soil concentration (mg/kg)

Calculation of DF for indoor infiltration pathway:

$$DF_i = \frac{1}{\alpha}$$

DF_i = dilution factor from soil gas concentration to indoor air concentration (unitless)
α = attenuation coefficient
= (contaminant vapour concentration in the building)/(vapour concentration at the contaminant source)

$$D_T^{eff} \approx D_a\left(\frac{\theta_a^{10/3}}{n^2}\right)$$

D_T^{eff} = overall effective porous media diffusion coefficient based on vapour-phase concentrations for the region between the source and foundation (cm^2/s)
D_a = pure component molecular diffusivities in air (cm^2/s)
θ_a = vapour-filled porosity (unitless)
n = total soil porosity (unitless)

$$Q_B = L_B W_B H_B (ACH) \big/ \left(3600\ s/h\right)$$

Q_B = building ventilation rate (cm^3/s)
L_B = building length (cm)
W_B = building width (cm)
H_B = building height, including basement (cm)
ACH = air exchanges per hour (h^{-1})

$$Q_{soil} = \frac{2\pi\ \Delta P\ k_v\ X_{crack}}{\mu \ln\left[\frac{2\left(Z_{crack}\right)}{r_{crack}}\right]}$$

Q_{soil} = volumetric flow rate of soil gas into the building (cm^3/s)
ΔP = pressure differential ($g/cm{\cdot}s^2$)
k_v = soil permeability to vapour flow (cm^2)
X_{crack} = length of idealized cylinder (cm)
μ = vapour viscosity ($g/cm{\cdot}s$)
Z_{crack} = distance below grade to idealized cylinder (cm)
r_{crack} = radius of idealized cylinder (cm)

$$\alpha = \frac{\left(\frac{D_T^{eff} A_B}{Q_B L_T}\right)\exp\left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}}\right)}{\exp\left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}}\right) + \left(\frac{D_T^{eff} A_B}{Q_B L_T}\right) + \left(\frac{D_T^{eff} A_B}{Q_{soil} L_T}\right)\left[\exp\left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}}\right) - 1\right]}$$

D_T^{eff} = effective porous media diffusion coefficient (cm^2/s)
A_B = building area – floor and subgrade walls (cm^2)
Q_B = building ventilation rate (cm^3/s)
L_T = distance from contaminant source to foundation (cm)
Q_{soil} = volumetric flow rate of soil gas into the building (cm^3/s)
L_{crack} = thickness of the foundation (cm)
D_{crack} = effective vapour-pressure diffusion coefficient through the crack (cm^2/s); assumed to be equal to D_T^{eff}
A_{crack} = area of cracks through which contaminant vapours enter the building (cm^2)

For the effective diffusion coefficient through the crack (D_{crack}), it is assumed that a coarse, granular material is used as the base for the floor and footings. Therefore, it is assumed that the cracks are filled with coarse soil, even if the native soil is fine/medium textured. Consequently, D_{crack} will be the same as D_T^{eff} for coarse soils, regardless of the surrounding soil texture.

Protection of Groundwater Pathways

The groundwater model includes four components:

Soil/Leachate Partitioning (DF1)

$$SQG_{GW} = C_L \left\{ K_d + \left(\frac{\theta_w + H'\theta_a}{\rho_b} \right) \right\}$$

SQG_{GW}	=	soil quality guideline for the protection of groundwater (mg/kg) (i.e. SQG_{PW}, SQG_{FL}, SQG_{IR}, SQG_{LW})
C_L	=	allowable leachate concentration at source (mg/L) – calculated below
K_d	=	distribution coefficient (cm^3/g) – see Appendix A
θ_w	=	water filled porosity (unitless)
H'	=	dimensionless Henry's Law constant = H x 42.32
H	=	Henry's Law constant (atm-m^3/mol)
θ_a	=	air-filled porosity (unitless)
ρ_b	=	soil bulk density in contaminant partitioning zone (g/cm^3)

Unsaturated Groundwater Zone (DF2)
Note – for generic guideline development, contamination is assumed to be in contact with groundwater, and DF2 = 1 ($C_L = C_z$)

$$C_L = C_z \Big/ \exp\left[\frac{b}{2\partial_u} - \frac{b}{2\partial_u}\left(1 + \frac{4\partial_u L_{US}}{v_u}\right)^{1/2} \right]$$

$$v_u = \frac{I}{\theta_w R_u}; \quad R_u = 1 + \frac{\rho_b}{\theta_w} K_d$$

C_L	=	allowable chemical concentration in leachate at the source (mg/L)
C_z	=	allowable chemical concentration in leachate at the water table (mg/L) calculated below
b	=	thickness of unsaturated zone below the source (m) = d – Z
d	=	depth from surface to groundwater surface (m)
Z	=	depth to bottom of contaminated soil (m)
∂_u	=	dispersivity in the unsaturated zone (m) = 0.1b
L_{US}	=	decay constant for chemical (y^{-1}) in unsaturated zone:

$$L_{US} = \frac{0.693}{t_{1/2US}} \left(e^{-0.07d}\right) \left(1 - \frac{D_{1/2US}}{365}\right)$$

$t_{1/2US}$	=	chemical half-life in unsaturated zone (years)
$D_{1/2US}$	=	days with temperature <0°C
v_u	=	average linear leachate velocity (m/y)
I	=	infiltration rate (m/y) = precipitation minus runoff and evapotranspiration
θ_w	=	water-filled porosity (unitless)
R_u	=	retardation factor in unsaturated zone (unitless)
ρ_b	=	soil bulk density in unsaturated zone (g/cm^3)
K_d	=	distribution coefficient (cm^3/g) – see Appendix A

Mixing Zone Unsaturated/Saturated (DF3)

$$C_z = C_{gw}\left\{1 + \left(\frac{Z_d K_H i}{IX}\right)\right\}$$

C_z	=	allowable chemical concentration in leachate at the water table (mg/L)
C_{gw}	=	allowable chemical concentration in groundwater at the source (mg/L) – calculated below
Z_d	=	average thickness of mixing zone (m) – calculated below
K_H	=	hydraulic conductivity in the saturated zone (m/y)
i	=	hydraulic gradient (unitless)
I	=	infiltration rate (m/y) = precipitation minus runoff and evapotranspiration
X	=	length of source parallel to groundwater flow (m)

Calculation of average thickness of mixing zone:

$Z_d = r + s$; Z_d cannot exceed d_a

r	=	mixing depth available due to dispersion and diffusion (m)
	=	0.01 X
X	=	length of source parallel to groundwater flow (m)
s	=	mixing depth available due to infiltration rate and groundwater flow rate (m)

$$s = d_a\left\{1 - e^{-\frac{2.178 XI}{K_H i d_a}}\right\}$$

d_a	=	depth of unconfined aquifer (m)
I	=	infiltration rate (m/y) = precipitation minus runoff and evapotranspiration
K_H	=	hydraulic conductivity in the saturated zone (m/y)
i	=	hydraulic gradient (unitless)

Saturated Groundwater Zone (DF4)

Note: for a receptor located at the edge of the contaminant source, DF4 = 1 ($C_{gw} = C_w$)

Note that this equation is a function of time since release; the value of 't' must be determined such that the worst-case result is found. This may be done iteratively by solving the equation with various values of t and using the worst-case result.

$$C_w(x,y,z,t) = \left(\frac{C_{gw}}{4}\right)\exp\left\{\left(\frac{x}{2\partial_x}\right)\left[1-\left(1+\frac{4L_s\partial_x}{v}\right)^{1/2}\right]\right\} erfc\left[\frac{x - vt\left(1+\frac{4L_s\partial_x}{v}\right)^{1/2}}{2(\partial_x vt)^{1/2}}\right]$$

$$\left\{erf\left[\frac{(y+Y/2)}{2(\partial_y x)^{1/2}}\right] - erf\left[\frac{y-Y/2}{2(\partial_y x)^{1/2}}\right]\right\}$$

$$v = \frac{K_H i}{n_e R_f}; \quad R_f = 1 + \frac{\rho_b}{n} K_d$$

C_w	=	allowable chemical concentration in water at receptor (mg/L) (i.e. drinking water guideline, source guidance value for groundwater, FL guideline, irrigation water guideline, livestock watering guideline as appropriate)
x	=	distance from source to receptor (m)
x,y,z	=	Cartesian coordinates relating source and receptor (m); y, z assumed to be 0
t	=	time since contaminant release (years)
C_{gw}	=	allowable chemical concentration in groundwater at source (mg/L)
∂_x	=	longitudinal dispersivity tensor = 0.1x
∂_y	=	lateral dispersivity tensor = $0.1\partial_x$
L_s	=	decay constant (y^{-1}) in saturated zone:

$$L_s = \frac{0.693}{t_{1/2S}}\left(e^{-0.07d}\right)$$

d	=	depth from surface to groundwater surface (m)
$t_{1/2S}$	=	biodegradation half-life in saturated zone (y)
v	=	velocity of contaminant (m/y)
K_H	=	hydraulic conductivity in the saturated zone (m/y)
i	=	hydraulic gradient (unitless)
n	=	total porosity of soil = 1 - ρ_b/2.65 (unitless)
n_e	=	effective soil porosity (unitless); generally assumed to be the same as total soil porosity (n)
Y	=	source width (m) perpendicular to groundwater flow
R_f	=	retardation factor (unitless)
ρ_b	=	soil bulk density in saturated zone (g/cm^3)
K_d	=	distribution coefficient (cm^3/g) – see Appendix A

Ingestion of Contaminated Produce, Meat and Milk

Threshold Chemicals

$$SQG_{FI} = \frac{(TDI - EDI) \times BW \times SAF}{(P_h \times P_c \times B_v) + (M_h \times M_c \times B_p \times SIR_c) + (MK_h \times MK_c \times B_m \times SIR_c)} + BSC$$

[12]

Non-Threshold Chemicals

$$SQG_{FI} = \frac{RSD \times BW}{(P_h \times P_c \times B_v) + (M_h \times M_c \times B_p \times SIR_c) + (MK_h \times MK_c \times B_m \times SIR_c)} + BSC$$

[13]

where:

SQG_{FI} = soil quality guideline for food (produce, meat, milk) ingestion (mg/kg)
TDI = tolerable daily intake (mg/kg/d)
EDI = estimated daily intake (mg/kg/d)
RSD = risk-specific dose (mg/kg/d)
BW = receptor body weight (kg)
SAF = soil allocation factor (unitless)
P_h = proportion of produce home-grown (0.5 for agricultural; 0.1 for residential)
P_c = produce consumption rate (kg/d)
B_v = biotransfer factor for produce
M_h = proportion of meat home produced (0.5 for agricultural; 0 for residential)
M_c = meat consumption rate (kg/d)
B_p = meat biotransfer factor (d/kg)
SIR_c = soil ingestion rate for cattle (= 0.9 kg/d)
MK_h = proportion of milk home produced (1.0 for agricultural; 0 for residential)
MK_c = dairy product consumption rate (kg/d)
B_m = dairy product biotransfer factor (d/kg)
BSC = background soil concentration (mg/kg)

Note: the above equations assume the biotransfer factors for meat and milk are in units of d/kg (mg/kg in food per mg/day intake). If a biotransfer factor in units of mg/kg in food per mg/kg in soil, then the SIR_c terms should be omitted from the equations.

Off-Site Migration Pathway

$SQG_{OM} = 14.3 \times SQG_A - 13.3 \times BSC$

where:
SQG_{OM} = off-site migration check (mg/kg)
SQG_A = soil quality guideline (SQG_E or SQG_{HH}) for the agricultural land use (mg/kg)
BSC = background concentration of the contaminant in the receiving soil (mg/kg)

Soil and Food Ingestion Pathway

Daily Threshold Effects Dose

DTED = lowest ED / UF

where:
DTED = daily threshold effects dose of the consumer (mg/kg bw-day)
ED = lowest effects dose (mg/kg bw-day)
UF = uncertainty factor (if needed)

Soil Ingestion Rate

SIR = (DMIR x PSI)

where:
SIR = the soil ingestion rate of the consumer (kg dw soil/day)
DMIR = geometric mean of the available dry matter intake rates for the consumer (kg dw/day)
PSI = geometric mean of available soil ingestion proportions reported with the DMIR

Food Ingestion Rate

FIR = DMIR - SIR

where:
FIR = the food ingestion rate for the species selected as the consumer (kg dw food/day)
DMIR = geometric mean of the available dry matter intake rates for the consumer (kg dw/day)
SIR = the soil ingestion rate of the consumer (kg dw soil/day)

Ingestion Guideline for Primary Consumers

$$SQG_{1C} = \frac{0.75 \times DTED_{1C} \times BW_{1C}}{\left(SIR_{1C} \times BF\right) + \left(FIR_{1C} \times BCF_1\right)}$$

where:

SQG_{1C} = soil quality guideline for soil and food ingestion for the primary consumer (mg/kg)
$DTED_{1C}$ = daily threshold effects dose of the primary consumer (mg/kg bw_{1C}-day)
BW_{1C} = body weight (kg)
SIR_{1C} = soil ingestion rate (kg dw soil/day)
BF = bioavailability factor (unitless)
FIR_{1C} = the food ingestion rate for the species selected as the primary consumer (kg dw food/day)
BCF_1 = bioconcentration factor (unitless)

Ingestion Guideline for Secondary Consumers

$$SQG_{2C} = \frac{0.75 \times DTED_{2C} \times BW_{2C}}{[(SIR_{2C} \times BF) + (FIR_{2C} \times BAF_2)] \times AF_{FR} \times AF_Y}$$

where,
SQG_{2C} = soil quality guideline for soil and food ingestion for the secondary consumer (mg/kg dw soil)
$DTED_{2C}$ = daily threshold effects dose for the secondary consumer (mg/kg bw-day)
BW_{2C} = body weight of the species used in the $DTED_{2C}$ (kg)
SIR_{2C} = the soil ingestion rate for the species used in the $DTED_{2C}$ (kg dw soil/day)
BF = bioavailability factor (unitless)
FIR_{2C} = the food ingestion rate for the species used in the $DTED_{2C}$ (kg dw food/day)
BAF_2 = bioaccumulation factor (unitless)

Ingestion Guideline for Tertiary Consumers

$$SQG_{3C} = \frac{0.75 \times DTED_{3C} \times BW_{3C}}{[(SIR_{3C} \times BF) + (FIR_{3C} \times BAF_3)] \times AF_{FR} \times AF_Y}$$

where:
SQG_{3C} = soil quality guideline for soil and food ingestion for the tertiary consumer (mg/kg dw soil)
$DTED_{3C}$ = daily threshold effects dose for the tertiary consumer (mg/kg bw-day)
BW_{3C} = body weight of the species used in the $DTED_{3C}$ (kg)
SIR_{3C} = the soil ingestion rate for the species used in the $DTED_{3C}$ (kg dw soil/day)
BF = bioavailability factor (unitless)
FIR_{3C} = the food ingestion rate for the species used in the $DTED_{3C}$ (kg dw food/day)
BAF_3 = bioaccumulation factor (unitless)

APPENDIX I
DEFAULT PARAMETERS FOR GUIDELINE DEVELOPMENT

This appendix presents the default values used to calculate soil quality guidelines, along with relevant background information. These values may be updated from time to time; it should be noted that existing soil quality guidelines are not normally revised when these default values are changed.

Table I.1 Human Receptor Characteristics[a]

Parameter	Symbol	Infant (0 – 6 mo)	Toddler (7 mo - 4 y)	Child (5 – 11 y)	Teen (12 – 19 y)	Adult (20+ y)
Body Weight (kg)	BW	8.2	16.5	32.9	59.7	70.7
Air Inhalation Rate (m^3/d)	IR	2.1	9.3	14.5	15.8	15.8
Soil Inhalation Rate (kg/d)[d]	IR_S	$1.60x10^{-9}$	$7.07x10^{-9}$	$1.10x10^{-8}$	$1.20x10^{-8}$	$1.20x10^{-8}$
Water Ingestion Rate (L/d)	WIR	0.3	0.6	0.8	1.0	1.5
Soil Ingestion Rate (kg/d)	SIR	0.00002	0.00008	0.00002	0.00002	0.00002
Skin Surface Area (m^2)						
- Hands	SA_H	0.032	0.043	0.059	0.080	0.089
- Other[b]	SA_O	0.146	0.258	0.455	0.223	0.250
Dermal Loading to Skin (kg/m^2-event)						
- Hands[c]	DL_H	0.001	0.001	0.001	0.001	0.001
- Other[c]	DL_O	0.0001	0.0001	0.0001	0.0001	0.0001
Dermal Exposure Frequency (events/d)[b]	EF	1	1	1	1	1
Produce Ingestion Rate (g/d)	P_c	155	172	259	347	325
Meat Ingestion Rate (g/d)[c]	M_c	52	86	123	170	166
Milk Ingestion Rate (g/d)[c]	MK_c	664	592	613	583	286

a – all values from Health Canada, 2003 unless otherwise specified
b – arms assumed to be exposed for adults and teens; arms and legs assumed to be exposed for infants, toddlers and children
c – Richardson, 1997
d – IRs were calculated as daily inhalation rate (m3/day) X $7.6x10^{-10}$ kg/m3; the latter value is the assumed airborne concentration of suspended soil particulate above a contaminated site (HC, 2004).

Body Weight

Average body weights were recommended by Health Canada (2003), based on surveys conducted in 1981 for adults (CFLRI, 1981) and 1970-1972 for children (EHD, 1992). Weight increases were observed in the Canadian population over the period from 1970 through 1988 (Richardson, 1997).

Inhalation Rate

The inhalation rates were generated using data on time-activity information for the Canadian population combined with ventilation rates reported for different activity levels (Richardson, 1997).

Water Ingestion Rate

Water ingestion rates are based on a study of Canadian tap water consumption conducted during 1977-1978 (NHW, 1981), involving questionnaires and individual water consumption diaries.

Soil Ingestion Rate

Soil ingestion rates are based on a study conducted by Angus Environmental (1991), as well as Health Canada recommendations.

Skin Surface Area

The recommended skin surface areas are based on equations developed by US EPA for estimation of skin surface area from weight and height, along with Canadian weight and height data.

Soil to Skin Adherence

Soil loading to skin was studied in both field and controlled trials by Kissel et al. (1996, 1998). Loading was found to be related to the type of activity and soil moisture, and was found to be higher on hands than arms and legs. Loadings were not found to be markedly different between children and adults. The recommended soil to skin adherence values were adapted from this study, and reflect typical exposure conditions.

Produce, Meat and Milk Ingestion Rates

Food ingestion rates are based on a Nutrition Canada Survey (NHW, 1977) conducted between 1970 and 1972. While some Canadian dietary habits are believed to have changed since then, Health Canada (2003) assumed that the magnitude of these changes would likely be relatively small in relation to other uncertainties related to estimating human intake of contaminants.

The produce ingestion rate used herein is based on consumption of vegetables and fresh fruit; the meat ingestion rate is based on the ingestion of beef, pork, veal and lamb, and the milk ingestion rate is based on the ingestion of whole, 2% and skim milk (not milk products).

Table I.2 Soil and Hydrogeological Parameters

Parameter	**Symbol**	**Soil Type**	
		Coarse-grained	**Fine-grained**
Saturated Hydraulic Conductivity (m/y)	K_H	320	32[a]
Hydraulic Gradient	i	0.028	0.028
Recharge (Infiltration rate) (m/y)	I	0.28	0.20
Organic Carbon Fraction (g/g)	foc	0.005	0.005
Soil Bulk Density (g/cm^3)	ρ_b	1.7	1.4
Water Content	M_W/M_S	0.07	0.12
Total Soil Porosity	n	0.36	0.47
Vapour-Filled Porosity	θ_a	0.241	0.302
Moisture-Filled Porosity	θ_w	0.119	0.168
Soil Vapour Permeability (cm^2)	k_v	6×10^{-8}	10^{-10}

a – 32 m/y is applied for the protection of potable water, since lower hydraulic conductivities would likely not result in sufficient aquifer yield to support consumptive use. It is recommended that this same value also be applied for the protection of freshwater life.

Saturated Hydraulic Conductivity

The default values for hydraulic conductivity were chosen to reflect typical aquifers encountered in Canada. The coarse-grained soil value (320 m/y) is representative of silty sand (Freeze & Cherry, 1979), and was selected because low values are more conservative (i.e. result in lower guidelines) for pathways where there is no offset distance between the contamination and the groundwater receptor. This value may not be conservative for pathways involving saturated zone transport; however, higher saturated hydraulic conductivities would likely be at least partially offset by associated lower hydraulic gradients. A hydraulic conductivity of 3.2 m/y is representative of silt (Freeze & Cherry, 1979); however, for fine-grained soil a value of 32 m/y is used. This higher value is recommended for the protection of potable water pathway, since lower hydraulic conductivities would likely not result in sufficient aquifer yield for use as a potable water source. The higher value of 32 m/y is also recommended for the protection of aquatic life because it will result in guidelines that are more conservative and consistent with the desired level of protection.

Hydraulic Gradient

The hydraulic gradient is a dimensionless quantity describing the steepness of the water potential gradient. In unconfined aquifers, it is roughly equivalent to the gradient of the water table. A hydraulic gradient of 0.028 is recommended as a default value (CCME 1996); hydraulic gradient is inversely correlated with the saturated hydraulic conductivity. It should be noted that where the hydraulic gradient of a site is known to differ significantly from 0.028, calculation of Tier 2 guideline values should use the site-specific hydraulic gradient in the calculations for all groundwater pathways.

Recharge

Over long periods of time, groundwater recharge can be estimated by subtracting evapotranspiration and surface runoff from precipitation rates. The default recharge values are based on data for Halifax, Nova Scotia, adapted from the Atlantic PIRI project, and are expected to be representative of high-rainfall areas in Canada, which are most sensitive to soil-to-groundwater cross contamination.

Organic Carbon Fraction

The default organic carbon fractions for coarse and fine-grained soils are based on a review of the organic carbon contents of various Canadian subsoils undertaken in support of the Canada-Wide Standard for Petroleum Hydrocarbons in Soil (PHC CWS) (CCME, 2000).

Soil Bulk Density and Moisture Content

The default soil bulk densities and moisture contents were chosen to be representative of typical sand (coarse-grained) and clay (fine-grained) soils.

Porosity

The total soil porosity is calculated from the soil bulk density, assuming a particle density of 2.65 g/cm^3. The moisture-filled porosity is calculated as the soil bulk density multiplied by the moisture content (assuming a water density of 1 g/cm^3). The vapour-filled porosity is obtained by subtracting the moisture-filled porosity from the total porosity.

Soil Vapour Permeability

Soil vapour permeability is discussed further in Appendix E. The default values were selected to be consistent with the hydraulic conductivities defined for the soil types, since these parameters are closely related.

Table I.3 Site Characteristics

Parameter	SYMBOL	VALUE
Contaminant Source Width (m)	Y	10
Contaminant Source Depth (m)	Z	3
Contaminant Source Length (m)	X	10
Distance to Surface Water (m)	x	10
Distance to Potable Water User (m)	x	0
Distance to Agricultural Water User (m)	x	0
Distance from Contamination to Building Slab (cm)	L_T	30
Depth to Groundwater (water table) (m)	d	3
Thickness of Unsaturated Soils Beneath Contamination (m)	b	0
Days with surface temp. < 0^{o}C (days)	$D_{1/2US}$	365
Depth of unconfined aquifer (m)	d_a	5

Source Width, Depth and Length

Dimensions of the contaminated area are assumed based on typical contaminated sites in Canada. Length is defined in the direction parallel to groundwater flow, while width is defined in the direction perpendicular to groundwater flow.

Distance to Surface Water

It is assumed for purposes of generic guideline development that a surface water body could be located 10 m from the remediated soils. An offset distance is considered possible for this pathway, since the locations of surface water bodies are normally relatively unchanging.

Distance to Potable and Agricultural Water Users

Potable water and agricultural water users are assumed to be located at the downgradient edge of the remediated soils. Inclusion of an offset distance for these pathways on a generic basis may lead to inappropriate or unmanageable water use restrictions; however, offset distances may be incorporated on a site-specific basis where appropriate.

Distance from Contamination to Building Slab

Consistent with the development of the PHC CWS, it is assumed that 30 cm of clean fill are present beneath any building.

Depth to Groundwater and Thickness of Unsaturated Soils Beneath Contamination

For generic guideline development purposes, it is assumed that contaminated soils are in direct contact with groundwater.

Days with Surface Temperature < 0^oC

At the default value for this parameter (365 days), no biodegradation occurs in the unsaturated zone. Jurisdictions may specify regional default values for days with surface temperature <0^oC. This parameter only has an effect if the soil contamination is not in direct contact with the groundwater.

Depth of Unconfined Aquifer

The depth of the unconfined aquifer is assumed to be 5 m, as per the British Columbia Contaminated Sites Soils Task Group (CSST). The calculated mixing depth should not be allowed to exceed this value.

Table I.4 Building Parameters

Parameter	Symbol	Residential Basement	Residential Slab-On-Grade	Commercial Slab-On-Grade
Building Length (cm)	L_B	1225	1225	2000
Building Width (cm)	W_B	1225	1225	1500
Building Area (cm^2)	A_B	$2.7x10^6$	$1.5x10^6$	$3.0x10^6$
Building Height (cm)[a]	H_B	488	488	300
Thickness of Building Foundation (cm)	L_{crack}	11.25	11.25	11.25
Depth Below Grade of Foundation (cm)	Z_{crack}	244	11.25	11.25
Crack Radius (cm)	r_{crack}	0.2	0.2	0.26
Area of Crack (cm^2)	A_{crack}	994.5	994.5	1846
Length of Idealized Cylinder (cm)	X_{crack}	4900	4900	7000
Air Exchanges per Hour (1/h)	ACH	1	1	2
Pressure Differential ($g/cm\text{-}s^2$)	ΔP	40	40	20

a – including basement

Building parameters have been adapted from CCME (2000), and were originally based on a review of typical building characteristics and building codes. Air exchange rates and building pressure differentials are discussed in more detail in Appendix E.

The slab-on-grade residential building is typically more sensitive than the residential building with a basement with default site parameters, due to higher advective flow. Nonetheless, it is recommended that soil quality guidelines for the protection of indoor air quality be calculated for both scenarios for agricultural and residential/parkland land uses to ensure that the most sensitive exposure route is considered. For commercial and industrial land uses, only slab-on-grade construction will be considered Parameters for buildings both with and without basements are provided in Table I.4.

Required Chemical Properties

- Tolerable Daily Intake and/or Risk-Specific Dose
- Tolerable Concentration and/or Risk-Specific Concentration (volatile chemicals)
- Estimated Daily Intake (threshold chemicals)
- Background Soil Concentration
- Background Air Concentration (volatile threshold chemicals)
- Organic Carbon Partitioning Coefficient (K_{oc})
- Henry's Law Constant (volatile chemicals)
- Diffusion Coefficient in Air (volatile chemicals)
- Half-Life in the Saturated Zone (soluble chemicals)
- Water Solubility (soluble chemicals)

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