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Re. **BMI Plant Sites and Common Areas Projects, Henderson, Nevada**
NDEP Guidance on Ecological Risk Assessment

Dear Sirs and Madam:

Attachment A contains guidance developed for the NDEP regarding Screening Level Ecological Risk Assessment Guidelines. It is the expectation of the NDEP that this guidance will be used by the companies listed above to address ecological risk assessment issues. An electronic copy of this document will also be forwarded to you for your use.

If you have any questions, do not hesitate to contact me.

Sincerely,

Brian A. Rakvica, P.E.
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Screening Level Ecological Risk Assessment Guidelines for the BMI Complex, Henderson, Nevada

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Table of Contents

Table of Contents i
List of Figures iv
List of Tables v
List of Acronyms vi
Executive Summary viii

1.0 Introduction 1
2.0 Problem Formulation 7
2.1 COPC Identification 7
2.1.1 Frequency of Detection Analysis 7
2.1.2 Background Comparison 7
2.1.2.1 Distribution Shift Tests 7
2.2 Data Evaluation and Characterization 9
2.3 Ecological Scoping 10
2.4 Conceptual Site Model 12
2.4.1 Functional Food Webs 12
2.4.2 Contaminant Exposure Modalities 17
2.4.3 Exposure Modalities Across Major Taxa 18
2.4.4 CSM Summary 19
3.0 Screening-Level Assessment Endpoints and Receptor Selection 20
3.1 Terrestrial Receptors and Assessment Methods 21
3.2 Aquatic Receptors and Assessment Methods 23
4.0 Screening Analysis 25
4.1 Introduction 25
4.2 Analysis Basics 26
4.2.1 Toxicity Reference Value Development 26
4.2.1.1 Ecological Relevance of TRVs 27
4.2.1.2 Study Design 28
4.2.1.3 Approaches to TRV Development 31
4.2.2 Ecological Screening Levels 32
4.2.3 Hazard Quotient and Index Calculations 33
4.3 Terrestrial Wildlife Exposure Models 35
4.3.1 EPA Soil Screening Levels (Eco-SSLs) 36
4.3.2 Bioaccumulation Models for Terrestrial Plants and Invertebrates 39
4.3.2.1 Bioaccumulation by Plants in Soil 39
4.3.2.2 Bioaccumulation by Invertebrates in Soil 41
4.3.3 Generalized Vertebrate Wildlife Exposure Model 43
4.3.3.1 Generalized Ingestion Models for Vertebrate Receptors 47
4.3.3.2 Generalized Imbibition Models for Vertebrate Receptors 52
4.3.3.3 VOC Inhalation Models for Terrestrial Vertebrate Receptors 55

4.3.4 Biomagnification and Trophic Transfer Models for Aquatically-Based Contaminants 57
4.3.4.1 Sediments to Vertebrate Wildlife 57
4.3.4.2 Water to Vertebrate Wildlife 60
4.4 Aquatic Benchmark Value Comparisons 63
4.4.1 Water Quality Benchmarks 63
4.4.1.1 Summary of Water ESL Derivations 70
4.4.2 Sediment Quality Benchmarks 70
4.4.2.1 Summary of Sediment ESL Derivations 78
4.5 Radiological Contaminants 78
5.0 Uncertainty Analysis 79
5.1 Chemical Concentrations and Distributions 79
5.2 Background Comparisons 80
5.3 Frequency of Detection 81
5.4 TRV Selection and Development 81
5.5 Screening Analyses 82
5.6 HQ and HI – Single and Multiple Contaminants 84
5.7 Conceptual Exposure Models 85
5.8 Uncertainty's Epilogue 86
6.0 Risk Interpretation 86
7.0 References Cited 88

Appendix A Ecological Scoping Checklist

A-1.0 Part A - Scoping Meeting Documentation A-1
A-2.0 Part B - Site Visit Documentation A-2
A-3.0 Part C - Ecological Pathways Conceptual Exposure Model A-4
A-4.0 Part D - Signatures and certifications A-15

Appendix B General Assessment Endpoints for the BMI Complex and Affected Areas

B-1.0 Introduction to GAE Process and Application B-1
B-2.0 Overview of the GAE Process B-1
B-3.0 GAEs Based on Ecological Relevance B-2
B-3.1 Values Common to All Ecosystems B-3
B-3.1.1 Biological Diversity (Biodiversity) B-3
B-3.2.1 Functional Integrity B-5
B-3.1.3 Energy and Nutrient Dynamics B-6
B-3.2 Values Common to the Northern Mojave Desert Ecosystem B-6
B-3.2.1 Functional Components of the Northern Mojave Desert Ecosystem B-7
B-3.2.1.1 Food Webs Applicable to the BMI Complex B-7
B-3.2.2 Attributes of the Functional Components B-9
B-3.3 GAEs Based on Ecological Relevance B-13
B-3.3.1 Globally Relevant Endpoints B-13
B-3.3.2 Regionally Relevant Endpoints B-13

B-4.0 Values and GAEs For the BMI Complex and surrounds
Based On Societal Relevance B-14
B-4.1 Criteria for Management Goals B-14
B-4.2 GAEs Based on Societal Relevance B-15
B-5.0 Application of GAEs in the Ecological Risk Assessment Process B-15
B-6.0 References Cited (Appendix B) B-18

List of figures

Figure	Page
Figure 1. The entirety of the BMI Complex, showing geographical relationship to the Las Vegas Wash and immediate surrounds. (Figure prepared by MWH Americas, Inc., and used with permission of BMI.)	3
Figure 2. The BMI Complex with the "no-build" area highlighted and proximity to the Las Vegas Wash illustrated. (Figure prepared by MWH Americas, Inc., and used with permission of BMI.)	4
Figure 3. A stepwise approach to the SLERA process at the BMI site.	6
Figure 4. An Ecological Pathways Conceptual Exposure Model for terrestrial receptors.	12
Figure 5. An Ecological Pathways Conceptual Exposure Model for aquatic receptors.	12
Figure 6. A terrestrial food web organized by trophic categories and functional feeding guilds.	14
Figure 7. An aquatic food web organized by trophic categories and functional feeding guilds.	15
Figure 8. Ecological pathways conceptual exposure model for terrestrial receptors exposed to soil-based contaminants in the context of evaluation by Eco-SSLs.	38
Figure 9. Ecological conceptual exposure pathway for terrestrial wildlife exposed to contaminants in soils.	48
Figure 10. Ecological conceptual exposure pathway for terrestrial wildlife exposed to contaminants in drinking water.	53
Figure 11. Ecological pathways conceptual exposure model for terrestrial insectivores exposed to sediment-born contaminants.	59
Figure 12. Ecological pathways conceptual exposure model for terrestrial piscivores from water-born contaminants.	61
Figure 13. Ecological conceptual exposure model for aquatic receptors exposed to contaminants in water.	64
Figure 14. Ecological conceptual exposure model for aquatic receptors exposed to contaminants in aquatic sediments.	72
Appendix A	
Figure A-1. Ecological Pathways Conceptual Exposure Model for terrestrial receptors.	A-13
Figure A-2. Ecological pathways conceptual exposure model for aquatic receptors.	A-14
Appendix B	
Figure B-1. A terrestrial food web organized by trophic categories and functional feeding guilds.	B-8
Figure B-2. An aquatic food web organized by trophic categories and functional feeding guilds.	B-9

List of Tables

Table	Page
Table 1. Measures required for the burrow air VOC exposure model for the pocket gopher (Thomomys bottae)	56
Table 2. Method for obtaining a final non-radionuclide WQB.	65
Table 3. Secondary acute value factors (SAVFs) for estimation of Tier II secondary chronic values (SCVs).	69
Table 4. Summary of sources for water ESLs.	70
Table 5. Method for obtaining the final sediment ESL for non-radionuclides.	73
Appendix B	
Table B-1. Attributes of ecological components occupying the BMI Complex and surrounds.	B-10
Table B-2. Significant ecological attributes of functional subgroups.	B-11

Acronyms

ACR	Acute-Chronic Ratio
AE	Assessment Endpoint
AET	Apparent Effects Threshold
ARAR	Applicable or Relevant and Appropriate Requirements
AUF	Area Use Factor
BAF	Bioaccumulation Factor
BCG	Biota Concentration Guides
BMF	Biomagnification Factor
BMI	Basic Management Incorporated
CCC	Criterion Continuous Concentration
COPC	Constituent of Potential Concern
COPEC	Constituent of Potential Ecological Concern
CR	Concentration Ratio
CSM	Conceptual Site Model
CV	Chronic Value
DQO	Data Quality Objective
Eco-SSL	Ecological Soil Screening Level
EPA	Environmental Protection Agency
EPCEM	Ecological Pathways Conceptual Exposure model
EqP	Equilibrium Partitioning Method
ERA	Ecological Risk Assessment
ERL	Effects Range Low
ERM	Effects Range Median
ESL	Ecological Screening Level
FACR	Final Acute-Chronic Ratio
FAV	Final Acute Value
FCV	Final Chronic Value
FCV	Final Chronic Value
f_{oc}	fraction of organic carbon
FOD	Frequency of Detection
FPV	Final Plant Value
GAE	General Assessment Endpoint
GEAE	Generic Ecological Assessment Endpoint
GIS	Geographic Information System
GMAV	Genus Mean Acute Value
GMCV	Genus Mean Chronic Value
HI	Hazard Index
HQ	Hazard Quotient
INEEL	Idaho National Environmental Engineering laboratory
K_{ow}	Octanol-Water partition coefficient
LANL	Los Alamos National Laboratory

LC	Lethal Concentration
LD	Lethal Dose
LOAEL	Lowest Observed Adverse Effects Level
LOEC	Lowest Observed Effects Concentration
NAWQC	National Ambient Water Quality Criteria
NDEP	Nevada Department of Environmental Protection
NOAEL	No Observed Adverse Effects Level
NOEC	No Observed Effects Concentration
OW	(EPA) Office of Water
PEL	Probable Effects Level
SACR	Secondary Acute-Chronic Ratio
SAV	Secondary Acute Value
SCV	Secondary Chronic Value
SEC	Sediment Effects Concentration
SLERA	Screening Level Ecological Risk Assessment
SMACR	Species Mean Acute-Chronic Ratio
SMAV	Species Mean Acute Value
SMCV	Species Mean Chronic Value
SMDP	Scientific Management Decision Point
SOF	Sum of Fractions
SQB	Sediment Quality Benchmark
SQC	Sediment Quality Criteria
SQL	Sample Quantitation Limit
TEL	Threshold Effects Level
TRV	Toxicity Reference Value
TTF	Trophic Transfer Factor
UF	Uncertainty Factor
VOC	Volatile Organic Compound
WQB	Water Quality Benchmark
WQC	Water Quality Criteria

Executive Summary

This guidance describes the stepwise process for preparing a screening level ecological risk assessment (SLERA) for the "no-build" area of the Basic Management Inc. (BMI) Common Areas or "Complex," Henderson, Nevada. This SLERA may also be used to address other, relevant portions of the BMI Complex or areas affected by the BMI Complex. The no-build area is a terrestrial environment of approximately 200 acres located along the northern boundary of the BMI Complex, between the City of Henderson Birding preserve and the Las Vegas Wash. This SLERA process will also be applied to the area known as the Kerr-McGee Seep, which lies immediately north of the northern boundary of the BMI Complex and no-build area. Although the seep is not strictly within the no-build area, it is geographically contiguous, and contaminants from the no-build area have likely been transported to this spring. Water exiting the Kerr-McGee Seep enters the Las Vegas Wash, immediately to the north, with which all features of the no-build area and Kerr-McGee Seep are contiguous. The Las Vegas Wash is considered a feature of major ecological and recreational importance to the cities of Las Vegas and Henderson (SWCA 2005).

The purpose of this document is to provide the management goals and methodological guidance for the performance of a SLERA in accord with the U.S. Environmental Protection Agency's (EPA's) *Ecological Risk Assessment Guidance for Superfund* (EPA 1997), *Guidelines for Ecological Risk Assessment* (EPA 1998a), and *Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites* (EPA 1999). This document draws heavily from EPA's *Guidance for Developing Soil Screening Levels* (EPA 2003a and Attachments, EPA 2005), *Update of Ecological Soil Screening Level (Eco-SSL) Guidance and Contaminant Specific Documents* (EPA 2005), and EPA's *National Water Quality Criteria* (EPA 2002a), as well as other EPA support and guidance documents (EPA 1993, 1995a and b, 1996a and b, 2002b). Methods for this SLERA approach are derived largely from Los Alamos National Laboratory's (LANL's) *Screening-Level Ecological Risk Assessment Methods, Revision 2* (LANL 2004), and Idaho National Engineering and Environmental Laboratory's (INEEL's) *Methodology for Conducting Screening-Level Ecological Risk Assessments for Hazardous Waste Sites* (VanHorn et al. 1998, Hampton et al. 1998, Kester et al 1998).

This document is to serve as a guide and a step-wise methodological approach to completion of a screening-level ecological risk assessment for the BMI Complex and areas affected by the BMI Complex. Completion of a screening level ecological risk assessment that meets or exceeds the methodological detail proscribed in this document will result in the focus of further investigations of ecological risk on those factors that are understood to drive the risk potential for all areas evaluated. Multiple approaches are presented to most aspects of the SLERA in order to provide flexibility that may be required due to the specific nature of any single feature of the assessment; aspects that may arise from the form and nature of data to a thorough understanding of ecological components and processes relevant to the BMI Complex and surrounding areas. With the multiplicity of methods provided herein, a complete screening assessment, assessment of attending uncertainty, and screening-level risk interpretation can be readily achieved.

1.0 Introduction

This guidance describes the stepwise process for preparing a screening level ecological risk assessment (SLERA) for the "no-build" area of the Basic Management Inc. (BMI) Common Areas or "Complex," Henderson, Nevada. This SLERA may also be used to address other, relevant portions of the BMI Complex or areas affected by the BMI Complex. The BMI Complex is located north of the Henderson city center, immediately west of the Boulder Highway between Russell Road and Lake Mead Drive (Figure 1) additional lands owned by BMI are located east of Boulder Highway. The no-build area is a terrestrial environment of approximately 200 acres located along the northern boundary of the BMI Complex, between the City of Henderson Birding preserve and the Las Vegas Wash (Figure 2). This SLERA process will also be applied to the area known as the Kerr-McGee Seep, which lies immediately north of the northern boundary of the BMI Complex and no-build area. Although the seep is not strictly within the no-build area, it is geographically contiguous, and contaminants from the no-build area have likely been transported to this spring. Water exiting the Kerr-McGee Seep enters the Las Vegas Wash, immediately to the north, with which all features of the no-build area and Kerr-McGee Seep are contiguous. The Las Vegas Wash is considered a feature of major ecological and recreational importance to the cities of Las Vegas and Henderson (SWCA 2005).

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The objective of this guidance is to provide a stepwise process to guide ecological risk assessors and focus the site assessment on drivers of potential ecological risk. In this context, the primary drivers of potential ecological risk are the contaminants attributable to historical industrial operations that potentially played a role in and/or may be currently impacting biotic populations and communities of organisms that occupy the BMI Complex and areas affected by the BMI Complex. Concern for biotic receptors at and around the BMI Complex are reflected in the management goals for the site. The following management goals are relevant to determining if contaminants are impacting biota:

1. Protect wildlife populations that are either currently present or may inhabit the site in the future, based on habitat mitigation or restoration efforts. These include plants, invertebrates, and vertebrate wildlife relevant to the BMI Complex and affected areas.

2. Protect special status species that are currently present or may inhabit the site in the future, based on habitat mitigation or restoration efforts.
3. Determine if on-site contaminants are adversely affecting plants, invertebrates, or vertebrate wildlife.
4. Minimize bioaccumulation and biomagnification of contaminants of concern.

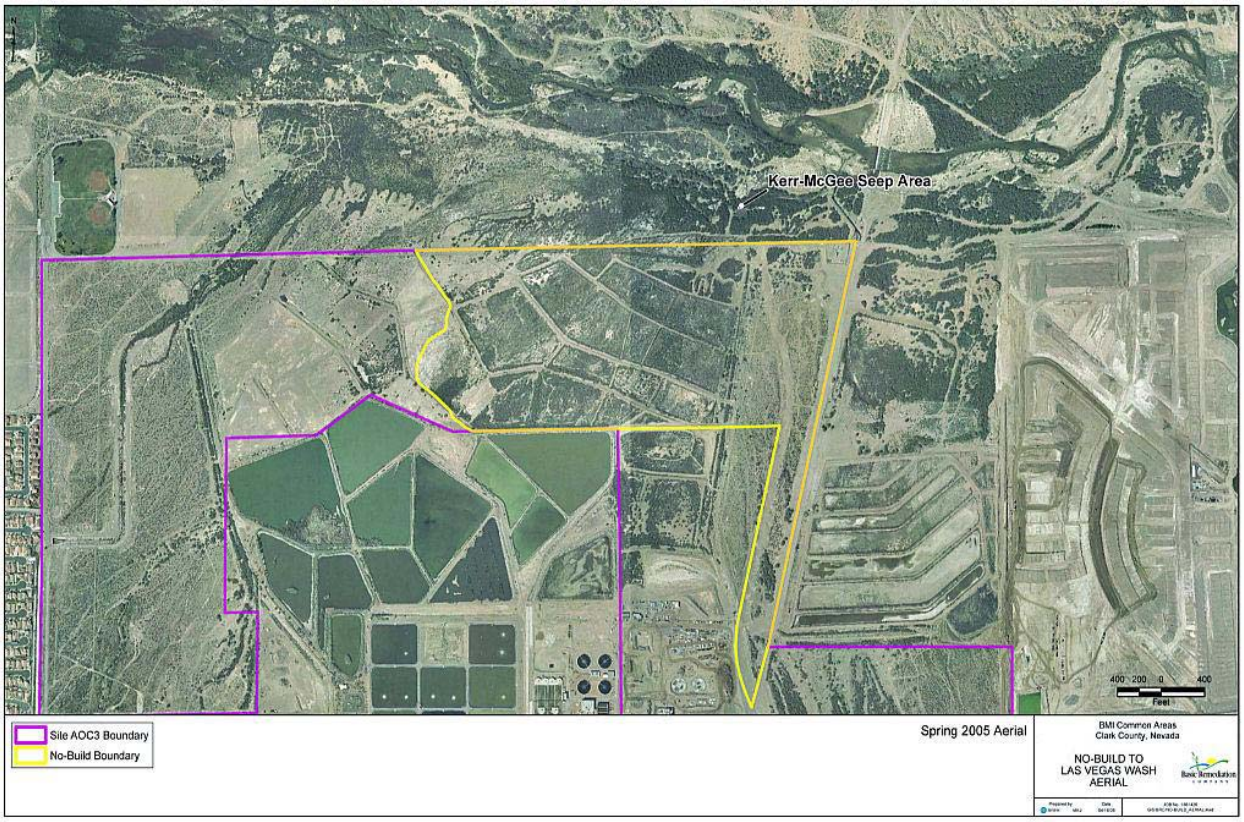
SLERA methods provide high confidence that any potentially adverse impact(s) to ecological receptors resulting from exposure to contaminants are not overlooked or eliminated from consideration in investigations of ecological risk. Thus, the SLERA is intentionally protective, and will identify constituents that clearly pose no threat to biota, and eliminate those constituents from further assessment in the ecological risk assessment process. For constituents that are not eliminated from further risk evaluation, site-specific evaluations are needed to accurately clarify and characterize risks, and to provide adequate information for risk management decisions. The SLERA leads risk assessors to more focused, detailed and representative investigations for contaminants of potential concern for ecological risk, assessment endpoints, and associated measures of exposure, effect, and receptor/ecosystem characteristics. In cases where SLERA methods are applied to multiple sites, it also provides a common metric for risk comparison among contaminants of potential ecological concern and provides a tool for prioritizing site investigations, investigations between sites or contaminated areas within a given site, and corrective actions.

The SLERA requires completion of a conceptual site model (CSM) for the distribution, transport, and fate of contaminants for the areas of interest relating to the BMI Complex, and how contaminants interface with site biota. The CSM must contain descriptive text for the geographic, climatic, geologic, edaphic, hydrologic, and biological conditions of the site. Media that have been identified as potentially contaminated on the site include surface soil, subsurface alluvium, surface sediments, spring water and groundwater. The CSM must convey a thorough understanding of distribution, transport, and fate of contaminants as they pertain to the various contaminated media, how these are affected by site physical and biotic conditions, and the ultimate pathway of contaminant conveyance to site biota.

Contaminants include inorganic and organic chemicals and radionuclides. The CSM must evaluate existing analytical data and supporting site survey and measurements in the context of present-day contaminant concentrations, locations, distributions, and transport and fate processes. Data Quality Objectives (DQOs) must have been developed to describe the type and amounts of data needed to characterize the nature and extent of contamination. DQOs should be developed and documented following EPA guidance (EPA 2006a, EPA 2001) or equivalent methods based on the scientific method. Analytical data must have been validated and assessed to determine if DQOs were met and therefore considered adequate to characterize the nature and extent of site contamination. Data Quality Assessment should follow EPA methods or the equivalent (EPA 2006b and 2006c).

Figure 1. The entirety of the BMI Complex, showing geographical relationship to the Las Vegas Wash and immediate surrounds. (Figure prepared by MWH Americas, Inc., and used with permission of BMI.)

Figure 2. The BMI Complex with the "no-build" area highlighted and proximity to the Las Vegas Wash illustrated. (Figure prepared by MWH Americas, Inc., and used with permission of BMI.)



Physical aspects of the CSM are not discussed in detail in this document, but this document includes specific requirements for ecological components of the CSM. Ecological components of the CSM are designed to show pathways of contaminant transport and exposure to biological receptors. The ecological components of the CSM should be based on an understanding of the site's current and potential biota such that the resulting assessment can evaluate risk to contaminants consistent with the management goals stated above. It is on the basis of the CSM that representative ecological receptors and/or receptor groups are selected and evaluated in the SLERA.

This document is organized as a stepwise description of the SLERA process, consisting of a problem formulation phase, assessment endpoint and screening-level ecological receptor identification (which may be considered as part of the problem formulation phase), an analysis phase, and a screening-level risk characterization and interpretation phase. These phases effectively constitute Steps 1 and 2 of the EPA's (1997) ecological risk assessment (ERA) process. The problem formulation phase consists of data evaluation and characterization, an "ecological scoping" evaluation, and development of the CSM. The selection of ecological receptors and/or receptor groups for evaluation of screening-level ecological risk naturally follows the completion of the CSM. The analysis phase consists of methods for utilizing existing data for the purpose of screening site analytes for each medium in accord with accepted guidances and criteria (as mentioned above) and with respect to identified ecological endpoints

and receptors. The risk characterization and interpretation phase consists of a summary of the screening analysis, identifying chemicals that must be considered as contaminants of potential ecological concern (COPECs), and a discussion of the uncertainties involved in the SLERA process. Uncertainties may consist of identification of data and information gaps that affect the CSM or screening analyses, uncertainties in analysis of any of the chemical components and their effects on site ecological receptors, and uncertainties that arise from assumptions underlying the methods employed for the SLERA process. An evaluation of the identified pathways for COPEC uptake by receptors and uncertainties in the analysis is necessary for COPEC refinement. The final result of the SLERA process is a reduced and focused list of contaminants and representative receptors or receptor groups for further consideration of ecological risk.

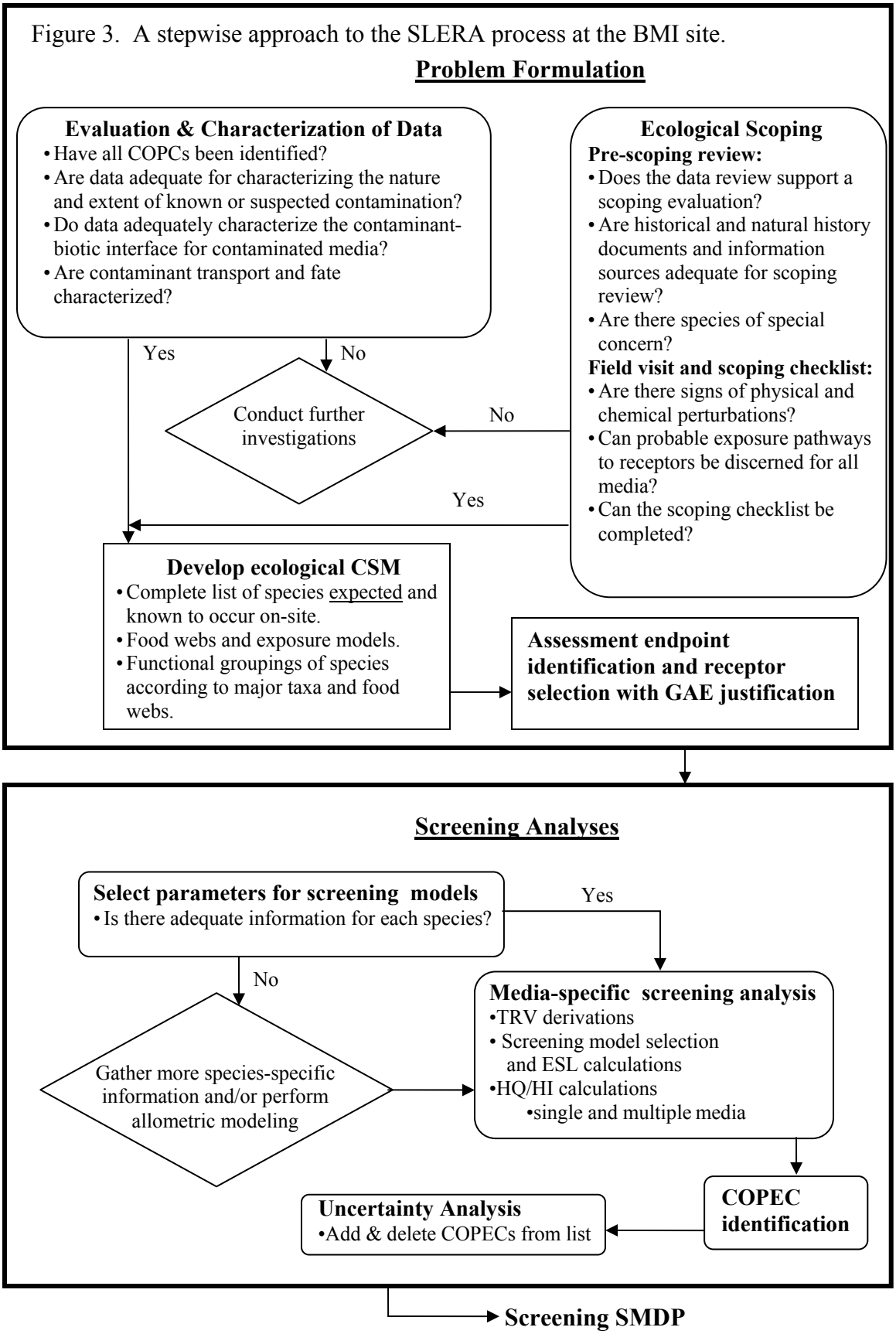
Figure 3 outlines the stepwise approach to SLERA. Further details of the SLERA process are provided in Sections 3.0-6.0 and ensure that consistent methods and decision points for risk management are clearly evaluated with minimal ambiguity of procedure.

The screening SMDP (scientific management decision point) of Figure 3 is a decision nexus from which an evaluation will be made for risk management. As part of the screening SMDP, the risk assessor communicates the results of the SLERA to the risk manager. The risk manager needs to decide whether the information available is adequate to make a risk management decision and might require technical advice from ecological risk assessment experts to reach a decision. There are only three possible decisions at this point:

1. There is adequate information to conclude that ecological risks are negligible and therefore no need for remediation on the basis of ecological risk.
2. The information is not adequate to make a decision at this point, and the ecological risk assessment process will continue to a baseline ecological risk assessment (Step 3 of the ERA process [EPA 1997]).
3. The information indicates a potential for adverse ecological effects, and a more thorough assessment is warranted, and the ecological risk assessment process will continue to a baseline ecological risk assessment (Step 3 of the ERA process [EPA 1997]).

Note that the SMDP made at the end of the screening-level risk calculation will not set preliminary cleanup levels, because screening-level assessments are based on calculations intended to be highly protective and are derived to avoid an underestimation of risk. Requiring a cleanup based solely on screening values would not be technically defensible. Thus, the risk manager should document both the decision arrived at with the SMDP, and all basis for it.

Figure 3. A stepwise approach to the SLERA process at the BMI site.



2.0 Problem Formulation

Problem formulation for the SLERA consists of finalizing identification of contaminants of potential concern (COPCs), data evaluation and characterization, ecological scoping, and completion of the Conceptual Site Model (CSM) as it pertains to the functional ecology and biota of the site. Each of the steps of problem formulation are considered in detail in the following subsections.

2.1 COPC Identification

The identification of COPCs is typically carried out as part of the site characterization process, which precedes the SLERA. Portions of COPC identification that are directly pertinent to the problem formulation phase of the SLERA are included here for completeness. If the list of COPCs for the site has not been finalized, this step should be completed before proceeding to subsequent steps. Completion of COPC identification at this step includes a frequency of detection (FOD) analysis and background comparisons.

2.1.1 Frequency of Detection Analysis

The frequency of detection (FOD) is the number of samples in which a contaminant was positively detected divided by the total number of sample results for that COPC. Given that data provide adequate coverage for the area of concern, analytes that have a detection frequency of less than 5% are eliminated as COPCs from subsequent processes of the screening analysis. The natural caveat to this portion of the analysis is that the geographic extent of sampling cannot exceed that of the known or suspected extent of contamination, nor can known contaminants be distributed saltatorially (spottily) such that detection might be easily missed. Additionally, for there to be an FOD of less than 5%, there must also have been at least 21 samples in the area of known or suspected contamination. Analytes for which analyses do not meet these criteria are passed to the background comparison portion of the screen.

2.1.2 Background Comparison

Establishment of site background concentrations of various chemical constituents, but particularly inorganic elements and compounds, provides a formative basis on which to evaluate exceedence. One simple method to conduct a background comparison is to calculate a single value, usually an upper percentile of the data or an upper confidence limit on the upper percentile of the data. However, there are other methods available for determining if site concentrations are greater then background concentrations. Approaches generally include various data visualization tools, e.g., box plots or other graphical methods, and simple two-sample comparison tests. Some additional information on statistical testing options are discussed below.

2.1.2.1 Distribution Shift Tests

Distribution shift tests compare the potentially affected site data to the entire distribution of background or reference site concentrations, instead of simply comparing to a single threshold value representing background (e.g., the 90th percentile of background). A distribution shift test

is used to determine whether site data is systematically greater than background or reference site data. Several types of distribution shift tests are available. These tests are presented below, and where there are multiple options, the preferred statistical method in each group is indicated.

The result of performing each statistical test on two data sets (one that represents background and one that represents the site) is a test statistic and an associated significance level (also known as a p-value). The significance level is the probability that the test statistic would be as large as or larger than the one produced if the two data sets were from the same distribution (i.e., both were from the background distribution). When the significance level is small, this indicates that it is not likely that the two data sets came from the same distribution. It is standard to consider "small" to be less than 0.05 (i.e., such a large test statistic would occur by chance less than 1 out of 20 times when the sampled populations are the same).

To detect an overall distribution shift between potentially affected sites and background or reference site data, the following tests may be employed: the Student's t-test, the Wilcoxon rank sum test (or Mann-Whitney U-test), or the Gehan modification to the Wilcoxon test. These tests are to be performed as one-sided tests with the null hypothesis that site concentrations are not greater than background (or reference area concentrations), and the alternative hypothesis that the site is greater than background (or reference area concentrations).

Student's t-test. A parametric, two-sample test that determines whether the mean concentration of site data is statistically greater than the mean concentration of background or reference site data (Gilbert 1987). It is the most powerful test when data from both sets are distributed normally. Data analysts should be aware that the t-test performs well for some deviations from normality but, in the absence of normality, increased power may be obtained through nonparametric methods (Miller 1986, p. 40-44). Note that Miller (1986) contains a helpful discussion of the robustness limitations of the t-test. Normality can be assessed visually using a normal probability plot (also known as a "probit" plot or "normal quantile-quantile" plot). Formal tests for normality may be performed first, such as the Shapiro-Wilk W test (Gilbert 1987, p. 158). Results from multi-increment soil sampling are appropriate for the t-test, because they are expected to conform to a normal statistical distribution.

Wilcoxon rank sum test (or Mann-Whitney U-test). This test is the nonparametric equivalent to the t-test (Gilbert 1987, Gilbert and Simpson 1992). The Wilcoxon test pools site and background or reference site data into one aggregate set and determines whether the average rank of the site data is greater than that of the background data. The Wilcoxon rank sum test is recommended when non-detects are relatively infrequent (less than 10%) and all have the same detection limit. The non-detects are treated as tied at a value less than the smallest detected concentration. The Wilcoxon rank sum test will have about the same or more power than the t-test for most distributions (Gilbert and Simpson 1992).

Gehan modification to the Wilcoxon rank sum test. When, as is frequently the case for environmental data, some of the data are "censored" or reported as below a detection limit, and especially when not all the detection limits are identical, the Gehan modification to the Wilcoxon test is useful (Gehan 1965). The Gehan test uses a modified ranking of sample results to accommodate non-detected values together with detected values, and then applies the Wilcoxon

rank sum test. The Gehan test is recommended when non-detects are relatively frequent (greater than 10% and less than 50%). It handles data sets with non-detects reported at multiple detection limits in a statistically robust manner (Gehan 1965, Millard and Deverel 1988). The Gehan test is not recommended if either of the two data sets has more than 50% non-detects. The Gehan test is identical to the Wilcoxon rank sum test when applied to results containing no non-detects. The Gehan test is the preferred test to the Wilcoxon rank sum test because of its applicability to a majority of environmental data sets.

To detect distribution shifts between the upper range of the potentially affected site data and the background or reference site data, the following distribution shift tests may be employed: the quantile test or the slippage test. The quantile and slippage tests are discussed below.

Quantile test. The quantile test determines whether more of the observations in the top 20% (or other chosen percentile) of the combined site and background (or reference site) data sets come from the site data set than would be expected by chance, given the relative sizes of the site and background data sets. If the relative proportion of the two populations being tested is different in the top 20% of the data than in the remainder of the data, the distributions may be partially shifted due to a subset of site data. This test is capable of detecting a statistical difference when only a small number of potentially affected site concentrations are elevated (Gilbert and Simpson 1992). The quantile test is the most useful distribution shift test for potentially affected sites at which samples from a release represent a small fraction of the overall data collected. The quantile test is applied at a pre-specified quantile or threshold, and we have selected 80th percentile for this project. The test cannot be performed if more than 80% (or, in general, more than the chosen percentile) of the combined data are non-detected values. It can be used when the frequency of non-detects is approximately the same as the quantile being tested. For example, in a case with 75% non-detects in the combined background or reference site and potentially affected site data set, application of a quantile test comparing 80th percentiles is appropriate. The threshold percentage can be adjusted to accommodate the detection rate of an analyte, or to look for differences further into the distribution tails. The quantile test is more powerful than the Wilcoxon (or Gehan) test for detecting differences when only a small percentage of the potentially affected site concentrations are elevated.

Slippage test. This test is based on the maximum observed concentration in the background or reference site data set and the number ("n") of potentially affected site concentrations that exceed the maximum concentration in the background or reference data set (Gilbert and Simpson 1990, pp. 5-8). The result (p-value) of the slippage test is the probability that "n" site samples (or more) exceed the maximum background or reference site concentration by chance alone. The test accounts for the number of samples in each data set (number of samples from the site and number of samples from background or the reference site) and determines the probability of "n" (or more) exceedences (of the maximum background or reference site concentration) if the two data sets came from identical distributions.

2.2 Data Evaluation and Characterization

Data evaluation and characterization are central to the ecological scoping process and the development of the Conceptual Site Model (CSM). Site-specific data must be deemed adequate

to characterize the nature and extent of contamination based on site-specific data quality objectives (DQOs). The DQOs in support of the SLERA must address the type, amount, and quality of environmental data needed for decision-making. Data that are reviewed for the SLERA should be done so in light of DQO criteria. Specifically, the DQOs should provide the rationale for the statistics used to characterize contaminant concentrations, including the sampling design basis (i.e., statistical or judgmental approaches), the basis for data analysis, and how data and accompanying statistics will be utilized to evaluate risk potential in the SLERA. Generally, data adequacy involves determining the spatial and temporal contaminant concentrations for all media identified in the CSM. If a potentially contaminated medium identified in the CSM is not directly sampled, then it must be stated in the DQOs how it will be evaluated. Concurrence of data adequacy must be achieved before proceeding with any further SLERA analyses.

Data adequacy for the SLERA can be addressed with the following checklist:

- All inputs to the SLERA decision have been identified including representative receptors, species of special concern, and contaminants of potential concern (COPC).
- Spatial coverage of data representing the site is complete, and all statistical criteria (sample size, location, and distributional criteria) for site characterization have been identified.
- Temporal coverage is evaluated and addressed as needed, including the potential for off-site transport of contaminants and contaminated media.
- Sample coverage is complete for all media accessible to site biota and appropriate detection limits have been developed and subsequently attained. Quantitation of each COPC in each medium is consistent and comparable in terms of reported units, and units are consistent with those requires for risk-based calculations.
- Methods for eliminating COPCs based on detection frequency or background comparisons have been detailed in the DQO documentation.
- At minimum, data are adequate to provide a reasonable maximum concentration of each COPC for the screening evaluation in each medium. If other statistics are used, for instance the 95% upper confidence limit of the mean, then the calculational methodology, including the methods for handling non-detect values, must be specified in the DQO documentation. One may refer to EPA (2002b) for additional guidance on statistical methods for calculating representative concentrations for use in risk assessments.

2.3 Ecological Scoping

Ecological scoping involves coupling data evaluation and characterization with a site visit by the risk assessment team in order to complete the Scoping Checklist (Appendix A). The Scoping Checklist is designed to aid the risk assessor in identifying potential pathways of contaminant conveyance to receptors that may occur on a given site. A series of questions in the checklist is intended to guide a risk assessor to a highly descriptive and conceptual treatment of the potential for contaminant uptake and/or contact by any of the biota of the site under consideration. Contact may occur in one or more medium, and may occur due to ingestion, imbibition, inhalation, and/or dermal contact. If a site is contaminated with radionuclides, contact may be indirect, resulting from ionizing radiation.

Adequate completion of the Scoping Checklist requires thorough knowledge of the plant and wildlife species currently inhabiting the site or that may potentially inhabit the site in the future (including any species of special concern, such as state or federally listed endangered or threatened species), as well as a thorough knowledge of the site's history in the context of potential contamination. Completion of the scoping checklist includes:

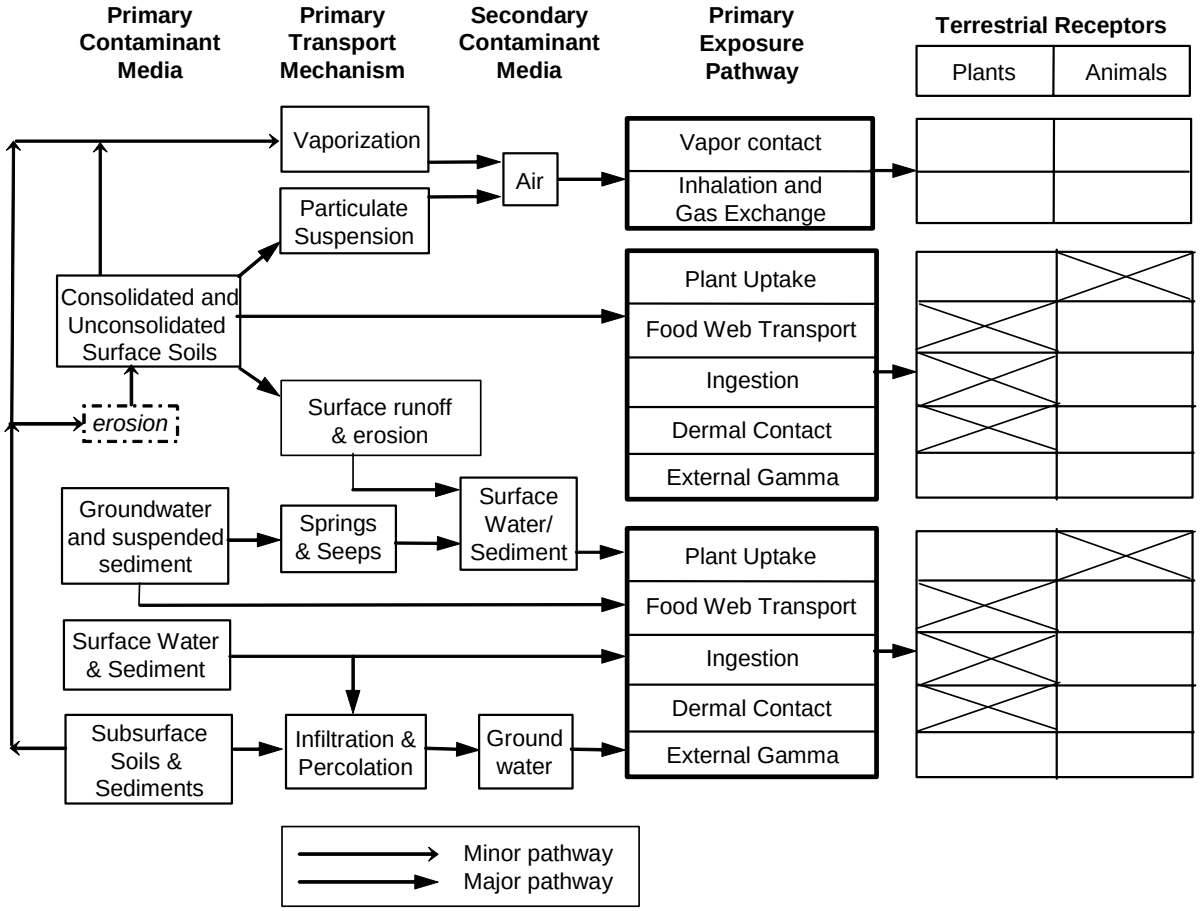
- geographic delineation of a site or sites of (potential) contamination documented with survey coordinates and mapped, typically using GIS software;
- specification of the current and potential biotic community;
- nature of contaminant releases and impacted media, including horizontal and vertical extent;
- evaluation of data adequacy, including sample locations and a visual inspection relative to site identification, media, and potential transport processes;
- identification of the presence or absence of ecological receptors at the site, including associations with known potential contaminant release sites and/or contaminated media, as well as physical disturbance of the site.

Note that sites that lack receptors due to physical or biotic disturbances, or evident contamination *do not* qualify as sites for which no SLERA is completed. Additionally, seasonal effects on biotic activity and biotic composition at any given site must be considered when a site visit is conducted as part of the scoping process.

Completion of the Scoping Checklist includes the completion of the Ecological Pathways Conceptual Exposure Model (EPCEM). The EPCEM is a high-level conceptual model that depicts the pathways of contact that the biota of the site potentially interact with contaminated media. The EPCEM forms an integral part of the biological address of the CSM. The EPCEM for terrestrial and aquatic organisms are presented in Figures 4 and 5. In the course of developing the EPCEM, the risk assessor must consider plants and wildlife that currently inhabit or may potentially inhabit the site given the effects of attenuation and/or restoration. For example, the site may not currently have desert tortoise, but the site may represent acceptable habitat for desert tortoise or the habitat could be restored to acceptable desert tortoise habitat in the future. The reason for this depth of consideration is that without a clear understanding of the biotic conditions that may realistically occur at the site following remediation or natural attenuation, the process for identifying potential contaminant effects on biota of the site is compromised by a present-day view of community composition, structure, and function.

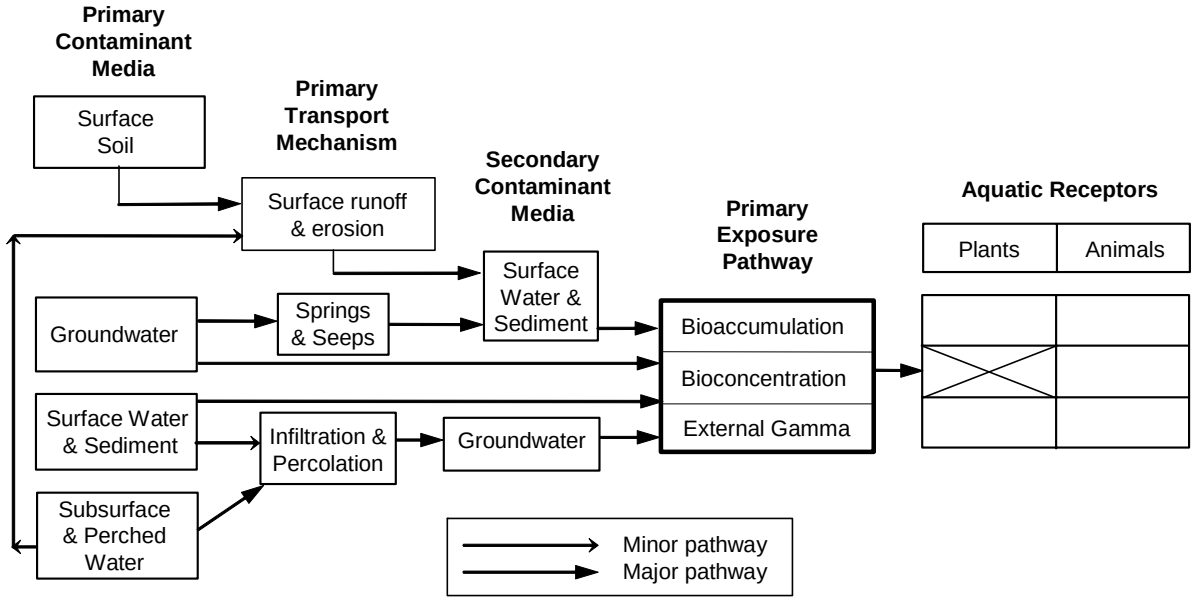
To develop the EPCEM, organisms should be understood in terms of the media with which they may come in contact, including the primary and secondary modalities of contact. As mentioned, this knowledge should extend to a description of the site's current and potential biota.

Figure 4. An ecological pathways conceptual exposure model for terrestrial receptors.



Boxes marked with "X"s indicate incomplete pathways. Open boxes indicate potentially complete pathways.

Figure 5. An ecological pathways conceptual exposure model for aquatic receptors.



Boxes marked with "X"s indicate incomplete pathways. Open boxes indicate potentially complete pathways.

2.4 Conceptual Site Model

The scoping evaluation is integral to the continuing development of the CSM. As discussed previously, the biological components of the CSM should include a description of the current and potential biota for the site, data adequacy for screening analyses, biotic relationships with the site under consideration, contaminant distribution, and the potential for contaminant contact by biota, as illustrated by the EPCEMs developed during ecological scoping. For the "no-build" area of the BMI Complex, two independent reports show the greatest site-specificity for describing the plants and wildlife: 1) *Audubon International's Ecological Design, Provenance, Henderson, Nevada* (Woolbright et al. 2000); 2) *Draft Biological Constraints Report and Wildlife Corridor Suitability Analysis for Provenance, A Green community in Henderson, Nevada* (EDAW 2002). The biota listed in these reports include vascular plants, mammals, birds, reptiles, and fish. No mention is made in either report of invertebrates (either aquatic or terrestrial) or amphibians, although both groups are known for representative species on the BMI Complex site (Nelson and Roline 2005, SWCA 2005). Other resources for description of the biotic communities applicable to the BMI Complex may also be available, such as publications by the Las Vegas Wash Coordination Committee, more general published descriptions of northern Mojave Desert flora and fauna, as well as electronically-based publications (e.g., State of Nevada Department of Conservation and Natural Resources Natural Heritage Program, found at http://heritage.nv.gov/index.htm). The U.S. Fish and Wildlife Service should be consulted for record of any species of special concern in the area (such as those listed in EDAW 2002, Tables B-1 and B-2), and regional experts may be consulted for expertise and/or concerns regarding biotic inventory.

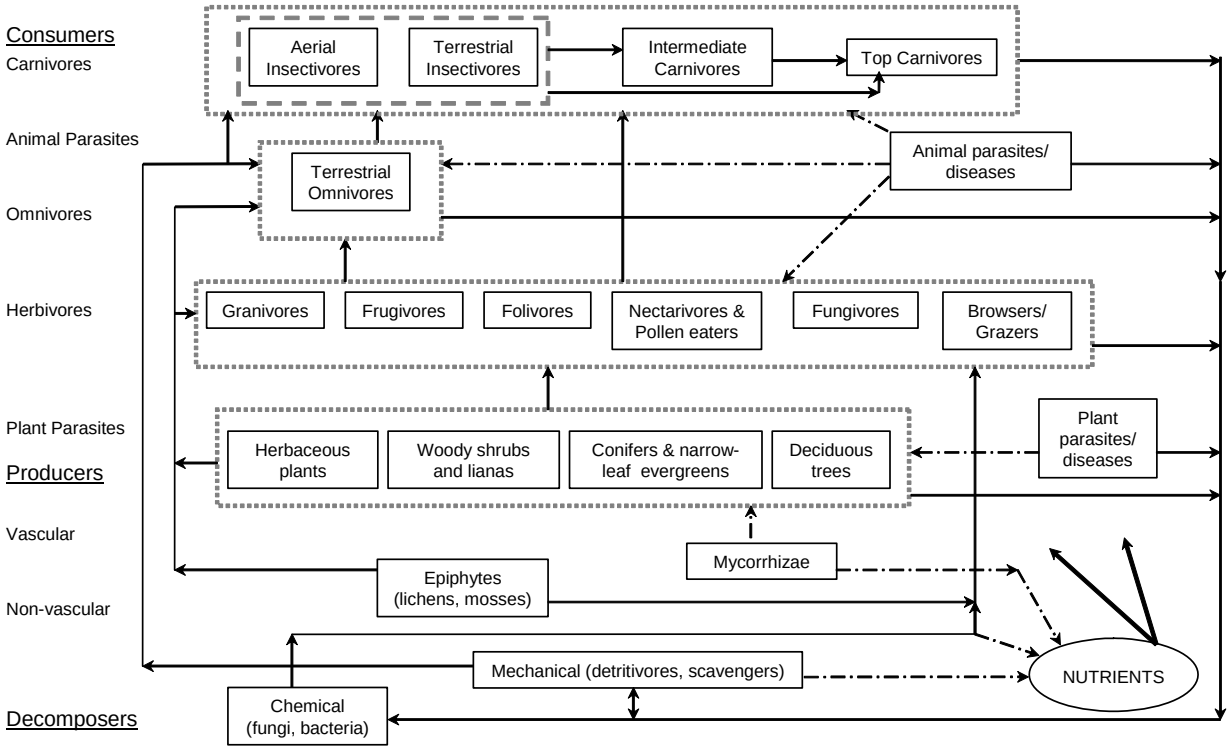
2.4.1 Functional Food Webs

In order to conceptually organize the biotic inventory for the CSM, it is useful to employ a functional foodweb approach (LANL 2004, Hampton et al. 1998, Kester et al 1998, Reagan et al. 1999, VanHorn et al. 1998). The functional food webs (Figures 6 and 7) aid in evaluating dietary exposure pathways and specifying ecologically relevant groups of organisms for an exposure assessment. The food web structure captures the functionally relevant biotic assimilation and associative relationships between organismal groups in context of trophic categories and feeding relationships, and is ultimately important for receptor selection. A food web diagram depicts pathways of food consumption (and energetic flow) in a biotic system by means of boxes and connecting arrows. Boxes in a functional food web diagram represent biotic groups organized by feeding guilds, which are organisms grouped by similar feeding roles in various trophic positions in the diagram. Feeding guilds in a functional food web diagram represent functional assemblages and *do not* represent taxonomic groups. (An entire taxonomic group, e.g., Class *Aves* or *Mammalia*) may be organized into a functional food web diagram.) Arrows in a food web diagram define the major direction of energy flow between biota, e.g., from producers to primary consumers to predators, and ultimately back to decomposers and nutrient pools in the medium of consideration.

A food web organized by trophic categories and feeding guilds forms a basis for receptor selection. Receptor selection has traditionally been done by selecting individual species as

representatives of major taxa from each guild or trophic category (e.g., EPA 2003a, LANL 2004). A more generalized approach utilizes pooled information from each major taxon of consideration for each functional feeding guild within each of the trophic categories (Hampton et al. 1998). In either case, fundamental taxonomic considerations must be made for each of the functional food web models (Figures 6 and 7), and organisms chosen as representatives of major taxa or functional feeding categories should be taxonomically and functionally similar so as to provide coverage for all species in the guild.

Figure 6. A terrestrial food web organized by trophic categories and functional feeding guilds.

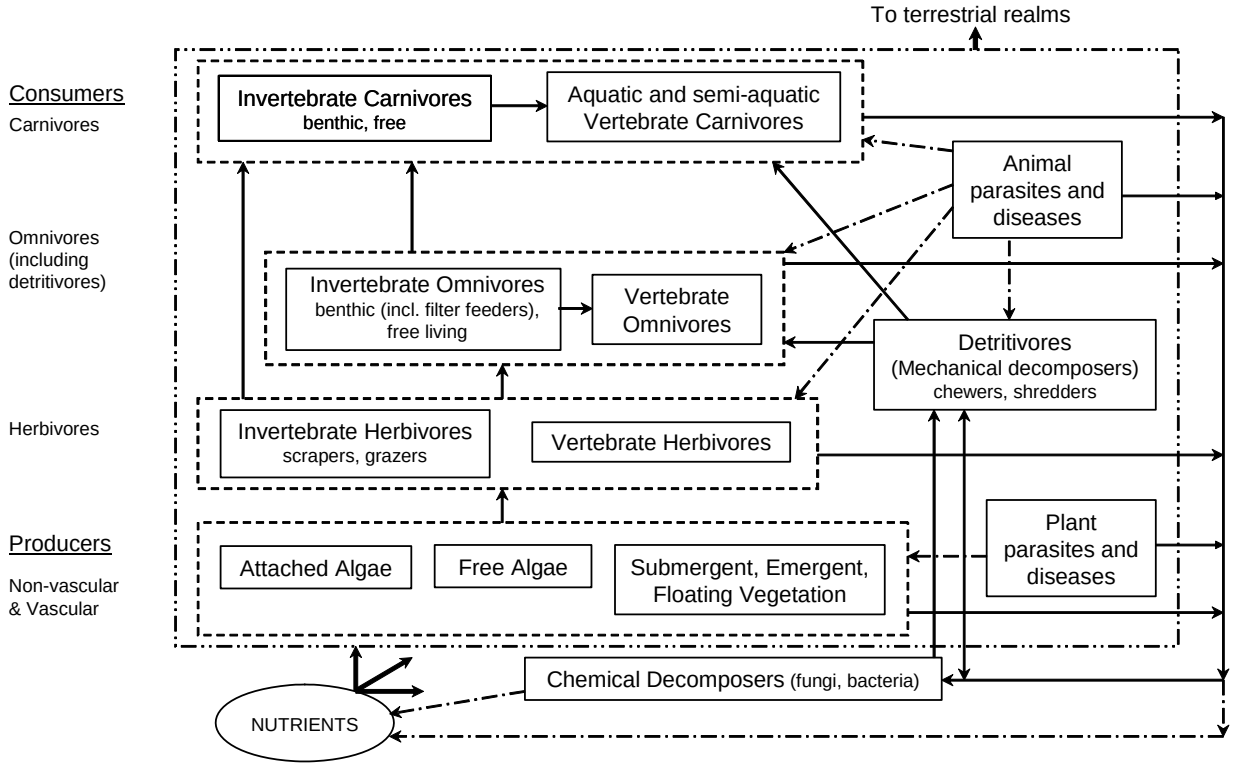


As seen in Figures 6 and 7, the food webs include three fundamental trophic positions: producers (vascular and nonvascular plants); consumers (herbivores, omnivores, carnivores, and parasites); and decomposers. Within these basic trophic levels, finer positions in the hierarchy of trophic categorization may exist, e.g., primary consumers, secondary consumers, and so on. For the purpose of this methodology, the highest trophic category that will be considered will be the category of secondary consumer, which in this case is defined as any of the purely flesh-eating and insectivore categories.

It is useful to complete a food web for each realm (terrestrial and aquatic) and for each of the major taxonomic groups. For both the terrestrial and aquatic realms, the major taxonomic considerations are very broad for the screening level assessment. Taxonomic categories are chosen, in large part, due to meaningful applicability to site receptors and potential receptors. Additionally, toxicological data is limited in applicability to within taxonomic class (Hampton et al. 1998). Generally speaking, taxonomic categories for consideration in the terrestrial realm include macrophytic producers (primarily vascular plants), soil-dwelling invertebrates (but which

may concern multiple classes or phyla), and vertebrates, including mammals, birds, reptiles, and amphibians (terrestrial phase). In the aquatic realm, taxonomic categories typically include macrophytic producers (macrophytic green algaes and vascular plants), aquatic invertebrates (but which may concern multiple classes or phyla), and vertebrates, including fish, mammals, birds, reptiles, and amphibians (aquatic phase).

Figure 7. An aquatic food web organized by trophic categories and functional feeding guilds.



For the "no-build" area of the BMI Complex, the major taxonomic categories for consideration in both terrestrial and aquatic realms will include macrophytic producers, invertebrates, mammals, birds, reptiles (terrestrial only), amphibians, and fish (aquatic only). Any or all of the taxonomic groups may require consideration of species whose life cycle has terrestrial and aquatic phases. In the context of the development of the EPCEMs (for scoping) and the CSM, each of these broad taxonomic categories will be considered across each of the functional feeding categories, as applicable to the site's current and potential flora and fauna. Of course, there is taxonomic overlap in both realms, and special consideration may be required for species that occupy one realm and then another for different parts of their life cycle.

Microorganisms in soil and water may be of considerable concern, but are typically not considered in a SLERA (EPA 2003a). If concerns for microorganisms arise for the consideration of ecological risk, then this group of organisms should be considered in a baseline ecological risk assessment (EPA 1997 Step 3, EPA 2003a Attachment 1-2).

The following trophic categories will be considered as terrestrial receptor (functional) groups, thus providing broadly protective coverage for each of the major taxa according to the functional food web categories[1]:

- photosynthetic macrophytic plants
- invertebrates as primary decomposers (soil dwelling herbivores and detritivores)
- mammals:
    - herbivores (all plant diet)
    - omnivores (mixed plant, flesh, and invertebrate diet)
    - insectivore (all invertebrate diet)
    - carnivores (all flesh diet)
- birds:
    - herbivores (all plant diet)
    - omnivores (mixed plant, flesh, and invertebrate diet)
    - insectivore (all invertebrate diet)
    - carnivores (all flesh diet)
- reptiles
    - herbivores (all plant diet)
    - insectivore (all invertebrate diet)
    - carnivores (all flesh diet)
- amphibians (land phase)
    - insectivore (all invertebrate diet)

[1]Categories will be considered as applicable (if populated with one or more species) and according to the known and potential flora and fauna of the BMI Complex.

Similarly, for the aquatic realm, the following categories will be considered as aquatic receptor groups, thus providing broadly protective coverage for each of the taxa[1]:

- photosynthetic macrophytic plants:
    - sessile (anchored in sediments or near-shore soils)
    - free (floating in the water column or anchored only to rocks with no sediment-based nutrient requirements)
- invertebrates as primary consumers and decomposers
    - free water column and macrophytic surfaces
    - sediment and lithic surfaces
- fish
    - herbivores
    - insectivores
    - carnivores
- amphibians (water phase)
    - herbivores
    - insectivores
    - carnivores
- birds
    - aerial insectivores
    - carnivores (piscivores and others)

- mammals
    - aerial insectivores (bats)
    - carnivores (piscivores and others)

[(1)]Categories will be considered as applicable (if populated with one or more species) and according to the known and potential flora and fauna of the BMI Complex.

2.4.2 Contaminant Exposure Modalities

In the context of contaminant exposure and completion of the EPCEMs, consideration of all major taxa requires a list of species known to occur or may potentially occur on the no-build area of the BMI Complex, their primary biotic realms, and their primary functional feeding (trophic) categories. Once all plant and wildlife species known to occur or may occur on-site have been categorized by major taxon, then by trophic category, then the modality of exposure to contaminated media can be considered. Organismal development and behavior dictate the primary pathways for chronic exposure to contamination. One must bear in mind that acute exposure to contamination is a secondary consideration in a SLERA to the broader concerns of chronic exposure due to organismal development and behavior (see Section 4.2.1).

Organismal development bears upon contaminant exposure when considering where an organism carries out its various life stages. For most organisms, one can broadly consider development in the terrestrial or aquatic realm. In each realm, the primary mode of contaminant absorption is dictated by an organism's interaction with the contaminated medium in which it carries out daily functions of growth, maintenance and reproduction. Again, for most organisms, development through all life stages is carried out in a single realm, and interactions with various physical media is a consequence of developmental constraints. For example, terrestrial plants are sessile organisms whose development is divided between two primary media: soil and air. Sorptive tissues for terrestrial plant nutrition are in the soil (root systems) and in the air (leafy tissues). Phreatophytic plants may access groundwater resources for sorption of water. For aquatic plants, sorption of nutrients may be from aquatic sediments and near-shore soils, the water column, and the air. Thus, for plants, the primary contaminant exposure pathway is by sorption from the medium in which the plant is anchored and harbored, both of which are developmentally dictated.

Concern arises, however, if an organism spends one portion of its life cycle in one medium (e.g., water) and another portion of its life cycle in another medium (e.g. on dry land), such as many amphibians. In the western United States, toads, some species of frogs and salamanders are examples of organisms that exhibit developmental differences in their occupation of terrestrial and aquatic realms in the course of their natural life cycles. Some species of invertebrates may also exhibit such differences, although most invertebrates are more consequentially exposed to contaminants in one medium or another due to a dominance of developmental and phenological traits. Species that occupy dual realms according to developmental conditions may require consideration for each realm in the SLERA.

Behavioral modalities of contaminant exposure and uptake tend to dominate over developmental modalities for most vertebrate species. (Exceptions, of course, may be for anurans, salamanders, and fish, all of whom absorb dissolved chemicals across skin and/or gill surfaces.) All vertebrate species are consumers (primary, secondary, etc.), and are therefore vulnerable to contaminant

uptake by ingestion and imbibition. These exposure pathways tend to dominate by direct consumption of contaminated media and/or by exposure to contaminants by food chain transfer. Other behavioral modalities of contaminant exposure include bathing and preening (dermal exposure), and inhalation of airborne particulates and volatilized constituents.

Some vertebrates spend considerable portions of days or seasons in and out of terrestrial and aquatic realms as part of natural conditions of feeding, shelter obtention, and/or reproductive behavior. Species that occupy dual realms according to behavioral conditions may require consideration for each realm in the SLERA.

2.4.3 Exposure Modalities Across Major Taxa

For plants, exposure is mediated (as outlined above) by direct contact with contaminated media (soil and/or sediment, and water). Unless data exist that specifically relate the effects of contaminant exposure to particular plant species that occupy the BMI site, then plant receptors will be considered "generic" plants in two categories: (1) all vascular, photosynthetic terrestrial plants; and (2) all macrophytic, photosynthetic aquatic plants. For aquatic plants, exposure to potentially contaminated sediment requires that pertinent exposure information be derived from sources that considered aquatic plant forms anchored in sediments and near-shore soils. Free-floating vegetation may be considered solely with respect to contaminants in the free-water column. Data in support of exposure calculations must be derived from sources appropriate to the realm that is occupied by the "generic" plant receptors. Groundwater that is accessible as a nutritive resource to terrestrial phreatophytes will also be considered for contaminant exposure for plant receptors.

For invertebrates, exposure can vary considerably due to developmental aspects of any given species. In general, two modes of exposure to invertebrates will be considered as primary among all modalities: (1) for terrestrial invertebrates, direct contact and sorption across dermal membranes from soil; and (2) for aquatic invertebrates, direct contact with contaminated sediments and also from the free-water column. For the BMI Complex, a "generic" invertebrate will be considered with respect to the terrestrial (soil) and aquatic (free water and sediment) realms. If applicable, groundwater that is accessible to invertebrate receptors as a nutritive or habitat resource will also be considered for contaminant exposure to invertebrate receptor groups. Data in support of exposure calculations must be derived from sources appropriate to the realm that is occupied by the "generic" invertebrate receptors.

The primary modality of contaminant exposure for vertebrate receptors is considered the ingestion and imbibition pathways (EPA 2003a Attachment 1-3). The vertebrate receptor groups outlined in the "Functional Food Webs" section will be considered for the ingestion and imbibition pathways at the BMI Complex, in their respective terrestrial and aquatic realms. Media of concern are soils, sediments, and surface water that acts as a habitat or nutritive resource. If applicable, groundwater that is accessible to terrestrial receptors as a nutritive resource will also be considered for contaminant exposure to vertebrate receptor groups.

In addition to the receptor groups identified above, receptor groups that will be considered due to special developmental or behavioral concerns will include fossorial animals. This category is not

explicitly covered by the functional food web analysis, but is of evident consideration in the EPCEM for terrestrial receptors (Figure 6). Fossorial animals spend large portions of time in belowground burrows and tunnels due to obligate foraging and/or reproductive behavioral habits. Burrows and tunnels are engineered for conductive air flow, but such habits may expose fossorial animals to vapor phase contaminants, including volatile organic chemicals (VOCs), that are more highly concentrated than in air at or above the ground surface. While VOCs do not generally attain levels that can adversely affect organisms in well-mixed surface air, they have greater potential to affect organisms exposed to subsurface air, where vapor concentrations can rise above that of ambient (surface) air. For example, the antelope ground squirrel *(Ammospermophilus leucurus)* and the Merriam's kangaroo rat *(Dipodomys merriami)*, both of which are recorded to occur on or near the BMI Complex or the areas affected by the BMI Complex, spend vast amounts of time underground in burrows, and also raise young in underground dens. Larger mammals, including coyote *(Canis latrans)*, blacktail jackrabbit *(Lepus californicus)*, and desert cottontail *(Sylvilagus auduboni)*, all of which are recorded from the BMI complex, are known to den underground, and many predators of fossorial mammals may also spend a measurable portion of their time in belowground burrows. Additionally, many reptiles spend large portions of time belowground, such as the banded gecko. Burrowing owls, a common species in the Las Vegas region (also highly tolerant of urban environments), spend substantial amounts of time belowground.

Fossorial mammals will be considered as receptors in an inhalation pathway for any VOCs of concern in soil. Albeit organisms other than mammals are also exposed to contaminants in subsurface air, the lack of toxicity data for non-mammalian receptors makes it difficult in practice to evaluate other taxa. Fossorial mammals will be the representative receptors for subsurface air contaminants for all taxonomic groups, because fossorial mammals can be assumed to spend 100% of their time belowground. In addition, respiratory and respiration rates in mammals are typically greater than that for other fossorial organisms, such as reptiles and invertebrates, and likely equivalent to those for birds. Thus, the use of fossorial mammals as the sole receptor group for fossorial mammals appears to be a conservative choice for receptor selection covering fossorial animals, in general, in the context of VOC contamination.

Other inhalation pathways, particularly that for contaminants carried by fugitive dust, are considerable in the EPCEM, but are also considered relatively minor compared to the ingestion and imbibition pathways in the SLERA (EPA 2003a Attachment 1-3). Similarly, dermal contact of mammalian and avian receptors with contaminated soil and water for this SLERA analysis is considered a minor exposure pathway compared with the ingestion and imbibition pathways (EPA 2003a Attachment 1-3). Therefore, these analyses will not be formally modeled, although some discussion of their contributions to contaminant exposure will be required in the uncertainty analysis portion of the SLERA.

2.4.4 CSM Summary

The CSM identifies spatial and ecological scales at which drivers for risk may be operating. For site biota, the end result of the CSM is a diagrammatic exposition and descriptive explanation of the pathways of contaminant exposure (an exposure scenario) for the biotic communities, populations, species, and in particular, species of special concern. The CSM must therefore

begin with an understanding of the biota, as well as a categorization of the species occupying a given site by major taxa, trophic categories, and functional feeding guilds. Supplemental to taxonomic and functional categorization are considerations of the developmental and behavioral modalities of contaminant exposure for each of the taxa across functional feeding guilds. These understandings are used to develop appropriate profiles for organismal exposure to contaminated media, as depicted in the EPCEMs. Interpretation of the resulting CSM will aid in defining the appropriate model for contaminant uptake to employ in the screening analyses for each of the taxa across functional feeding guilds.

3.0 Screening-Level Assessment Endpoints and Receptor Selection

After completion of the Problem Formulation phase of the SLERA and prior to the screening analysis, assessment endpoints require identification and receptors require selection. The identification and selection of screening level assessment endpoints and receptors follows from the delineation of management goals (see Introduction) and general assessment endpoints (GAEs; discussed below and in Appendix B), and the completion of the CSM. To reiterate, management goals specifically applicable to assessment endpoints and receptor selection are:

1. Protect wildlife populations that are either currently present or may inhabit the site in the future, based on habitat mitigation or restoration efforts.
2. Protect special status species that are currently present or may inhabit the site in the future, based on habitat mitigation or restoration efforts.

Management goals and GAEs go hand-in-hand, as is clear from a definition of GAEs from Reagan et al. (1999):

> *GAEs are intended to reflect ecological values of broad significance to risk managers and other stakeholders. GAEs encompass ecological and human use values at all levels of ecological organization (ecosystems, communities, and individual species).*

The EPA (2003b) recognizes GAEs (which they call GEAEs for generic ecological assessment endpoints) as *"generally applicable to a wide range of ecological risk assessments because they reflect the programmatic goals of the Agency* [EPA]*, they are applicable to a wide variety of environmental issues, and they may be estimated using existing assessment tools."* GAEs and management goals provide defining value conditions for the specification of assessment endpoints for the ecological risk assessment process.

The EPA defines assessment endpoints (AEs) and related exposure terminology in their 1998a guidance (EPA 1998a, Appendix A). An assessment endpoint is defined as *"An explicit expression of the environmental value that is to be protected, operationally defined by an ecological entity and its attributes."* An ecological entity may be an individual organism, a population, a species, a habitat, a community, or another relevant and ecologically differentiable object or set, each of which is naturally derived as an outcome of the stated management goals and/or GAEs. *Attributes* of the ecological entity include those biological matters that may be

concerned with longevity, survival, reproduction, behavior, population or community viability and structure, and all attending characteristics of form and function.

The EPA (1998a) defines receptor as *"the ecological entity exposed to the stressor. This term may refer to tissues, organisms, populations, communities, and ecosystems."* Also in accord with EPA (1998a), stressor is defined as *"any physical, chemical or biological entity that can induce an adverse response."* Thus, *"exposure is the contact or co-occurrence of a stressor with a receptor"* (EPA 1998a). An "exposure factor" (EPA 1993) is a model parameter (quantity) used to calculate the quantitative level of exposure of a particular chemical to a receptor via a specific conveyance (e.g., dietary or incidental ingestion, dermal uptake, etc.). It is important to bear in mind the broad applicability of these definitions when selecting ecological receptors for the SLERA, which is done below.

For the BMI Complex, receptors will be defined and selected according to (1) the taxonomic group in question and (2) the functional group to which an organism belongs. Receptors will be either representative species within each major taxon and functional group, or they will be "exposure factors" that are used to calculate the hazard quotient or hazard index (Section 4.2.3) for any given functional group within a major taxon. These concepts will be explained and expanded upon below, as well as within sections that describe methods for assessment of screening-level hazard calculations for the various media of concern.

Since approaches to screening-level calculations and criteria differ for the different combinations of taxa, functional groups, and media, the assessment endpoints and receptors for each specific combination also differ. Each of the combinations are outlined in the following subsections and the rationale are provided for each of the assessment endpoint and receptor selections.

3.1 Terrestrial Receptors and Assessment Methods

- ***Plants***

The remarkable phylogenetic diversity of plants that comprise desert floras is not well represented by any single species that may occupy a given geographic region. This diversity is even more poorly represented in our detailed understanding of the complex chemical relationships that the various plant species may have with their environment. Thus, when it comes to toxicological examinations of plant species, these are typically concerned with species important for human consumption and do not concern wild plants.

Due to the non-specificity of wild plants for the majority of toxicological studies on plant life, a "generic" terrestrial plant will represent all plants of the terrestrial realm as a receptor. For any given contaminant, therefore, the full phylogenetic diversity of plants will be considered for the derivation of exposure parameters that are pertinent to screening analyses (see subsequent sections). As a consequence of these broad considerations, the full breadth of toxicological literature germane to terrestrial plants will be utilized in support of screening-level analyses. Those studies that are considered germane, their assessment and related measurement endpoints, and derived exposure parameter values relevant to screening-level analyses, will be explicitly documented in support of the SLERA.

- ***Invertebrates***

Similar to plant life, invertebrates also represent a phylogenetically diverse and complex set of receptors. In desert ecosystems, invertebrates are remarkably diverse and the levels of their terrestrial associations span all levels of functional groups, save producers. Also similar to plants, scientific understanding of the complex chemical relationships that invertebrates may have with the environment are poorly understood. Toxicological studies on invertebrates are typically limited to agriculturally or horticulturally important species, and the vast diversity of species associated with native ecosystems have gone greatly overlooked.

Due to the non-specificity of invertebrates for the majority of toxicological studies on invertebrate life, a "generic" invertebrate will represent all invertebrates of the terrestrial realm as a receptor. For any given contaminant, therefore, the full phylogenetic diversity of invertebrates will be considered for the derivation of exposure parameters that are pertinent to screening analyses (see subsequent sections). As a consequence of these broad considerations, the full breadth of toxicological literature germane to invertebrates will be utilized in support of screening-level analyses. Those studies that are considered germane, their assessment and related measurement endpoints, and derived exposure parameter values relevant to screening-level analyses, will be explicitly documented in support of the SLERA.

- ***Vertebrates***

For each of the terrestrial vertebrate groups (Mammals, Birds, Reptiles, Amphibians), many more studies of the effects of toxins on vertebrate life have been conducted that are relevant to species found in natural environments than are similarly available for wild plants and invertebrates. Due to the breadth and depth of toxicological study on vertebrate species, the level of phylogenetic similarity that exists between test species and wild counterparts is vastly improved over the same for plants and invertebrates. As a result of improved toxicological understandings and relevance of test subjects to wild species, the selection of vertebrate receptors and the attending development of relevant exposure parameters for the SLERA, is also greatly improved.

In the case of terrestrial vertebrates, receptors that are representative of any single functional food category may be either a representative species (and attending exposure parameters) or a set of exposure parameters that represent an agglomerate specific to the functional group, as with Hampton et al. (1998) and the methods used for the INEEL. Full agreement between NDEP and BMI Complex stakeholders must be made for receptor selection, for each taxonomic and functional group combination, in advance of performing the screening analysis.

In the case of utilizing a representative species (*a.k.a.* surrogate species) as a receptor for a single taxonomic and functional group combination in the SLERA, protocol for selection of a representative species will follow those appropriate for the final methods chosen for screening-level analyses and the calculation of hazard quotients and indices (Section 3.0). For example, if Eco-SSL methods are utilized for soil screening of a particular contaminant, then the protocol for selecting representative species as outlined by EPA (2005) and supporting documentation will be

employed; likewise, if another methodology is used for soil screening analyses, then the appropriate protocol for selection of a representative species will be set forth and clearly documented. Development of exposure parameters specific to the representative species will be fully documented and supported according to the methods of the screening-level analyses.

In the case of utilizing an approach similar to Hampton et al. (1998) for receptor selection, the following protocol will be used for the development of exposure parameters and the attending calculation of hazard quotients and indices:

- Highest proportion of soil (as proportion of diet by dry weight) ingested for all receptors in a functional feeding group.
- Highest ingestion rate (dry weight) to body weight (dry weight) ratio for all receptors in a functional feeding group.
- Highest imbibition rate (water) to body weight (dry weight) ratio for all receptors in a functional feeding group.
- Chronic exposure duration set to 1.0 (year-round).
- The contaminated area will be considered to comprise 100% of the representative "species'" home range.

If in the case that representative species are chosen for trans-class taxonomic representation (e.g. a bird used to represent a reptile), then a clearly conservative approach to receptor selection and attending exposure parameter estimation will be set forth.

All justifications for receptor selection and attendant exposure parameter development should be thoroughly documented for the SLERA. Attendant uncertainties for all receptor selections will be fully discussed in the Uncertainty Analysis (Section 5.0) of the SLERA.

3.2 Aquatic Receptors and Assessment Methods

➢ ***Plants***

Photosynthetic autotrophs (producers) in the aquatic environments of desert ecosystems are less species-diverse in comparison with terrestrial realms, however, the phylogenetic diversity of aquatic plants can be extraordinary. For example, green algaes (which are commonly encountered in desert springs) are not even considered "plants" by some modern systems of classification, and in the most inclusive taxonomy belong to an entirely different taxonomic division than angiosperms (flowering plants), such as duckweed *(Lemna minor),* another common spring inhabitant.

In terms of screening-level risk assessment, however, the most basic concern is that criteria exist that are broadly inclusive of aquatic autotrophic taxa, and that consideration be given to the form of contamination and the primary modality of exposure (to sediments or free-water contaminants). Therefore, receptors for aquatic plants will be "generic" and befitting the protocol for assessment methods contemporary with screening methodologies accepted by the EPA and NDEP. Methodological protocol for assessment of aquatic conditions for the screening of contaminants in either free-water or sediment are outlined in detail in Section 4.4.

Note that screening-level aquatic benchmark comparisons in no way supercede Chapter 445A (Water Controls) of the State of Nevada Legislative Statutes. Adherence to all applicable statutes for water quality are assumed to form a basis of lawful compliance that falls outside of the purview of this SLERA document.

- ***Invertebrates***

Invertebrates that occupy or may potentially occupy freshwater environments in the area of the BMI Complex (including the Las Vegas Wash) are surprisingly diverse (USGI 2005). Similar to aquatic plants, however, methods for assessing screening-level ecological risk are broadly addressed in aquatic benchmark comparisons for free-water and sediment. The aquatic invertebrate receptor will be a "generic" aquatic invertebrate, and protocol for assessment methods will be contemporary with screening methodologies accepted by the EPA and NDEP. Methodological protocol for assessment of aquatic conditions for the screening of contaminants in either free-water or sediment are outlined in detail in Section 4.0.

- ***Vertebrates***

Truly aquatic vertebrate receptors at the BMI Complex are limited to those that may occupy the area surrounding the Kerr-McGee Seep. Fish in the Kerr-McGee Seep appear to be limited to a single species, mosquitofish *(Gambusia affinis)*. Amphibians that may occur in the Kerr-McGee Seep are likely limited to two species, the bullfrog *(Rana catesbeiana)* and the Woodhouse's toad *(Bufo woodhousii)*. Both of these species have obligate life stages that are purely aquatic. Similar to aquatic plants and invertebrates, however, methods for assessing screening-level ecological risk are broadly addressed in aquatic benchmark comparisons for free-water and sediment. Thus, the aquatic vertebrate receptors will be a "generic" fish (with a strong preference for toxicological studies specific to mosquitofish), and a "generic" freshwater amphibian (with strong preferences for toxicological studies specific to bullfrogs and/or Woodhouse's toads). Protocol for assessment methods will be contemporary with screening methodologies accepted by the EPA and NDEP for either free-water or sediment, and are outlined in detail in Section 4.4.

For certain groups of birds and mammals, as well as for the terrestrial phases of amphibians, special consideration must be given to insectivores where biomagnification of contaminants from aquatic media is of concern for the conditions of a food chain model. For screening-level risk assessment considerations, vertebrate receptors must be chosen for special concerns of biomagnification from (1) aquatic media to emergent invertebrates to vertebrate insectivores, and also for biomagnification from (2) aquatic media to fish (or amphibians) to vertebrate piscivores. These models are similar to those for terrestrial secondary consumers, and are considered part of the Wildlife Exposure Model (Section 4.3).

As with terrestrial vertebrate receptors, receptors for concerns of food chain modeling from contaminated aquatic media can be selected as representative species or an agglomerate representative of the taxonomic and functional group combination (as with Hampton et al. 1998 and the methods used for the INEEL). Full agreement between NDEP and BMI Complex

stakeholders must be made for receptor selection, for each taxonomic and functional group combination, in advance of performing the screening analysis.

In the case of selecting a representative species for the taxonomic and functional group combination, development of exposure parameters specific to the representative species will be fully documented and supported according to the methods of the screening-level analyses.

In the case of utilizing an approach similar to Hampton et al. (1998) for receptor selection, the following protocol will be used for the development of exposure parameters and the attending calculation of hazard quotients and indices:

- Highest ingestion rate to body weight ratio for all receptors in feeding guild
- Chronic exposure duration set to 1.0 (year-round)
- All dietary consumption of insects/invertebrates from contaminated water source

Consideration of developmental and behavioral aspects of each species considered for receptor (and/or exposure factor parameter) selection will aid in completion of the selection process. If developmental or behavioral conditions for any given species require that it be included in the functional food web for both the aquatic and terrestrial realms, then a species may appear in more than a single food web. Similarly, if developmental or behavioral conditions for any given species require that it be included in more than a single functional feeding guild, then a species may appear in more than a single guild for the purposes of the screening analysis.

4.0 Screening Analysis

4.1 Introduction

The intent of the screening analysis is to consider media-specific data in a stepwise process that progressively eliminates COPCs from consideration as potential stressors to ecological receptors. At each step, COPCs that cannot be eliminated from consideration are passed along in the analysis until all screening criteria have exhausted the possibility of elimination; the remaining COPCs are then considered COPECs. Ultimately, all COPECs are considered in the uncertainty analysis phase for final processing and potential elimination. The uncertainty analysis may also be used as a tool for adding in COPCs that were eliminated from consideration due to other conditions, such as adequacy of data coverage or uncertainty of contaminant transport and fate concerns.

The screening analysis maintains as a base assumption that the problem formulation for the SLERA is complete and that assessment endpoints and receptors have been appropriately identified. To reiterate, the problem formulation (Section 2.0) includes data evaluation and characterization, ecological scoping, and completion of the CSM. The selection of screening-level assessment endpoints and receptors (Section 3.0) follows naturally as a transitional process from problem formulation to screening analyses.

The screening analysis is a stepwise process that should follow an ordered approach. Subsections that follow provide an ordered approach that will serve as a guide for a risk assessor to chart their way through the screening analysis. Analysis basics are first reviewed in order to elucidate key elements required in the screening analysis. The following list summarizes the subsections of the screening analysis by major heading, and forms a hierarchy of methods to follow in the screening analysis.

- Analysis Basics
  - Toxicity reference value identification and development
    - Ecological relevance of Toxicity Reference Values (TRVs)
    - Study design
    - Approaches to TRV development
  - Ecological screening levels
  - Hazard quotient and index calculations
- Terrestrial Wildlife Exposure Models
  - EPA soil screening levels (Eco-SSLs)
  - Bioaccumulation models for terrestrial plants and invertebrates
    - Terrestrial plant bioaccumulation model
    - Terrestrial invertebrate bioaccumulation model
  - Generalized ingestion and imbibition models
    - Generalized ingestion models for vertebrate receptors
    - Generalized imbibition models for vertebrate receptors
  - VOC inhalation models for vertebrate receptors
  - Biomagnification and trophic transfer models for aquatically-based contaminants
    - Sediments to vertebrate wildlife
    - Water to vertebrate wildlife
- Water and Sediment Benchmark Value Comparisons
  - Sediment quality benchmarks
  - Water quality benchmarks
- Radiological Contaminants

4.2 Analysis Basics

4.2.1 Toxicity Reference Value Development

Development of toxicity reference values (TRVs) is necessary in order to identify media-specific contaminant concentrations that will cause no measurable chronic detriment to biota in the no-build area of the BMI Complex or other areas that may be evaluated in the future. A review of TRV development is provided in order to illuminate the foundational quality of their application in the SLERA. Additionally, a discussion of criteria for development for TRVs is provided in order to aid the risk assessor and risk managers in an understanding of the primary sources of uncertainty that may arise due to TRV derivation.

A TRV is defined as “a dose for a receptor that is likely to be without appreciable risk of deleterious effects from chronic exposure” (Kester et al. 1998), and “a dose above which

ecologically relevant effects might occur to wildlife species following chronic dietary exposure and below which it is reasonably expected that such effects will not occur" (EPA 2005). Kester et al.'s (1998) TRV definition is, perhaps, more broadly inclusive than that of EPA (2005), as it additionally pertains to organisms that may be dosed by means other than dietary, but each clearly identify a TRV as a dose indicative of a threshold above which biotically detrimental effects may occur. Both EPA (2005) and Kester et al. (1998) make clear that the assessment of threshold effects for a particular toxin consists of a hazard evaluation and a dose-response assessment, although their specific language and exact methodologies differ. The hazard evaluation is intended to identify COPC-specific nature and severity of effects, while the dose-response evaluation is used to predict the chemical exposure levels that may result in effects. These aspects of TRV development will be elaborated upon in subsequent subsections.

A foundational understanding of TRV derivation is necessary to assist the risk assessor and site managers determine the representativeness of TRV derivation results for individual species or functional groups of organisms, and identify uncertainties of those results. Representativeness of TRVs is at the center of identifying uncertainties, particularly in light of site-specific conditions (Podolsky and Newell, 2002). In order to address representativeness and uncertainty in TRV derivation, it is imperative that the derivation of TRVs and associated measures be made explicit and transparent, and that the methods for derivation are applied uniformly and consistently for each broad classification of chemical constituents, including (1) inorganic contaminants, (2) organic contaminants, and (3) radionuclides, as well as for each receptor in a functional group.

The methods used to derive TRVs have significant influences on the results of a SLERA, since TRVs form the basis for the elimination of COPCs by hazard quotient calculations (Section 4.2.3). Differences or similarities in derivation of TRVs (by different organizations or individuals) are influenced by the data set from which they were derived, the type of endpoint(s) selected, the age or life stage categorization of a test organism, the use of biological and/or statistical significance in determining effect levels, the selection of dose rate parameters, and the determination of exposure duration and/or test period (Podolsky and Newell, 2002). Although approaches to TRV derivation are quantitatively defined, experimental differences, subjectivity in the interpretation of experimental results, chosen levels of statistical significance for threshold effects, and various numerical "penalties" of uncertainty ("uncertainty factors" or "adjustment factors"), may combine to create quantitative differences in final TRV derivation, depending upon approach (EPA 2005, Kester et al. 1998, Podolsky et al. 2001, Podolsky and Newell 2002). Consistency in the interpretation of experimental results and uniformity of applied uncertainty factors (UFs) is of central importance to the derivation of TRVs.

4.2.1.1 Ecological Relevance of TRVs

The ecological risk assessor must bear in mind that for this SLERA that receptors were selected on the basis of functional groups and may be representative species or exposure factors that collectively (and conservatively) represent traits of all organisms in the functional group (Sections 2.4.1 and 3.0). Thus, organisms in a functional group should share similar potential for contaminant exposure by means of media-specific exposure pathways, and should also share a similar biological response to various modalities of toxin exposure (Hampton et al. 1998).

These assumptions belie the development of TRVs and associated measures for all functional groups.

The aim of calculating a TRV is to quantify the dose of a COPC, by whatever media or exposure modality, at which there is no appreciable response from ecological receptors. Thus, the ultimate goal in TRV development is to generate a value that is adequately protective of receptors that reside in the ecological system of concern. In order to generate such a value the data set must include biotically-relevant studies (relevant taxa and exposure modalities) and the data must be of a nature that is robust enough to be scientifically defensible. There are multiple approaches to deriving TRVs, each with specific procedural requirements. However, there are some general considerations that are, to some degree, incorporated into all approaches to TRV derivation, and these are described below.

4.2.1.2 Study Design

The relevance of the toxicological study design to the ecological concerns of the risk assessment is measured (qualitatively) by the similarity of the test species to receptors, the exposure medium or route, the bioavailability of the chemical form under experimental conditions vs. those in the natural environment, the exposure duration and frequency in experimentation, the targeted endpoint (whole organism, organ, behavior, etc.), and the measured effect level. Relevance of the toxicological study naturally bears upon prediction of the risk of adverse health effects to organisms in the natural system under consideration, as a result of exposure to chemical stressors in the environment. The closer the match of a study design to the characteristics of target receptors in the ecosystem of concern, the higher the degree of certainty in the assessment results. We will summarize each of the aforementioned study design elements in paragraphs below.

- ***Test Species***

The phylogenetic relationship (degree of relatedness) of a test species to a receptor in the ecosystem of concern is important because closely related organisms are more likely to share trait that lead to similar responses to toxins in their environment. Major differences in physiology, as are found between distantly related taxa, can lead to differences in metabolism and bodily distribution of any given chemical. Differential physiological handling and anatomical sequestration of a chemical and its forms (or states) can lead to significant differences in the toxicological properties of the chemical, and therefore the effects on a given organism. In general, the test species used should be in the same taxonomic class (e.g., mammalian or avian) as the receptor of concern in order to ensure that gross physiological traits are matched. Wide physiological differences may occur between species of organisms in the same taxonomic class (e.g., mammals), therefore, closer phylogenetic matches (e.g., as among congeners, or even those of the same taxonomic family) are preferred over those that are more distant. Unfortunately, close matches of test organisms with those in the natural system of concern are often unavailable. Increasing levels of uncertainty due to differences in test and target organisms should be thoroughly discussed in the Uncertainty Analysis (Section 5.0).

- ***Exposure Medium and Route***

The medium in which a chemical is harbored can have significant influence on the bioavailability of the chemical to organisms that interact with that medium. Conditions of solubility, pH, particulate size, foodstuffs, and all attendant conditions of chemistry can profoundly influence the absorption of a chemical across gut linings, mucous membranes, skin, or any other active or passive organ (including those pertaining to plants). The bioavailability of a chemical can lead to differences in absorption (dose), and therefore the expressed toxicity. For considerations of TRV development, the exposure modality of test (experimental) organisms should be the same as those determined for wild organisms that occupy the site under consideration. For example, if the primary exposure modality is incidental dietary ingestion of soil, then toxicological studies that mimic this modality are preferred for the development of a TRV. For TRVs that have been established in literature, those that were established on a basis that best resembles that of the exposure modalities of organisms on the site of concern are preferred over any others. However, availability of toxicological studies that closely match exposure modalities of natural organisms do not always allow for such specificity. Increasing levels of uncertainty due to differences exposure modalities of test and wild organisms should be thoroughly discussed in the Uncertainty Analysis (Section 5.0).

- ***Test Chemical***

The chemical form of the test toxin, compared to that found in contaminated media of the natural system of concern, influences bioavailability and expressed toxicity, as previously discussed. For example, some metal salts (e.g., sulfates) are less soluble than others (e.g. nitrates), both in the gut of many animals, as well as in most soils. For organic chemicals, isomeric form may strongly influence sorption and toxicity of a given chemical species. Thus, in order to provide more reliable results for TRV development, preference is for the use of chemical species in toxicological experimentation that match the form of those found in media of the site of concern. Increasing levels of uncertainty due to differences of chemical form utilized in tests vs. those found in site media should be thoroughly discussed in the Uncertainty Analysis (Section 5.0).

- ***Exposure Duration & Frequency***

The match of the exposure duration is important because of the specificity of the amount of time for a particular chemical to cause an adverse effect in an organism. If time allotted in experimental duration is insufficient, toxicological effects may not have time to become manifest, or may manifest incompletely or differently than under appropriate exposure duration. Typically, studies of less than 90 days are considered acute, 90-180 days are considered subchronic, and greater than 180 days are considered chronic. In general, chronic studies are the most ecologically relevant since organisms in the natural environment are repeatedly exposed to toxins (perhaps over a lifetime) due to habitual or obligate interaction with the medium of concern. Moreover, chronic effects are more difficult to detect in organisms under natural conditions, due to the insidious nature of chronic toxicological effects.

The frequency of exposure is also important because of metabolism and the ability of the body to respond and recover to a chemical assault when exposure is not continuous. In general,

continuous exposures under experimental conditions best characterize persistent exposure to toxins in the natural environment. Persistent exposure provides a worst-case scenario for chronic exposure conditions to toxins, and are therefore preferred for development of TRVs in order to not underestimate potentially adverse dose thresholds of toxicological effects.

Studies of acute or subchronic exposure duration increase the level of uncertainty that arises from extrapolating from experimental conditions to those found for organisms in the wild that may be encountering a lesser but chronic dose from a given chemical constituent. Likewise, toxicological studies that do not implement a steady and persistent dose of a given chemical constituent to receptors introduce a greater level of uncertainty in the derivation of TRVs. These uncertainties should be discussed thoroughly in the Uncertainty Analysis (Section 5.0).

- ***Endpoint***

Experimental measures and resultant effects on test organisms should be representative of those relevant to the biota occupying the site of concern. The endpoint of an experiment is the targeted effect, and is quantified by its measure of expression. Effects that are commonly measured in toxicological experimentation are mortality, morbidity (including impaired growth or development, impaired organ states, neurological impairment, and hematological effects), reproductive impairment or success, and non-adaptive behaviors that may impede survival or reproduction. Impairments that cause reproductive decline or mortality are considered to have the potential to produce population-level effects. However, morbid effects and those that result in non-adaptive behaviors may also have strong applicability to the development of TRVs and require professional judgment for their employment. Uncertainties that arise from the manifestation of effects, selected endpoints, and the measurement of effects should be thoroughly discussed in the Uncertainty Analysis (Section 5.0).

- ***Effect Level***

The goal of TRV development is to generate a "no observed adverse effects level" (NOAEL) for each COPC. The EPA (1997) defines a NOAEL as "the highest level of a stressor evaluated in a toxicity test or biological field survey that causes no statistically significant difference in effect compared with controls or a reference site." The calculation of a NOAEL in toxicological experimentation requires the evaluation of multiple effects levels where a "no effects" level (or concentration) is quantitatively identified to form a bounding condition. This level of quantitation is not always targeted in toxicological experiments, and many experiments may target a single dose for measurable effects or may only attempt to establish a lowest observed adverse effects level (LOAEL) without simultaneously establishing a NOAEL. Extrapolation of a NOAEL from a LOAEL or a lethal dose (LD) value increases uncertainty, and any form of extrapolation to a NOAEL (vs. a bounding measurement) should be discussed in the Uncertainty Analysis (Section 5.0). When extrapolation is required from a LOAEL or LD to a NOAEL, UFs are typically applied by a well-defined formulaic relationship that ensures a reasonable margin of certainty that potential risk is not underestimated.

- ***Experimental Design and Documentation***

The design of toxicological experiments is critical to the proper analysis and interpretation of the results. An appropriate study design and sufficient documentation also lends a higher degree of certainty to the assessment results. Experimental methods that improve the potential for TRV development include the use of control groups, multiple exposure groups to establish bounding conditions, and the use of statistical tests that are appropriately applied for interpretation of data.

Naturally, a TRV for any chemical constituent and receptor is improved with the robustness of the data set from which it is derived. Data are improved with the number of unique studies and identified endpoints, the number of bounded effects measurements, and the number of test species relevant to a single receptor or functional group. Statistical identification of a distribution of effects further improves the ability of the risk assessor to identify threshold effects levels, thus improving the identification of a TRV.

Derivation of TRVs introduces uncertainty to the SLERA (as discussed). That uncertainty must be adequately offset in the final calculation of the TRV for each COPC and receptor or functional group, so that potential risk of adverse effects from a given dose is minimal. Uncertainty factors (UFs) are, in a sense, penalties to a final determination of a TRV based on experimental conditions. In particular, the magnitude of UFs may be substantially influenced by experimental parameters such as the test exposure duration, test period, measure of effects (e.g., no adverse effects level or "NOAEL," lowest adverse effects level or "LOAEL," lethal dose or "LD," and whether the study was chronic, subchronic, or acute, etc.), and critical life stage (e.g., developmental, reproductive) of test organisms. Uncertainty is also generated by the lack of taxonomic and trophic similarity of the test organism compared to the species of concern for the SLERA. A TRV based on the same primary toxicity study can vary by several orders of magnitude depending on the approach used to characterize experimental uncertainty. Consistency in the application of UFs to the derivation of TRVs is critical for reliable development of TRVs in ecological risk assessment.

4.2.1.3 Approaches to TRV Development

There are a number of acceptable approaches to TRV development, including: EPA Eco-SSLs (EPA 2005); methods of Kester et al. (1998) employed at the Idaho National Environmental Engineering Laboratory (INEEL); methods of Los Alamos National Laboratory (LANL 2004); the U.S. Army Wildlife Toxicity Assessment Program (USACHPPM 2000); Oak Ridge National Laboratory (Efroymson et al. 1997a and 1997b, Sample et al. 1996, Sample et al. 1998). Other methods exist and may be thoroughly explored.

The derivation of a TRV is generally a three step process that includes (1) a literature search, (2) review and selection of appropriate primary references, and (3) statistical derivation of the TRV. The inclusion or exclusion of particular studies, the handling of data, and the statistical methods (including the use of UFs) for TRV derivation can significantly affect derived values. The approach that is selected should be based on the needs of the project, should pertain directly to methods used to calculate potential risk to individual species (e.g., species of special concern),

representative species for functional groups, or composite analyses for functional groups, and should be well documented and transparent.

For individual species representative of a functional group, dose-response thresholds and TRVs may be derived from toxicological experiments performed on that species or species that are demonstrated as phylogenetically and toxicologically similar. If a composite receptor approach is taken to populating exposure parameters for a given functional group, then the TRV for the functional group is identified by a reasonable and statistically justified means from all members of the group (e.g., a geometric mean of all derived NOAEL values).

In the case of a species of special concern (species identified as sensitive, threatened, or endangered by local, county, state, or federal governmental agencies), TRV derivation must be considered for the individual species and should bear as much species-specific toxicological information as is available for each COPC. However, toxicological information on species of special concern may be substantially lacking. Thus, the risk assessor may be compelled to consider species that are closely related phylogenetically (e.g., congeners), and may also require original research.

For taxonomic groups and/or COPCs for which no criteria exist for screening analysis, including those for which no toxicological data or benchmark comparisons exist, a weight of evidence approach must be taken for any screening-level risk assessment conditions or parameters. Such considerations would include toxicological data from surrogate taxonomic groups and toxicological data for chemical analogs. Explicit professional judgment and justification must be documented for any and all weight of evidence approaches, and must be thoroughly discussed as part of the Uncertainty Analysis (Section 5.0).

Methods for final TRV calculation and/or selection are subject to final approval by NDEP in advance of proceeding with the screening analysis for the BMI Complex. Concurrence of TRV selection methods will require that development of TRVs for receptors will be thoroughly documented, explicitly and transparently exposited, UFs will be uniformly and consistently applied, and the final uncertainty in the analysis of each TRV be consistently evaluated. Each TRV chosen for screening analysis methods for the BMI Complex SLERA will be cited to appropriate methodologies and the database from which it was selected. All TRVs used for the BMI Complex SLERA will be disclosed, and full documentation of any TRV value selected may be required.

4.2.2 Ecological Screening Levels

Following the development of TRVs, ecological screening levels (ESLs) may be calculated. ESLs are media-specific concentrations of a COPC that presumably confer the TRV dose (of that COPC) to a given receptor. ESLs are calculated for each representative receptor or functional group, based on parameters derived for uptake, including bioaccumulation, ingestion, imbibition, inhalation, and biomagnification pathways (Section 4.3). Refer to the appropriate section for details regarding the calculation of ESLs.

4.2.3 Hazard Quotient and Index Calculations

The hazard quotient (HQ) is the ratio of the representative exposure concentration of a COPC in a given medium to a threshold effects concentration of the same constituent in the medium of concern (LANL 2004). This definition is the same as that of VanHorn et al. (1998). For a specified exposure medium, the HQ is calculated as:

$$HQ = \frac{\text{exposure concentration}}{\text{effects concentration}} \qquad \text{Equation 1}$$

Equation 1 may also be written as:

$$HQ = \frac{exposure\ rate}{TRV} \qquad \text{Equation 2}$$

where the *exposure rate* is the receptor-specific intake or uptake of a given COPC from the medium of concern, and the TRV is the receptor-specific toxicity reference value of a given COPC, given the modality of exposure. The rates are the same for both the denominator and numerator, thus the HQ is a non-dimensional index.

Given the relationship of Equations 1 and 2, as well as the definition of an ESL (see above), the HQ may also be written in the form of media-specific concentrations:

$$HQ = \frac{[\text{COPC}]_{i,j}}{\text{ESL}_{i,j}} \qquad \text{Equation 3}$$

where,

- $[\text{COPC}]_{i,j}$ is the concentration of COPC *i* (a specific chemical constituent) in medium *j* (soil, water, sediment, or air);
- $\text{ESL}_{i,j}$ is the COPC-specific ESL for the given medium.

In this case, the units of mass are the same for both the denominator and numerator, thus the HQ is a non-dimensional index (as with Equation 2).

The intent of the HQ is evident by inspection of Equation 3, that it is an index used to determine if the concentration of a given COPC exceeds that of the ESL for a particular receptor and for a particular medium. Clearly, if the index is <1, then the likely risk posed by the COPC of concern is minimal, when considered in isolation of other COPCs. This does not mean that *no* risk exists, rather that the potential risk posed by the COPC is likely less than that posed by the calculated TRV for the organism or organismal group (representative species, functional group) for the medium of contact. Given this functional relationship, and for a COPC considered the sole stressor in an organism's environment, that COPC may be dismissed from further screening if the HQ is <1. Conversely, if the HQ is ≥1, then the COPC must be retained for further ecological risk analysis as a COPEC. The HQ is simply a pass/fail criterion.

More complex conditions arise, however, if there is more than one COPC in a given medium or in multiple media, as is nearly always the case in screening analyses. The summary effects index is the hazard index (HI), and is simply the sum of the HQs for all COPCs in the medium of concern. Thus, the mathematical expression for the HI is:

$$HI_j = \sum_i HQ_{i,j} \qquad \text{Equation 4}$$

where,

- $HQ_{i,j}$ is the receptor-specific HQ for each COPC in medium *j* (either soil, water, sediment, or air);
- HI_j is the receptor-specific HI for medium *j*.

If a receptor is exposed to more than one medium for the same contaminant, then the multimedia HI is the sum of HIs across all media of concern (LANL 2004).

If the HI is ≥1, a judgment must be made as to how much "weight" an individual COPC must bear in a summary effect for any or all of the COPCs to be carried forward from screening as COPECs. In most screening methodologies, all COPCs in a given medium (or in multiple media if multimedia exposure conditions prevail) are considered COPECs if the HI is ≥1. VanHorn et al. (1998) mentions an alternative method where those COPCs that contribute $\geq {}^{1}/n$, where *n* is equal to the number of COPCs, to the HI are considered COPECs. Alternatively, LANL (2004) considers those COPCs contributing ≥0.3 to the HI for a given receptor are identified as COPECs for further ecological risk analysis. Agreement should be sought in advance by BMI Complex stakeholders with NDEP for criteria for HI evaluation that differ from the most widely accepted criteria that all COPCs in a given medium (or in multiple media if multimedia exposure conditions prevail) are considered COPECs if the HI is ≥1.

A significant point of uncertainty in the HI criterion for COPC inclusion or dismissal (as a COPEC), is that synergistic effects of multiple chemical exposures by one or more pathways, is in most cases not predictable. Uncertainties that weigh for or against the potential for a synergistic effect should be discussed in the Uncertainty Analysis (Section 5.0), as should be the potential for unidentified synergistic effects.

We emphasize that the HQ/HI analysis for the SLERA is a pass/fail criterion based on the relationship of potential effects from one or more pathways of exposure of receptors (or receptor groups) to one or more contaminants. This relationship is not probabilistic and therefore is not a calculation of risk, rather it is a method for minimizing the potential of negative effects from contamination that may bear upon receptors. It is therefore imperative that the HQ/HI *not* be interpreted in the context of the magnitude of risk imposed by any single contaminant on individual organisms or populations thereof.

4.3 Terrestrial Wildlife Exposure Models

In this section, we consider several methodologies that provide coverage of screening approaches acceptable for terrestrial receptors identified to occur or potentially occur on or near the BMI Complex and affected areas. The order that each of the methodologies are presented in represents a reasonable stepwise approach to the screening analysis.

We begin with a brief introduction and review of the EPA's Eco-SSL (EPA 2005) methods. Eco-SSL methods are used to derive risk-based soil screening levels that are broadly applicable to the evaluation of frequently encountered COPCs at hazardous waste site across the United States. These methods are considered central to ecological screening analyses that will be conducted for the BMI Complex and the affected areas, and are preferred for the receptor groups targeted by the Eco-SSL methods (see below).

We follow the introduction of Eco-SSL methods with generalized bioaccumulation and terrestrial wildlife exposure models that consider various non-dietary and dietary pathways for exposure of COPCs to wildlife (plants, invertebrates, and vertebrates). These models are intended to supplement the Eco-SSL screening methods for determining if terrestrial wildlife receptors may be at risk for receiving COPC doses exceeding their respective TRVs. Methods presented in the general models are not intended to replace the EPA's Eco-SSL methods, rather they are presented to provide basic modeling approaches to HQ, HI, and attending calculations that may be utilized when criteria for developing Eco-SSLs according to EPA Guidelines (EPA 2003a and Attachments, EPA 2005) cannot be met or are otherwise deemed insufficient for screening with respect to receptors identified for the BMI Complex and affected areas.

A first step in the generalized terrestrial wildlife screening process, is the evaluation of bioaccumulation by soil-dwelling plants and invertebrates, whose primary modality of COPC exposure is by direct tissue contact with soilborne contaminants. Simple models for plant and invertebrate bioaccumulation of soilborne contaminants are presented. If exposure to COPCs for terrestrial plants and invertebrates is more complex than methods and models address in this SLERA, (e.g., if site-specific scoping indicated that foliar uptake of airborne COPCs may be a primary exposure route for a contaminant), then this must be addressed in the Uncertainty Analysis (Section 5.0), and by further analysis beyond screening.

The generalized vertebrate wildlife exposure model presented involves uptake of contaminants from ingestion of contaminated soils and foodstuffs, imbibition of water, and inhalation exposure to volatile organic compounds (VOCs) for terrestrial vertebrates. Potential vertebrate receptors include mammals, birds, reptiles and terrestrial-phase amphibians. This model is successively broken down into its constituent parts so that soil, water, and air intake of contaminants may be considered independently, as may be necessitated for a particular receptor (or functional group) under specific exposure conditions. Simplification of the general model by constituent parts also allows a clearer understanding of the relative contributions of each exposure modality to the calculation of an HQ or HI for any receptor group. Multiple exposure pathways for any receptor (or functional group) necessitate the calculation of a multimedia HI.

A specialized model is presented for inhalation of volatile organic compounds (VOCs) as part of the generalized vertebrate wildlife exposure model. This model is applicable for vertebrate wildlife exposure to VOCs in a confined space, particularly underground burrows that harbor specialized organismal groups (fossorial organisms). The inhalation model naturally considers the flux rate of VOCs into an idealized burrow space.

Specialized biomagnification models are developed for potential foodchain transfer of COPCs from the aquatic realms to vertebrate receptors in terrestrial realms. These specialized waterborne and sediment-borne contaminant bioconcentration models involve assessment of direct bioaccumulation of COPCs from water and sediment to emergent insects, followed by food chain transfer to vertebrate insectivores. These models omit the incidental ingestion of soils in the diet of insectivores, which are idealized to consume only emergent insects. This omission is justified by the fact that the Eco-SSL methods and the generalized vertebrate wildlife exposure model include the assessment of contaminated foodstuffs and the incidental ingestion of contaminated soils. Thus, the intent of the bioconcentration models is to focus solely on the potential for foodchain transfer.

All COPECs identified by way of the Eco-SSL or generalized bioaccumulation and terrestrial wildlife exposure models should be considered for attending uncertainties and interpretations of risk (Sections 5.0 and 6.0) before being passed to the next level of ecological risk assessment. More complex and focused uptake models that address COPECs identified as an outcome of the SLERA may be required for receptors that occur or potentially occur on the BMI Complex and vicinity.

4.3.1 EPA Soil Screening Levels (Eco-SSLs)

Development of Eco-SSLs is found in EPA's *Guidance for Developing Soil Screening Levels* (EPA 2005), including the *Update of Ecological Soil Screening Level (Eco-SSL) Guidance and Contaminant Specific Documents* (EPA 2005), and EPA (2003a) and Attachments (see also http://mountain.epa.gov/ecotox//ecossl/SOPs.htm for Guidance and Attachments).

Eco-SSLs were developed to address the need for standard methods to be applied in the derivation of risk-based soil screening levels for contaminants commonly encountered at hazardous waste sites. Although Eco-SSLs were initially intended to provide sufficient protection for plants, invertebrates, soil microbial communities, amphibians, reptiles, mammals, and birds, the EPA concluded that there was insufficient toxicological information to support the development of Eco-SSLs for soil microbes, amphibians and reptiles. The generalized terrestrial wildlife exposure model will be necessary to address ecological screening for amphibians (terrestrial phase) and reptiles, but we generally caution that toxicological information may be insufficient for these organisms to support the development of TRVs and Eco-SSLs for many (if not most) chemical constituents. As for soil microbes, if concern arises during the ecological risk assessment process for soilborne microbial processes at (or around) the BMI Complex, these matters will necessarily be considered in a baseline analysis since all screening methodologies are insufficient for such analysis.

Eco-SSLs have been fully developed for 12 inorganic constituents, with 5 more pending (listed below). Eco-SSLs have been developed for only one organic constituent, with 5 more pending (listed below). Methods for the development of Eco-SSLs are fully described in the documents cited. Eco-SSLs are considered highly protective of terrestrial wildlife in the United States at large, and in each and every case where Eco-SSL screening is available, constituents in soil will be screened against the Eco-SSL values for the target receptor groups (plants, invertebrates, mammalian herbivores, insectivores, and carnivores, and avian granivores, insectivores, and carnivores). If a soilborne contaminant has a concentration less than the Eco-SSL for a receptor group, then that contaminant may be eliminated as a COPC for that receptor group. If a soilborne contaminant has a concentration greater than the Eco-SSL for a receptor group, then that contaminant will be passed on in the screening analysis and may be considered under the generalized terrestrial wildlife exposure model for each of the identified functional groups for the BMI Complex.

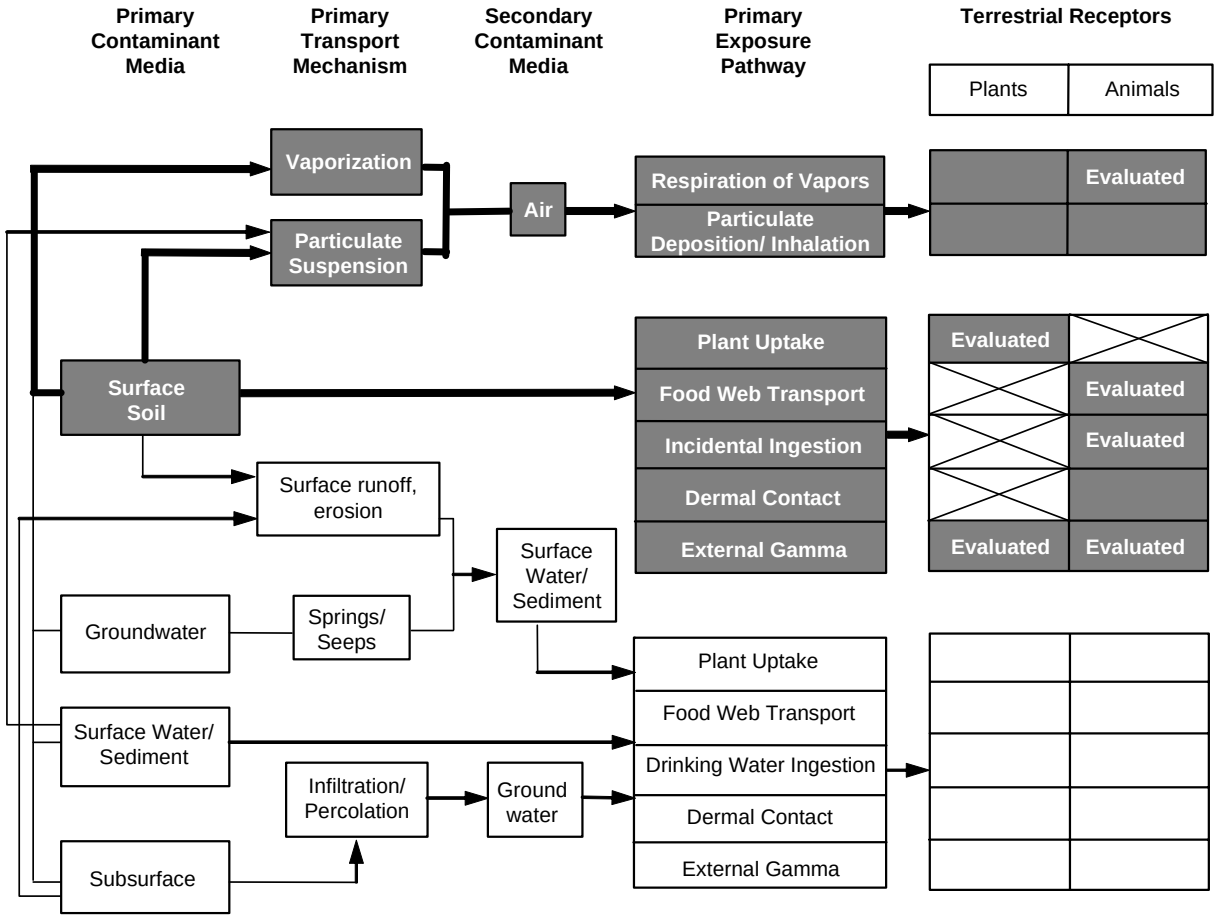
Pathways for soil-based contaminants to reach ecological receptors in the applicable context of evaluation by Eco-SSLs are illustrated in Figure 8. Figure 8 is applicable for all terrestrial receptors that may have direct contact with soilborne contaminants, and is therefore useful as an EPCEM for Eco-SSLs and the generalized terrestrial wildlife exposure model.

The following constituents currently have Eco-SSL guidance documents available or pending:

- Inorganic constituents with available Eco-SSLs
    - Aluminum
    - Antimony
    - Arsenic
    - Barium
    - Beryllium
    - Cadmium
    - Chromium
    - Cobalt
    - Copper
    - Iron
    - Lead
    - Vanadium
- Inorganic constituents with pending Eco-SSLs
    - Manganese
    - Nickel
    - Selenium
    - Silver
    - Zinc
- Organic constituents with available Eco-SSLs
    - Pentachlorophenol
- Organic constituents with pending Eco-SSLs
    - DDT and metabolites
    - Dieldrin
    - RDX

- Organic constituents with pending Eco-SSLs (continued)
    - TNT
    - Total PAHs

Figure 8. Ecological pathways conceptual exposure model for terrestrial receptors exposed to soil-based contaminants in the context of evaluation by Eco-SSLs.



Boxes marked in gray are potentially complete pathways for the specified exposure model; those labeled "Evaluated" are directly evaluated with standard screening tools identified in this document, while fields left gray but blank represent pathways for which no standard screening methodology exists. Fields marked with "X"s indicate incomplete pathways. Boxes with a blank field and no "X" indicate pathways irrelevant to specified exposure model.

The reader is referred to EPA (2005) and EPA (2003a) and Attachments for details regarding the methods of Eco-SSL derivation, requirements for employment of the methods, TRV development based on Eco-SSL methods, and the calculation of HQs. Uncertainties that may arise in the employment of Eco-SSL methods should be thoroughly discussed in the Uncertainty Analysis (Section 5.0).

4.3.2 Bioaccumulation Models for Terrestrial Plants and Invertebrates

Bioaccumulation is the process of an organism's uptake and sequestration of chemical constituents acquired from inorganic and organic media in its environment. Bioaccumulation is dependent on the presence and concentration of chemical constituents in the environment, and the physiological proclivity for an organism to sequester constituents and retain them over a measurable period of time. Retention of chemical constituents may occur in physiologically active tissues that act as a benefit or detriment for a given organism, according to concentration levels within the organism, metabolic needs, and biochemical pathways that may be beneficial or disruptive to an organism's growth, maintenance, reproduction, behavior, and ultimately, its survival and longevity. The threshold for negative action that any chemical may have in an organism's environment and bodily system is the threshold of accumulation that is of concern in the context of toxicology. Bioaccumulation does not imply that a chemical may reach a threshold of negative action since accumulation and loss by respiration and excretion (of various forms) are inherently rate-dependent processes. Chemicals that have greater staying time within an organism have a greater tendency to reach thresholds of negative action, and those for which an organism has lowest tolerance (as measured by concentration in specific biochemical processes) are of greatest concern as environmental toxins.

In this section, we consider the inherent processes of bioaccumulation by organisms in direct contact with contaminated substrates, namely terrestrial plants and invertebrates. Simple models for screening COPCs under these conditions are presented for use at and around the BMI Complex when Eso-SSLs are not available.

4.3.2.1 Bioaccumulation by Plants in Soil

Plants fix chemical constituents from the air and soil into tissues above and below the ground surface. Some constituents are exuded by way of transpiration or other processes and exit the plant via leaf stomata or other specialized organs. Other constituents are sequestered in roots or are transported to other organs and enter the physiochemical pathways of metabolism, thus becoming incorporated into proteins, nucleic acids, and other organic chemicals. Many organic and inorganic chemicals constitute the cellular componentry of plants, often being in fixed anatomical position as components of major plant structures, including roots, stems, leaves, bracts, reproductive structures, etc. Many of these structures are shed back into the soil, and eventually, the entire plant dies and returns to the soil. The accumulation of chemical constituents in plants is, therefore, a complex process. However, chemical constituents may reach toxic thresholds in plants and have been measured by a variety of toxicological studies through time.

The soil model for bioaccumulation by plants is a simple model:

$$C_{plant} = C_{soil} \cdot CR_{plant} \qquad \text{Equation 5}$$

where,

- C_{plant} is the concentration of a given COPC in plant tissue (mg COPC/kg plant, dry weight);
- C_{soil} is the concentration of a given COPC in soil, (mg COPC/kg soil, dry weight);
- CR_{plant} is the concentration ratio ([mg COPC/kg soil]/[mg COPC/kg plant], dry weight) for a given COPC.

For inorganic chemicals in plants, the CR_{plant} can be determined empirically by on-site measurements (preferred), or it may be determined from literature values, with Eco-SSLs as the preferred literature values. When Eco-SSLs are not available, other sources may be useful, e.g., Baes et al. (1984), Efroymson et al. (1997a), IT Corporation (1997). The CR_{plant} may be derived from a simple linear function or a non-linear function (EPA 2005) that is dependent on chemical concentration, physical and chemical properties of soil (including pH), biochemical activity and passivity of the plant, an the properties of ions, particularly the ionic radius and charge (Hooten and Myles 2006).

For organic chemicals, the CR_{plant} can be determined empirically by on-site measurements, or it may (less preferentially) be determined from the octanol-water partition coefficient and Equation 6 (LANL 2002).

$$CR_{plant} = 10^{1.31-0.385 \cdot log(Kow)} \qquad \text{Equation 6}$$

where, log(*Kow*) is the log-10 of the octanol-water coefficient. LANL (2002) makes the following observation regarding the *Kow*:

> "It is important to recognize that partition coefficients, such as the *Kow*, are in practice based on simple diffusion-equilibrium models and experimentation. For the plant transfer factor (Equation 6) this becomes important because the uptake of the organic chemical is determined solely from the interstitial water fraction of the soil. Conceptually speaking, therefore, any of the organic chemical that is adsorbed to inorganic and organic soil particulates is unavailable for plant uptake because it is not in the water fraction (interstitial water) of soil. This recognition makes it more difficult to estimate plant uptake because it is likely that the overall concentration of a chemical in soil is not representative of that which is available to plants."

A default value of 1.0 is assumed for the CR_{plant} when no values are otherwise available; this default is considered conservative by the EPA (2005).

The ESL for a given plant species is the soil-based TRV for a particular COPC for that species of plant. The HQ is calculated as follows:

$$HQ = \frac{C_{plant}}{TRV_{plant}} \qquad \text{Equation 7}$$

Frequently, the species of plants that are under consideration for a given site are not a match taxonomically or phylogenetically for plants considered in toxicological texts. Therefore, when deriving soil-based TRVs (ESLs) for plants, the derivation must be conservative, and the lowest experimentally-derived NOEC (no-effects concentration) for all plants considered for a given COPC must be the chosen TRV. This consideration tends to be protective of all plant species under consideration, which is typically diverse compared with experimental populations.

Uncertainties associated with the TRV (ESL) derivation for plants include the selection of the lowest TRV value for plants considered from those experimentally tested. In this context, uncertainty can be minimized by use of studies on plant species that are phylogenetically close to the species under consideration in the risk assessment. Additionally, uncertainty arises from differences in environmental conditions between those plants experimentally tested and those found in site conditions of the BMI Complex and vicinity. These conditions are myriad, but involve climate, soil structure and chemistry (including pH), plant and toxin chemistry, as well as a plethora of site-specific biotic conditions. Uncertainties regarding TRV derivation for plants should be discussed fully in the uncertainty analysis.

4.3.2.2 Bioaccumulation by Invertebrates in Soil

A complex array of invertebrates reside in desert soils and have been extensively studied and documented in the Mojave Desert (Rundel and Gibson 1996). Invertebrates that reside in soils tend to accumulate chemicals from soil by direct absorption across semi-permeable membranes as well as by soil ingestion (Markwiese et al., 2000, Markwiese et al. 2001). Ingestion of soils by invertebrates may complicate calculation of the concentration ratio for invertebrates (CR_{invert}) in the same manner that active transport of certain chemicals by plant roots may be complicating for CR_{plant}. Thus, the processes of chemical sequestration, transpiration and excretion are complicated for invertebrates, as are all processes of biotic uptake of contaminants. However, the soil contaminant uptake model for invertebrates is presented as a simple bioaccumulation model:

$$C_{invert} = C_{soil} \cdot CR_{invert} \qquad \text{Equation 8}$$

where,

- C_{invert} is the concentration of a given COPC in invertebrate tissue (mg COPC/kg invertebrate, dry weight);
- C_{soil} is the concentration of a given COPC in soil, (mg COPC/kg soil, dry weight);
- CR_{invert} is the concentration ratio ([mg COPC/kg soil]/[mg COPC/kg invertebrate], dry weight) for a given COPC.

For inorganic chemicals in invertebrates, the CR_{invert} can be determined empirically by on-site measurements (preferred), or it may be determined from literature values, with Eco-SSLs as the

preferred literature values. When Eco-SSLs are not available, other sources may be useful, e.g., IT Corporation (1997). The CR_{invert} may be derived from a simple linear function or a non-linear function (EPA 2005) that is dependent on chemical concentration, physical and chemical properties of soil (including pH), and biochemical activity.

For organic chemicals, the CR_{invert} can be determined empirically by on-site measurements, or it may (less preferentially) be determined by calculation. When calculated following primary sources, full documentation and review of the sources must be made available, otherwise, the calculation of the CR_{invert} will follow that of LANL (2002), after Connell and Markwell (1990) for an earthworm (see Equation 9).

$$CR_{invert} = \frac{L \cdot Kow^{y}}{c \cdot f_{oc}} \qquad \text{Equation 9}$$

where,

- L is the lipid fraction of the organism;
- c is the proportionality constant set equal to 0.66, following Connell and Markwell (1990);
- f_{oc} is the fraction of organic matter in soil;
- Kow is the octanol-water partition coefficient;
- y is the "nonlinearity constant" set equal to 0.05, following Connell and Markwell (1990), which can be thought of as the difference in relative affinity that an organic chemical has for soil organic matter and worm lipids (LANL 2002).

For more information and discussion of the CR_{invert}, consult LANL (2002). A default value of 1.0 is assumed for the CR_{invert} when no values are otherwise available; this default is considered conservative by the EPA (2005).

The ESL for a given invertebrate species is the soil-based TRV for a particular COPC for that species of invertebrate.

Frequently, the species of invertebrates that are under consideration for a given site are not a match taxonomically or phylogenetically for invertebrates considered in toxicological texts. (Indeed, many tests on the toxicological properties of soilborne contaminants have been conducted on earthworms, which do not occur in Mojave Desert alluvial soils.) Therefore, when deriving soil-based TRVs (ESLs) for invertebrates, the derivation must be conservative, and the lowest experimentally-derived NOEC (no-effects concentration) for all invertebrates considered for a given COPC must be the chosen TRV. This consideration tends to be protective of all invertebrate species under consideration, which is typically diverse compared with experimental populations.

Uncertainties associated with the TRV (ESL) derivation for invertebrates include the selection of the lowest TRV value for invertebrates considered from those experimentally tested. In this context, uncertainty can be minimized by use of studies on invertebrate species that are phylogenetically close to the species under consideration in the risk assessment. Additionally,

uncertainty arises from differences in environmental conditions between those invertebrates experimentally tested and those found in site conditions of the BMI Complex and vicinity. These conditions are myriad, but involve climate, soil, chemistry, pH, and a plethora of potential biotic conditions. Uncertainties regarding TRV derivation for invertebrates should be discussed fully in the uncertainty analysis.

4.3.3 Generalized Vertebrate Wildlife Exposure Model

The generalized vertebrate wildlife exposure model addresses exposure of wildlife to COPCs from ingestion of contaminated soil and foodstuffs, imbibition of contaminated drinking water, incidental dermal contact, and respiration of ambient air. This general model is presented as Equation 10 and is based on EPA's general wildlife exposure models (EPA 1993).

$$E_{total} = E_{oral} + E_{dermal} + E_{respiration} \qquad \text{Equation 10}$$

where,

- E_{total} is total exposure to a COPC (units are mass units, not weight, and are reported as mg COPC/kg medium/day);
- E_{oral} is oral exposure (foodstuffs plus ingestion of contaminated soil and water);
- E_{dermal} is dermal exposure (to media-born constituents: air, soil, water);
- $E_{respiration}$ is exposure through respiration or inhalation of ambient air.

For terrestrial vertebrates inhabiting the soil surface, it is assumed that most contaminant exposure to non-radiological chemicals is through the oral exposure pathway (Sample et al. 1997, EPA 2005), either by ingestion or imbibition. These two models are fully developed in this subsection for the calculation of HQs and HIs.

Dermal exposure to wildlife is typically mitigated by fur, feathers or scales covering the bodies of most vertebrates. Indeed, analysis has shown that dermal pathways contribute a small fraction of the dose obtained orally (EPA 2005), supporting guidance that indicates the ingestion route is most important to terrestrial animals (EPA 1997). Importantly, the incidental consumption of soil during grooming is included as a dietary component in the direct soil ingestion estimates. Thus, the dermal contact pathway is not typically assessed quantitatively in screening-level ecological risk assessments. If, during ecological screening, the dermal pathway is assessed as insignificant (in scoping, the CSM, or the uncertainty analysis), then it is dropped from further ecological risk consideration. If, however, the dermal pathway is considered potentially significant for a particular receptor group, the COPCs for this modality of exposure must be passed to further ecological risk analyses as COPECs.

Inhalation of contaminated air may contribute significantly to VOC exposure for species occupying burrows for a significant fraction of the time. Therefore, TRVs must be calculated for inhalation exposure for VOCs for burrowing animals, including mammals, birds (burrowing owl, if present), and reptiles. Information regarding the toxicological nature of VOCs for the various organismal groups to be assessed may be obtained empirically by on-site measurements

(preferred), or it may be determined from literature values, if available. Sources of information in support of TRV development of VOCs is uncommon in literature.

Inhalation of COPCs from fugitive dust and from vapors emitted at the soil surface are typically considered insignificant compared to other modalities of media-specific exposure (LANL 2004). Thus, the inhalation pathway is not typically assessed quantitatively in screening-level ecological risk assessments for organisms that are not obligated to significant portions of time in underground burrows. If, during ecological screening, the surface (particularly fugitive dust) inhalation pathway is assessed as being insignificant (in scoping, the CSM, or the uncertainty analysis), then it is dropped from further ecological risk consideration. If, however, the surface inhalation pathway is considered potentially significant for a particular receptor group, the COPCs for this modality of exposure must be passed to further ecological risk analyses as COPECs.

In cases where dermal and respiration exposure pathways may be deemed significant, the models presented by Hope (1995) should be used to evaluate these pathways.

When dermal and inhalation exposure modalities are removed from Equation 10, the terrestrial wildlife exposure model for non-radionuclides simplifies to Equation 11:

$$E_{total} = E_{oral} \qquad \text{Equation 11}$$

The oral exposure model used for terrestrial wildlife is from the *Wildlife Exposure Factors Handbook* (EPA 1993, Chapter 4), and expands E_{oral} into its component parts. This model provides a basis for estimating oral exposure of an inorganic or organic chemical toxicant in soil, food, and water, routinely ingested and imbibed on a daily basis. The model is intended to account for incidental ingestion of soil and contaminated drinking water from activities such as preening. When expanded into its components for ingestion and imbibition, Equation 11 takes the form:

$$E_{oral} = C_{soil} \cdot I_{soil} \cdot AUF_{soil} + C_{food} \cdot I_{food} \cdot AUF_{food} + C_{water} \cdot I_{water} \cdot AUF_{water} \cdot (1/d_{water}) \qquad \text{Equation 12}$$

where,

$$I = {}^{kg\ [\text{medium}]}\!\big/_{kg\ [\text{Body Weight}]\cdot \text{day}^{-1}}\text{, dry weight.} \qquad \text{Equation 13}$$

Since mass conversions are roughly equal to weight under most conditions for applying the wildlife model, the following interpretations are convenient for the parameters of Equations 12 through 13:

- E_{oral} is the estimated oral daily dose for a COPC (mg COPC/kg body weight/day). This is a total ingestion of a given COPC in units of mass from all dietary modalities, reported as follows:
- C_{soil} is the concentration of a COPC in soil (mg COPC/kg soil, dry weight).

- I_{soil} is the daily soil ingestion rate (kg soil/kg body weight/day, dry weights). Soil ingestion is calculated from a fraction of the dietary intake of soil (EPA 1993, Chapter 4).
- AUF_{soil} is the area use factor that represents the fraction of soil ingested from a contaminated area vs. the total area utilized for foraging and other activities where soil may be ingested. This is a proportion of area to area, and therefore bears no units. This fraction is set to unity for the initial screening.
- C_{water} is the concentration of a COPC in water (mg COPC/L of water). Since one L of water has a mass of approximately one kg at sea level, this parameter's units may be rewritten as mg COPC/kg water/day.
- d_{water} is the density of water. (This parameter varies according to elevation, but for the sake of the SLERA, this is considered 1 kg/L.)
- I_{water} is the daily water ingestion rate (kg water/kg dry body weight/day).
- AUF_{water} is the fraction of water ingested from a contaminated area vs. the total dietary water. This is a proportion and therefore bears no units. This fraction is set to unity for the initial screening.
- C_{food} is the concentration of COPC in food (mg COPC/kg foodstuffs, dry weight).
- I_{food} is the daily dietary ingestion rate (kg food/kg body weight/day, dry weights).
- AUF_{food} is the fraction of the diet derived from a contaminated area vs. the total. This is a proportion and therefore bears no units. This fraction is set to unity for the initial screening.

As a protective assumption appropriate for the SLERA, the area use factors (AUFs) are set equal to unity to indicate the animal receives all its exposure from the contaminated site. This conservatism may be discussed and expanded upon in the uncertainty analysis. Direct quantitative employment of AUFs in the context of ecological risk analysis is, however, beyond the scope of the SLERA, particularly if species-specific considerations are necessary.

Setting the AUFs and *d,* the density of water, to unity, Equation 19 may be re-written in a simplified form that will be more useful for ESL calculations.

$$E_{oral} = C_{soil} \cdot I_{soil} + C_{food} \cdot I_{food} + C_{water} \cdot I_{water} \qquad \text{Equation 14}$$

where,

- C_{soil} is the COPC concentration in soil (mg COPC/kg soil, dry weight);
- I_{soil} is the incidental dietary ingestion rate of soil (kg soil/kg body weight/day, dry weight);
- C_{food} is the COPC concentration in food (mg COPC/kg foodstuffs, dry weight);
- I_{food} is the ingestion rate of foodstuffs (kg foodstuffs/kg body weight/day, dry weight);
- C_{water} is the COPC concentration in water (mg COPC/kg water, or mg/L);
- I_{water} is the total ingestion rate of water (kg water/kg body weight/day, or L/kg body weight/day)
- Note: All measures of ingestion (except water) to be on a dry-weight basis. 1 L of water is equal to 1 kg water at sea level. For purposes of screening-level ecological risk assessment, this is also true for most land-based measures.)

An additional conservative assumption is that all foodstuffs (including water) come from sites that are under consideration in the SLERA. Similar to the AUF, the pros and cons of this conservatism may be expanded upon in the uncertainty analysis. Explicit quantitative assumptions that measurable proportions of foodstuffs from respective media originate on a site under consideration is beyond the scope of the SLERA analyses. If only one dietary medium (e.g. water) is considered as a contaminated dietary source, then the others drop out of Equation 14; similarly, if only two dietary media (e.g., soil and food) are considered as contaminated dietary sources, then the appropriate term drops out of Equation 14. All dietary water in the wildlife model is considered gathered from free water sources; acquisition of metabolic water is not considered in the SLERA.

An implicit assumption of the generalized vertebrate wildlife exposure model is that the bioavailability of a COPC from the various environmental media is comparable to the bioavailability of the contaminant in toxicological experiments (see TRVs, Section 4.2.1). Since little information currently exists on bioavailability conversions, a bioavailability term is not included in the generalized vertebrate wildlife exposure model. If concerns for the bioavailability of a COPC arise, and site-specific adjustments to bioavailability are possible due to quantitative chemical analysis, then this information should be included in the SLERA. If bioavailability concerns are qualitative, then these should be discussed in the uncertainty analysis. Further analysis of a constituent's bioavailability and how it may affect the wildlife model are beyond the SLERA.

The availability of COPCs to wildlife due to site-wide distributional considerations of COPC concentrations should be discussed in the CSM. Uncertainties that arise regarding the availability of COPCs to particular receptor groups or species should be discussed in the uncertainty analysis for the effects of conveyance to wildlife according to the wildlife model. How the attributes of COPC concentrations and their distribution in the media of concern may affect the wildlife model, are considerations to be thoroughly discussed.

The generalized vertebrate wildlife exposure model requires all measures of ingestion (except water) to be on a dry-weight basis. Because the EPA presents most food ingestion rates on a wet weight basis, these dietary constituents must undergo wet-to-dry weight conversions (EPA 1993). Biotic parameters (I_{soil}, I_{food}, and I_{water}) required for calculations of the wildlife oral exposure model, are either (1) specific to the representative species of the functional group (geometric mean if adequate data exist, or the maximum reported by a documented source), or (2) the maximum ratio of I_{soil}, I_{food}, and I_{water} chosen from among all species included in the functional group. (Again, the intent of a functional group approach is to be protective of *all* organisms in the functional group, without having to consider particular species as surrogate for the entire group.) More realistic exposure information may be considered in the uncertainty analysis.

To repeat (from Sections 2.4.1 and 3.0), functional groups considered for the wildlife oral exposure model are:

- mammals:
  - herbivores (all plant diet)
  - omnivores (mixed plant, flesh, and invertebrate diet)
  - insectivore (all invertebrate diet)
  - carnivores (all flesh diet)
- birds:
  - herbivores (all plant diet)
  - omnivores (mixed plant, flesh, and invertebrate diet)
  - insectivore (all invertebrate diet)
  - carnivores (all flesh diet)
- reptiles
  - herbivores (all plant diet)
  - insectivore (all invertebrate diet)
  - carnivores (all flesh diet)
- amphibians (land phase)
  - insectivore (all invertebrate diet)

Again, incomplete information for any given functional group may require the identification of surrogate species as representatives for the group. The selection of surrogate species must be done in agreement with risk assessors, the NDEP and stakeholders for the BMI Complex and vicinity.

The generalized vertebrate wildlife exposure model can be broken down into constituent parts in order to calculate HQs (and ESLs) on the basis of medium, and in order to consider the practical possibility that TRVs will differ according to medium. Ultimately, this will allow calculation of an HI for each receptor across all media of exposure.

4.3.3.1 Generalized Ingestion Models for Vertebrate Receptors

Terrestrial wildlife exposure to potentially contaminated soils is depicted in Figure 9, the EPCEM for soils.

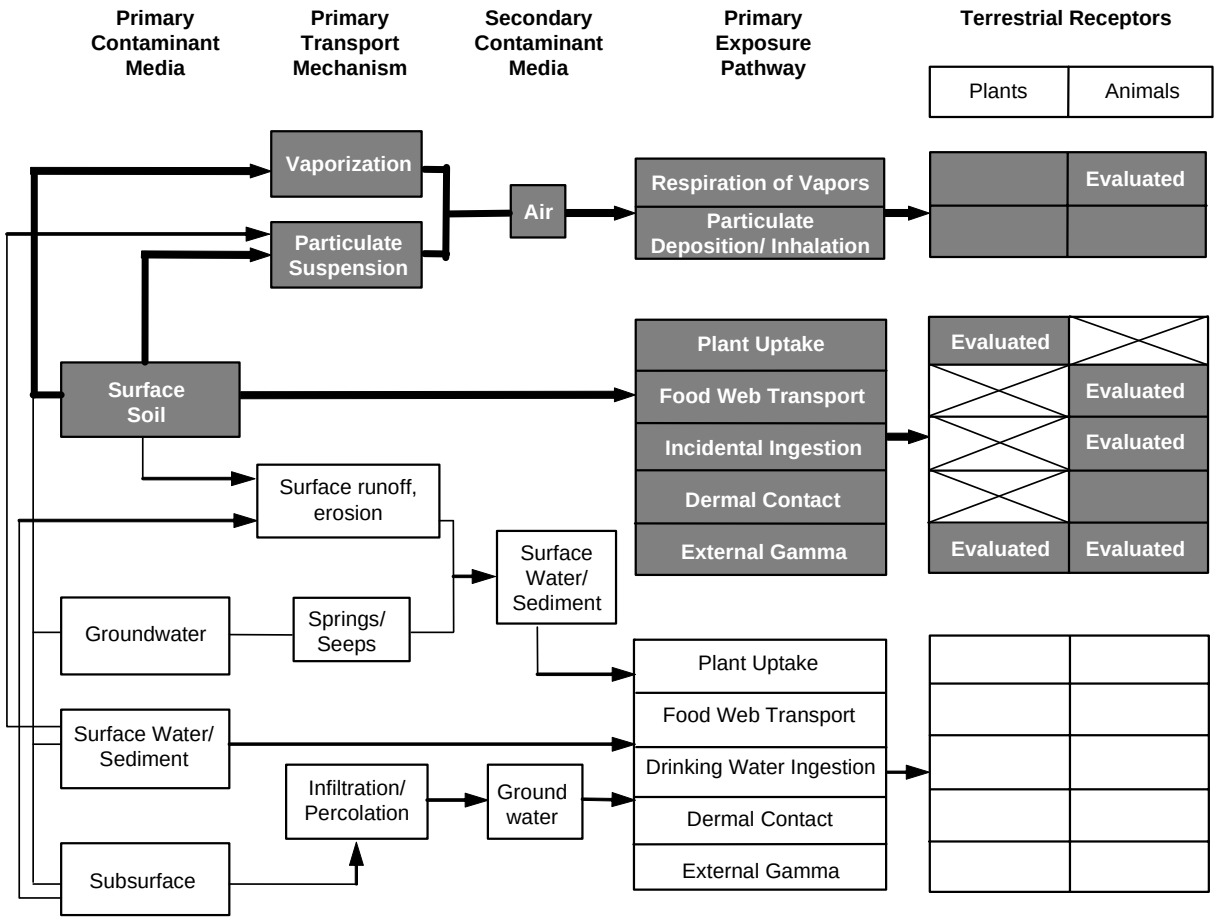
The conversion of soil concentration to an ingested dose requires the initial assumption that the intake of contaminated water is assumed to be zero. (This assumption will be convenient for calculating a soil ESL.) Next, the food intake is assumed to be entirely from the site of contamination, and is terrestrial in origin. Thus, the value of C_{food} can be related to an initial concentration in soil, since the pathway is from soil to plants or invertebrates, then to higher trophic levels:

$$C_{food} = C_{soil} \cdot CR_{food} \qquad \text{Equation 15}$$

where,

- C_{food} is the contaminant concentration in foodstuffs (mg contaminant/kg food, dry weight);
- C_{soil} is the contaminant concentration in soil (mg contaminant/kg soil, dry weight);
- CR_{food} is a concentration ratio (sometimes called a bioaccumulation factor [*BAF* of VanHorn et al. 1998] or transfer factor [*TF* of LANL 2004]) from soil to food ([mg COPC/kg soil]/[mg COPC/kg food], dry weight).

Figure 9. Ecological conceptual exposure pathway for terrestrial wildlife exposed to contaminants in soils.



Boxes marked in gray are potentially complete pathways for the specified exposure model; those labeled "Evaluated" are directly evaluated with standard screening tools identified in this document, while fields left gray but blank represent pathways for which no standard screening methodology exists. Fields marked with "X"s indicate incomplete pathways. Boxes with a blank field and no "X" indicate pathways irrelevant to specified exposure model.

Again, Equation 15 simply reflects the idea that all food intake is from the site of contamination, and is terrestrial in origin.

We may also consider a term for the total ingestion rate:

$$I_{total} = I_{soil} + I_{food} \qquad \text{Equation 16}$$

and,

$$I_{total} = I_{total}\left(f_{soil} + f_{food} \right) \qquad \text{Equation 17}$$

where,

- I_{total} is the total daily ingestion rate (kg food/kg body weight/day, dry weight);
- f_{soil} is the fraction of an organism's diet that is solely soil (fraction);
- f_{food} is the fraction of an organism's diet that is solely food (fraction).

Since the imbibition of water is set to zero, we can now rewrite Equation 14 in a series of equations with the introduction of a concentration ratio for food to indicate proportionate uptake of COPCs by a vertebrate wildlife receptor from soil:

$$\begin{aligned} Uptake_{oral} &= C_{soil} \cdot I_{soil} + C_{food} \cdot I_{food} \\ Uptake_{oral} &= I_{total}\left(C_{soil} \cdot f_{soil} + C_{soil} \cdot CR_{food} \cdot f_{food} \right) \\ Uptake_{oral} &= I_{total} \cdot C_{soil}\left(f_{soil} + CR_{food} \cdot f_{food} \right) \end{aligned} \qquad \text{Equation 18}$$

where,

- $Uptake_{oral}$ is the daily uptake of a contaminant by an organism via dietary soil plus food (mg COPC/kg body weight/day).

The proportion of food in the diet may be represented by a fraction, f_{food}, while the proportion of soil in the diet may be represented by the fraction f_{soil}. In order to apportion an organism's diet between the three basic food types (terrestrial plants, soil-dwelling invertebrates, and vertebrate flesh), f_{food} may be broken down as $f_{plant} + f_{invert} + f_{flesh}$. Thus, the contribution of contaminants from each of the food types may be written as:

$$f_{food} \cdot CR_{food} = f_{plant} \cdot CR_{plant} + f_{invert} \cdot CR_{invert} + f_{flesh} \cdot CR_{flesh} \qquad \text{Equation 19}$$

where,

- CR_{plant} is the concentration ratio from soil to plant (Section 4.3.2.1);
- f_{plant} is the fraction of the diet composed of plants;
- CR_{invert} is the concentration ratio from soil to soil-dwelling invertebrate (Section 4.3.2.2);
- f_{invert} is the fraction of the diet composed of plants;
- CR_{flesh} is the concentration ratio from vertebrate flesh (see *Definition of* CR_{flesh}, below);

- f_{flesh} is the fraction of the diet composed of plants.

Equations 15 through 19 allow us to understand the basic relationship between the oral dose of a constituent in soil to a terrestrial receptor. This dose calculation can be utilized directly in the calculation of the HQ for a COPC in soil *(HQ_{soil})* found in equation 20.

$$HQ_{i,j,soil} = \frac{I_{j,total} \cdot C_{i,soil} \left(f_{i,j,soil} + f_{i,j,plant} \cdot CR_{i,plant} + f_{i,j,invert} \cdot CR_{i,invert} + f_{i,j,flesh} \cdot CR_{i,flesh} \right)}{TRV_{i,j,ingestion}}$$

Equation 20

where,

- *i* is the index for COPC *i;*
- *j* is the index for receptor *j.*
- $TRV_{i,j,ingestion}$ is the TRV derived for COPC *i,* receptor *j,* by ingestion of contaminated soil and foodstuffs (mg COPC/kg body weight/day).
- All other variables are as previously defined.

In order to calculate a soil ESL for any given COPC, the HQ is set to unity and the equation is solved for the concentration of COPC *i* according to the parameters for each receptor *j*. This solution is performed as:

Equation 21

$$ESL_{i,j,soil} = C_{i,soil} = \frac{TRV_{i,j,ingestion}}{I_{j,total} \cdot \left(f_{i,j,soil} + f_{i,j,plant} \cdot CR_{i,plant} + f_{i,j,invert} \cdot CR_{i,invert} + f_{i,j,flesh} \cdot CR_{i,flesh} \right)},$$

for an HQ=1.

Note that the mathematical forms of the various concentration ratios (*CR* values) in Equations 20 and 21 require numerical solutions for their independent derivation according to the COPC. (See EPA [2005] for protocol on the calculation of the various model parameters, Section 4.2.1 for TRV derivation protocol, and Sections 4.3.2.1-4.3.2.2 for generalized CR_{plant} and CR_{invert} derivation. A generalized derivation of CR_{flesh} is found below.) Attendant variability in the calculation of *CR* values should be considered strongly in the calculation of HQ and ESL values, and *CR* values should be reasonably protective of each receptor and derived from well-documented methods. All experimental methods and results, and/or literature-based derivations of model parameters must be fully documented for each COPC and receptor, and must meet state-of-the-science criteria for their derivation (e.g., those outlined by EPA 2005).

Note also that the wildlife ESL model (Equation 21) shows the ESL as it relates proportionally to the TRV. It is clear that larger values of the TRV lead to larger ESL values, which indicates that the receptor may be more tolerant of a given COPC, and vice-versa. Organisms with a high intake to body weight ratio will have a lower ESL than those with a low intake to body weight ratio.

Uncertainties associated with soil ESLs involve interpretations of chemical toxicity and bioavailability (for any of the COPCs) reflected in the TRVs for oral ingestion of foodstuffs (excluding water) and the *CR*s (CR_{plant}, CR_{invert}, CR_{flesh}) between soil and food. Sources for toxicity and TRV or *CR* information may come from experimental circumstances and species that are substantially different than conditions/species being assessed in the SLERA. Additionally, the modalities and/or admixtures of toxin conveyance to experimental organisms are nearly always different than those considered in natural circumstances.

Uncertainties also arise regarding selection of parameter inputs to the ingestion calculations that represent a combination of idealized conditions. This is inevitable in the use of representative species or composite calculations for receptor groups. Within a selected trophic group, differences in diet (prey items) and other significant autecological parameters, are not typically accounted for when a representative species is selected or if maximum ingestion and idealized dietary fractions are utilized for calculations. For example, functional groups of carnivores may include species that have strictly mammalian or avian prey, which may confer differences in physiological pathways or levels of exposure due to levels of incidental soil ingestion, biochemical differences between prey items, as well as metabolic differences of their predators. Such concerns are well beyond the scope and considerations of the SLERA. However, general concerns for these and other uncertainties may be raised and evaluated in the uncertainty analysis.

- ***Definition of CR_{flesh}***

The concentration ratio for flesh, CR_{flesh}, may be derived from studies presented in primary literature regarding contaminant transfer from food (fodder or prey) to flesh (e.g., muscle tissue in beef cattle, Travis and Arms [1988]). The methods of Travis and Arms (1988) were reviewed by Burris et al. (2000) and later by LANL (2002). Equation 22 was derived in LANL (2002) for the transfer of inorganic chemicals from soil to flesh, given a specific COPC:

$$CR_{flesh} = \frac{CR_{beef} \cdot \left(I_{food} \cdot max\left[CR_{plant} \cdot \left\{1 - MC_{plant}\right\}, CR_{invert} \cdot \left\{1 - MC_{invert}\right\} \right] + I_{soil} \right)}{\left\{1 - MC_{flesh}\right\}} \qquad \text{Equation 22}$$

where,

- CR_{flesh} is the soil-to-flesh concentration ratio ([mg COPC/kg flesh]/[mg COPC/kg soil], dry weights);
- MC_{flesh} is the moisture content of flesh (0.68 for mammals such as mice, voles, rabbits, and birds, and passerines [EPA 1993, Table 4-1, p. 4-13]);
- CR_{beef} is the concentration ratio of all dietary constituents to beef ([mg COPC/kg beef fresh wt]/[mg COPC/day]);
- I_{food} is the fresh weight intake of food (the maximum of plants or invertebrates = 0.496 kg fresh wt/day, calculated as a composite average prey species);
- CR_{plant} is the soil-to-plant concentration ratio ([mg COPC/kg plant]/[mg COPC/kg soil], dry weights);
- MC_{plant} is the moisture content of plants (0.85 for leaves [EPA 1993, Table 4-2, p. 4-14]);

- CR_{invert} is the soil-to-invertebrate concentration ratio ([mg COPC/kg invertebrate]/[mg COPC/kg soil], dry weights);
- MC_{invert} is the moisture content of invertebrates (0.61 for beetles [EPA 1993, Table 4-1, p. 4-13]);
- I_{soil} is the ingestion of soil with food (0.0119 kg dry wt/day for composite average prey species).
- Note: Calculations of Equation 22 include corrections for fresh to dry weight.

For organic chemicals, other literature values for the CR_{flesh} may be available, or their uptake from soil may be calculated as in Equation 23. When borrowed or calculated from primary sources, full documentation and review of the sources must be made available. If modeled, the calculation of the CR_{flesh} will follow that of LANL (2002), after Burris et al. (2000):

$$CR_{beef} = CR_{flesh} = \text{fat} \cdot 10^{-1.79+0.414 \cdot log(Kow)} \qquad \text{Equation 23}$$

where,

- fat is the fat content of beef cattle, which is approximately 19% (LANL 2002).

For more information and discussion of the CR_{flesh}, consult LANL (2002). A default value of 1.0 is assumed for the CR_{flesh} when no values are otherwise available.

Methods of LANL (2002) are not mandated for this SLERA (and are secondary to methods of EPA 2005), rather the methods for final CR_{flesh} calculation and/or selection are subject to final approval by NDEP in advance of proceeding with the screening analysis for the BMI Complex and vicinity.

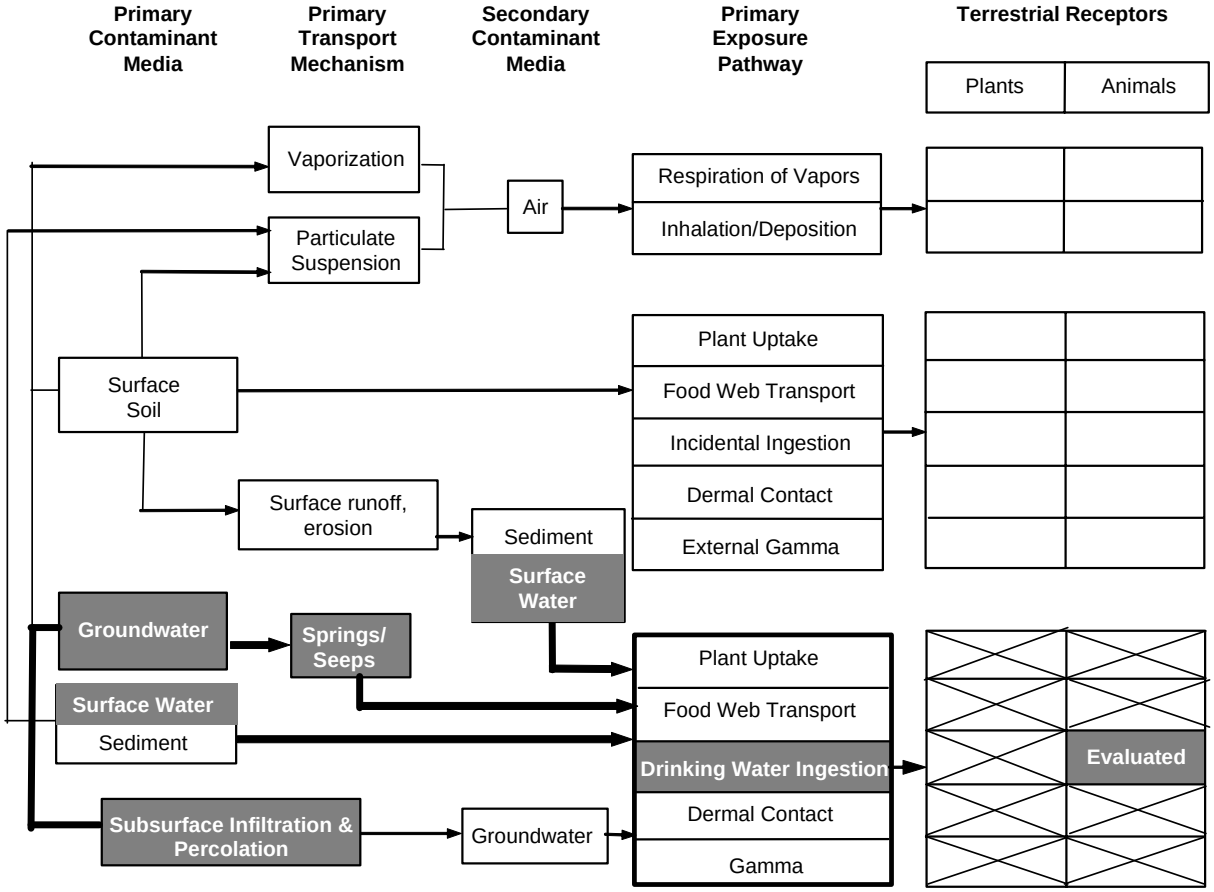
Concurrence of all *CR* (CR_{plant}, CR_{invert}, CR_{flesh}) selection methods will require that development of *CRs* for receptors will be thoroughly documented, explicitly and transparently exposited, and that the final uncertainty in the analysis of each COPC be consistently evaluated.

4.3.3.2 Generalized Imbibition Models for Vertebrate Receptors

Terrestrial wildlife exposure to potentially contaminated drinking water is depicted in Figure 10, the EPCEM for drinking water. As previously mentioned, screening-levels for wildlife in no way supercede Chapter 445A (Water Controls) of the State of Nevada Legislative Statutes. Adherence to all applicable statutes for water quality are assumed to form a basis of lawful compliance that falls outside of the purview of this SLERA document.

Evaluation of HQs for drinking water intake by vertebrate wildlife from contaminated sources requires the derivation of TRVs for COPCs in water and known rates of imbibition for each of the representative receptor species for each functional group. Again, incomplete information for any given functional group may require the identification of surrogate species as representatives for the group. The selection of surrogate species must be done in agreement with risk assessors, the NDEP and stakeholders for the BMI Complex. The basic assumption for drinking water intake is that water sources are not turbid, therefore toxicity is considered only for the dissolved fraction of a constituent in water. If contaminated water sources are turbid, then the sediment

Figure 10. Ecological conceptual exposure pathway for terrestrial wildlife exposed to contaminants in drinking water.



Boxes marked in gray are potentially complete pathways for the specified exposure model; those labeled "Evaluated" are directly evaluated with standard screening tools identified in this document, while fields left gray but blank represent pathways for which no standard screening methodology exists. Fields marked with "X"s indicate incomplete pathways. Boxes with a blank field and no "X" indicate pathways irrelevant to specified exposure model.

intake must be considered a dietary fraction of ingestion, which requires additional modeling beyond the SLERA. Hydrophobic constituents, constituents with low solubility and those that tend to be adsorbed to sediments (inorganic and organic fractions) typically do not form solutes, although this may be affected by pH and water hardness.

The generalized vertebrate wildlife exposure model (Equation 14) can be simplified for imbibition of a sole contaminated drinking water source with the elimination of ingestion of soil and foodstuffs (idealizing their contribution as equaling zero for the simplistic screening analysis of drinking water). This leaves the generalized models in the following form:

$$E_{water} = C_{water} \cdot I_{water} \qquad \text{Equation 24}$$

where,

- E_{water} is the daily exposure rate of contaminant to the organism via water imbibition (mg COPC/L water/day);
- I_{water} is the imbibition rate of water (L water/kg body weight/day),
- C_{water} is the concentration of a COPC in water (mg COPC/L water).

In order to calculate an HQ for contaminated water imbibition, a TRV is necessary that is specific to that medium and method of conveyance. TRVs for drinking water must be independently derived from TRVs for ingested foodstuffs since the physiological processes of absorption and biochemical pathways may differ substantially due to differences in bioavailability of a COPC as a solute in water versus a state bound or mixed in a food or soil matrix. Thus, toxicological information that supports the development of TRVs for wildlife drinking water must match the method of toxicant delivery (imbibition as a solute in drinking water) to test or observed organisms.

An HQ for the generalized imbibition model for vertebrate receptors is calculated as:

$$HQ_{i,j,water} = C_{i,water} \cdot I_{j,water} \qquad \text{Equation 25}$$

where,

- i is the index for COPC i;
- j is the index for receptor j.
- All other variables are as previously defined.

In order to calculate a soil ESL for any given COPC, the HQ is set to unity and the equation is solved for the concentration of COPC i according to the parameters for each receptor j. This solution is performed as:

$$ESL_{i,j,water} = C_{i,water} = \frac{TRV_{i,j,water}}{I_{j,water}} \qquad \text{Equation 26}$$

where,

- $TRV_{i,j,water}$ is the TRV derived for COPC i, receptor j, by imbibition of contaminated water (mg COPEC/kg body weight/day),

A default value of 1.0 is used for CR_{water} if no data are available for calculation of this parameter.

If no TRV_{water} is available, then a COPC in water must be considered in the uncertainty analysis. Potential risk of COPCs in water may be considered for wildlife with a weight-of-evidence approach that includes the use of water quality benchmarks (WQBs, Section 4.4.1). Further consideration for the potential effects of contaminated water on wildlife is provided in Section 4.3.4, Biomagnification and Trophic Transfer Models for Aquatically-Based Contaminants. One

must bear in mind, however, that the modalities of exposure for aquatic taxa and for wildlife through dietary conveyance of contaminants may differ considerably from direct imbibition of a COPC solute in water. Therefore, further analysis beyond the SLERA may be required when lines of evidence are inadequate for the general protection of wildlife from waterborne contaminants.

4.3.3.3 VOC Inhalation Models for Terrestrial Vertebrate Receptors

Quantitative evaluations of ecological risk do not typically include the inhalation pathway because contaminant exposure from ingestion is relatively more important for most chemicals. Simple fate and transport models indicate that vapor-phase contaminants are in relatively low concentrations in surface air and pose minimal threat to free-living organisms on or above the ground surface (Markwiese et al. 2003). Therefore, evaluation of surface air inhalation of VOCs is not typically warranted. However, for burrowing wildlife, especially fossorial animals that spend a large fraction of their lives belowground, burrow air exposure to volatile constituents, particularly volatile organic compounds (VOCs), is potentially a significant contaminant exposure pathway. Vapor-phase contaminants are not prone to bioaccumulation, so the pathways considered for burrow air are limited to inhalation of vapors and does not extend to dietary concerns for wildlife.

Organisms that may spend a considerable portion of time underground at or around the BMI Complex may include various mammals (e.g., antelope ground squirrel), birds (e.g., burrowing owl), and reptiles (e.g., western banded gecko). For such organisms that tunnel, den, nest, or search extensively for food below ground, consideration of VOC effects in the confines of burrow air is of concern. In order to address this concern, a wildlife model for a screening-level risk evaluation is a reasonable necessity.

A reasonable model for inhalation of VOCs in burrow air is presented for a mammalian receptor in LANL (2004). This model is presented below as a template for development of inhalation models that may be used for the projects related to the BMI Complex and vicinity. Similar modeling may be done for the burrowing owl and for a reptilian receptor. The final model(s) for inhalation exposure to VOCs in burrow air should be presented for fossorial mammals, burrowing owls, and reptiles.

Uncertainties for this form of model will likely arise from modeling inhalation rates, the derivation of TRVs for the specific receptors and/or receptor groups, and from taxon-specific uncertainties. Information on reptiles may be severely limited and a surrogate receptor may be required, which will contribute substantially to the uncertainty of HQ calculation for a reptile. All attending uncertainties should be discussed thoroughly in the uncertainty analysis.

- ***The LANL (2004) Model for Inhalation Exposure: Burrow Air***

The best estimate of burrow air is obtained by using soil pore-gas data collected from depths corresponding to those occupied by pocket gophers. Exposure parameters for the pocket gopher are provided in Table 1.

Table 1. Measures required for the burrow air VOC exposure model for the pocket gopher (*Thomomys bottae*)

Species	Parameter	Value	Units	Reference (page)	Notes
Pocket Gopher	Body weight	0.104	kg	Gonzales et al. (2000)	Laboratory-specific minimum measured field value
	Inhalation rate	0.089	m^3/day	EPA (1993) p. 3-12	Calculated from BW by Equation 3-20 in EPA (1993)

The gopher's inhalation rate is based on body weight, according to the allometric equation from Stahl (1967):

$$I_j = 0.5458 \cdot BW_j^{0.80} \qquad \text{Equation 27}$$

where,

- I_j is the average daily inhalation rate for mammalian receptor *j* (m^3 air volume/day),
- BW_j is the body weight for mammalian receptor *j* (kg body weight).

It is assumed that the gopher spends 100 percent of its time belowground, therefore, the exposure through air is described by:

$$E_{i,j,air} = C_{i,air} \cdot I_{j,air} \qquad \text{Equation 28}$$

where,

- $E_{i,j,air}$ is the estimated inhalation daily dose for COPC *i* and receptor *j* (mg COPC/kg body weight/day)
- $C_{i,air}$ is the concentration of COPC *i* in air inside the burrow (mg COPC/m^3 air volume)
- $I_{j,\,air}$ is the daily air inhalation rate scaled to the body mass of receptor *j* (m^3 air volume/kg body weight/day)

Therefore, the HQ can be expressed as shown in Equation 29:

$$HQ_{i,j,air} = \frac{C_{i,air} \cdot I_{j,air}}{TRV_{i,j,air}} \qquad \text{Equation 29}$$

where,

- $TRV_{i,j,air}$ is the TRV for COPC *i* and fossorial animal vertebrate receptor *j* (mg COPC/kg body weight/day).

In order to calculate a soil ESL for any given COPC, the HQ is set to unity and the equation is solved for the concentration of COPC *i* according to the parameters for each receptor *j*. This solution is performed as:

$$ESL_{i,j,air} = C_{i,air} = \frac{TRV_{i,j,air}}{I_{j,air}} \qquad \text{Equation 30}$$

- $ESL_{i,j,air}$ is the air ESL for COPC *i* and fossorial animal vertebrate receptor *j* (mg COPC/m^3 air volume)

4.3.4 Biomagnification and Trophic Transfer Models for Aquatically-Based Contaminants

Biomagnification is a process where chemical constituents with long staying times in biotic tissues are retained in low trophic-order organisms from dietary and non-dietary sources, then passed up a foodchain to consumers, thus becoming more concentrated with each trophic exchange. To expand upon this concept, these chemicals are most often accumulated in fatty (for animals) and starchy (for plants) tissues and tend to remain in metabolic reserve. These constituents are subsequently absorbed by consumers simultaneously with uptake of high-energy molecules in the digestive process, and are stored in the consumer's fatty, lymph, and organ tissues. Since consumers of contaminated prey are vulnerable to accumulations of quantities of a chemical in much higher concentrations than found in the physical environment, the repetition of the predatory (consumption) process tends to have the effect of magnifying chemicals with long staying times up the foodchain.

Processes of biomagnification are complex and may require site-specific or laboratory study of prey and predatory organisms beyond the level of the SLERA. Not all chemical constituents biomagnify, since many are excreted at a rate that equals or exceeds their sorption from the environment. Of particular concern for biomagnification are constituents such as organic mercury and various non-polar organic residues, such as DDT and is congeners, and PCBs (EPA 1995b).

Again, screening-levels for wildlife in no way supercede Chapter 445A (Water Controls) of the State of Nevada Legislative Statutes. Adherence to all applicable statutes for water quality are assumed to form a basis of lawful compliance that falls outside of the purview of this SLERA document.

4.3.4.1 Sediments to Vertebrate Wildlife

To address trophic transfer and biomagnification of COPCs from sediments, to emergent (aerial-phase aquatic) insects, to terrestrial vertebrate wildlife, a simple hazard model has been developed that idealizes exposure strictly by trophic transfer processes with no directs consumption of contaminated sediments. This is reasonable since emergent aquatic insects typically shed their final larval instar exoskeleton on clean substrates above the water level. This model is based on Equation 20 tailored for the diet of a terrestrial insectivore. The functional groups of vertebrate receptors for this exposure pathway (Figure 11) are:

- Aerial mammalian insectivore (sediment-based prey)
- Aerial avian insectivore (sediment-based prey)
- Amphibian insectivore (sediment-based prey)

The Figure 11 EPCEM indicates that several exposure pathways for sediments reaching terrestrial receptors are potentially complete, but only the food web transport pathway is evaluated directly for biomagnification of sediment-born contamination.

Limited modeling is needed to evaluate the HQ and develop sediment ESLs for the sediment trophic transfer model. The model shown in Equation 31 is based on the uptake of contamination from sediments by benthic insects, and the subsequent ingestion of emergent forms by a terrestrial insectivore. The HQ for the trophic transfer model is:

$$HQ_{i,j} = \frac{I_j \cdot CR_{i,invert}}{TRV_{i,j}} \cdot BMF_i \qquad \text{Equation 31}$$

where,

- I_j is the daily dietary ingestion rate for insectivore *j* (kg BW/kg ingested prey/day, dry weight);
- $CR_{i,invert}$ is a concentration ratio for COPC *i* in sediment to invertebrate ([mg COPC/kg invertebrate]/[mg COPC/kg sediment], dry weight).
- $TRV_{i,j}$ is the dietary TRV (food only) for COPC *i* and receptor *j* (mg COPC/kg body weight/day, dry weight);
- BMF_i is the sediment to invertebrate to vertebrate consumer biomagnification factor.

The determination of biomagnification factors (BMFs) may follow EPA (1995b, Appendix K) or another scientifically defensible methodology. BMFs should represent a trophic transfer of two or more. (BMFs are sometimes referred to as TTFs [trophic transfer factors].) This factor should be considered to equal unity for constituents that are not known to biomagnify.

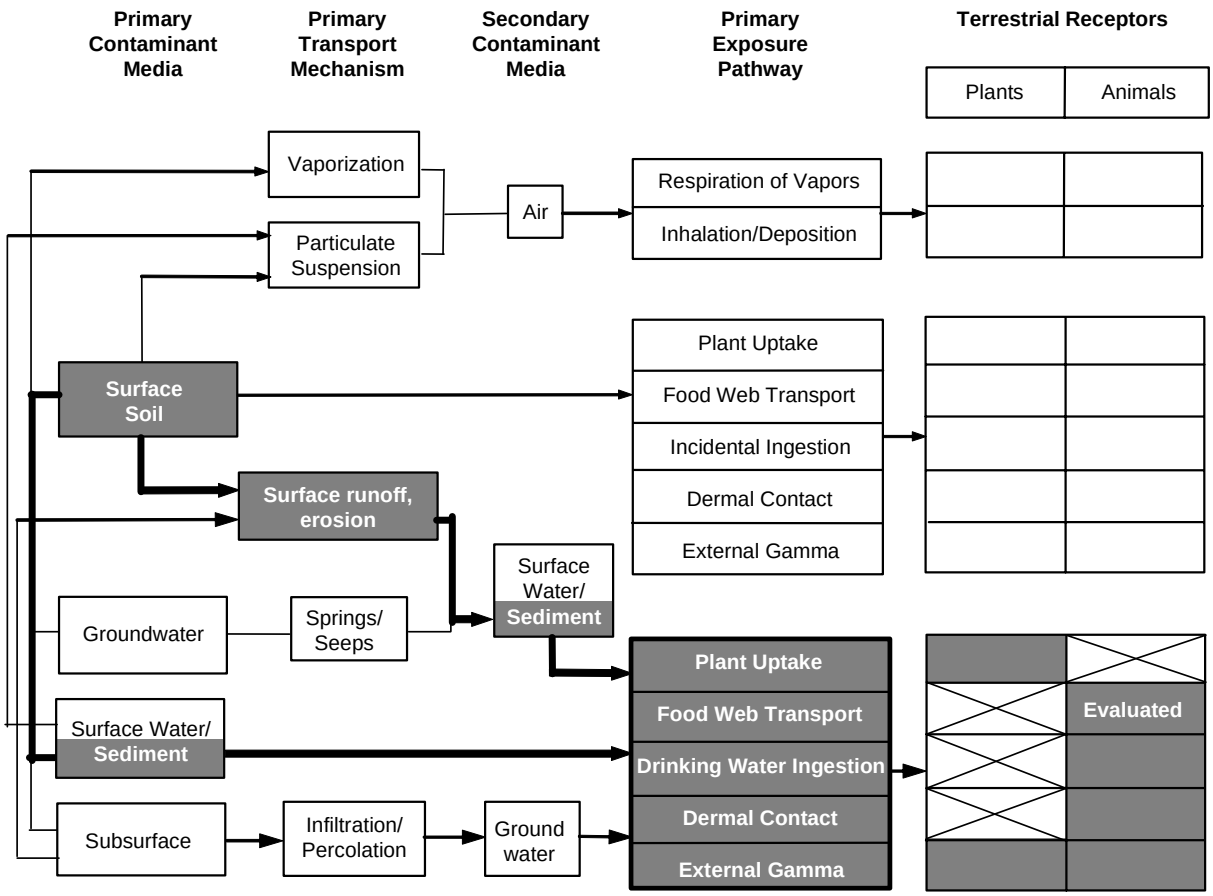
In order to calculate an ESL for any given COPC, the HQ is set to unity and the equation is solved for the concentration of COPC *i* according to the parameters for each receptor *j*. This solution is performed as:

$$ESL_{i,j} = \frac{TRV_{i,j}}{I_j \cdot CR_{i,invert} \cdot BMF_i} \qquad \text{Equation 32}$$

where:

- $ESL_{i,j}$ is the sediment ESL for COPC *i,* receptor *j* given the conditions of biomagnification, thus the BMF factor (mg COPC/kg sediment, dry weight),

Figure 11. Ecological pathways conceptual exposure model for terrestrial insectivores from sediment-born contaminants.



Boxes marked in gray are potentially complete pathways for the specified exposure model; those labeled "Evaluated" are directly evaluated with standard screening tools identified in this document, while fields left gray but blank represent pathways for which no standard screening methodology exists. Fields marked with "X"s indicate incomplete pathways. Boxes with a blank field and no "X" indicate pathways irrelevant to specified exposure model.

Transfer to higher trophic level (carnivores) is not accounted for by Equations 31 and 32, and should be addressed in the uncertainty analysis. Indeed, the *BAF* tends to increase in value with each trophic exchange.

Organisms that require simultaneous consideration as insectivores for emergent (aerial-phase aquatic) and terrestrial invertebrates, should also be evaluated with a multiple pathways approach for calculation of an HI (Section 4.3.2). The CSM should be consulted for consideration of simultaneous (multimedia) exposure models required for any of several receptor groups.

Uncertainties associated with the ESL derivation for emergent (aerial-phase aquatic) insect consumers include the selection of the TRV value for the insectivore functional group (mammals, birds, amphibians), for which there may be a dearth of information. In this context, uncertainty can be minimized by means of on-site studies (baseline risk assessment) and use of studies on

species that are phylogenetically close to the species under consideration in the risk assessment. Uncertainty also arises from selection of a concentration ratio that adequately describes the sequestration of COPCs from sediment to aquatic invertebrates. This relationship may be poorly developed for many COPCs, and may be necessarily modeled Additionally, *BMF*s may not be available for all chemicals and for all levels of foodchain transfer. Models used to describe the uptake of a COPC from sediment by a benthic insect (or other benthic invertebrate) must be explicitly exposited in the risk analysis. Any uncertainties regarding ESL derivation for terrestrial insectivores should be discussed fully in the uncertainty analysis.

4.3.4.2 Water to Vertebrate Wildlife

To address transfer of COPCs from water through the food chain, a trophic transfer model has been developed. This model is identical to Equation 31, but tailored for the diet of a terrestrial piscivore. This model is considered fundamentally different from the model for emergent insects from aquatic sediments, as it addresses the potential of biomagnification of constituents from the free water column to terrestrial vertebrate receptors with a primarily flesh diet. For the BMI Complex and vicnity, there is but one known species of fish, which is mosquitofish *(Gambusia affinis)*, an introduced species to the Kerr-McGee Seep and other water bodies in the vicinity. (Note that other water bodies in the vicinity may harbor more species of fish than mosquitofish.) Functional groups of consideration for this exposure pathway (Figure 12) are:

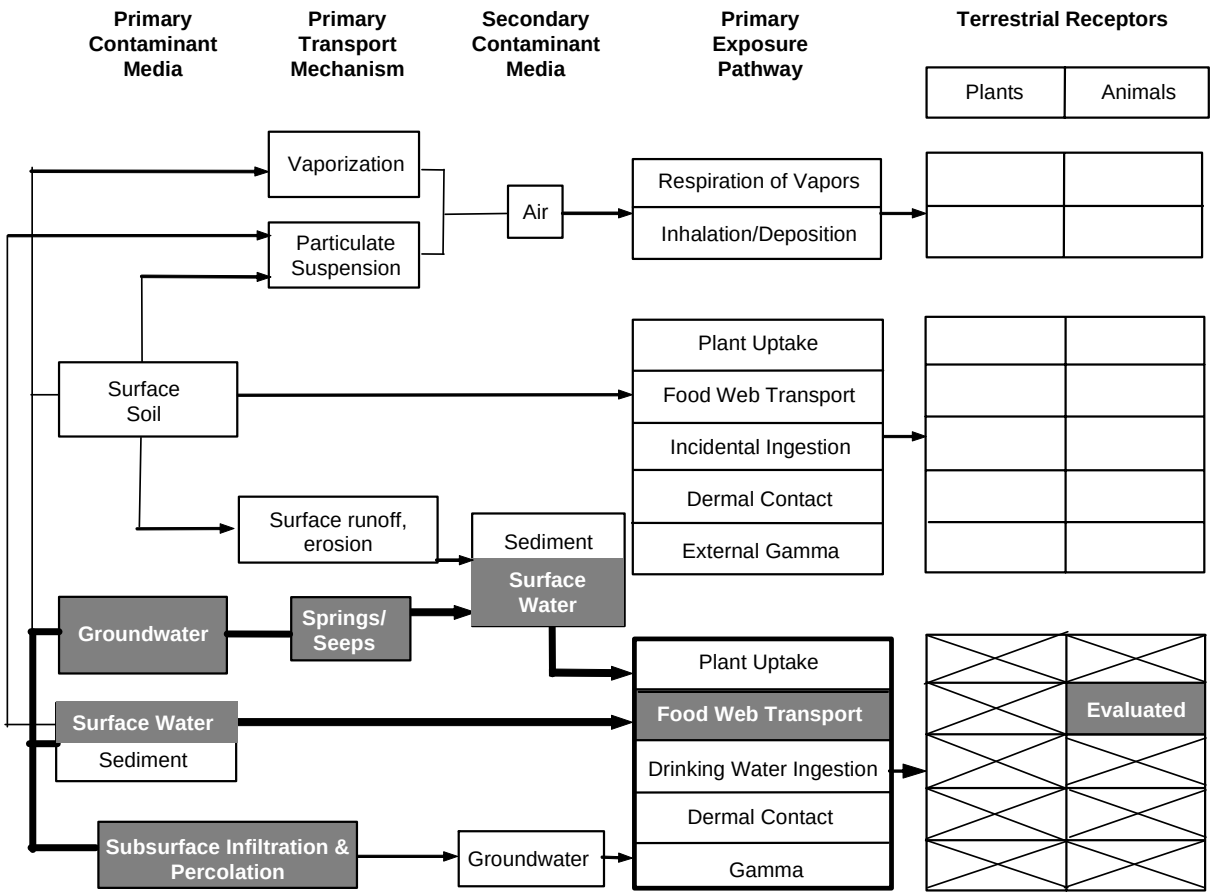
- Mammalian piscivore (prey from free water column)
- Avian piscivore (prey from free water column)
- Reptilian/Amphibian piscivore (prey from free water column)

It is possible that not all of the functional groups that are listed are found on the BMI Complex or vicinity, and this pathway may be demonstrated to be of inconsequential concern. However, consideration for the quality of water, particularly that issuing from the Kerr-McGee Seep, is of utmost importance and may not be dismissed without adequate demonstration that exposure pathways (as depicted in Figure 12) do not exist.

The Figure 12 EPCEM (water-born contaminant foodchain transfer pathway) indicates that several exposure pathways to terrestrial receptors are potentially complete water-born contamination.

Limited modeling is needed to evaluate the HQ and develop ESLs for the evaluation of contaminants dissolved in the free water column and their trophic transfer to terrestrial organisms via consumption of fish (or other aquatic vertebrates). The model shown in Equation 33 is based on the uptake of contaminants from free water to aquatic vertebrates, and their subsequent ingestion by a terrestrial piscivore.

Figure 12. Ecological pathways conceptual exposure model for terrestrial piscivores from water-born contaminants.



Boxes marked in gray are potentially complete pathways for the specified exposure model; those labeled "Evaluated" are directly evaluated with standard screening tools identified in this document, while fields left gray but blank represent pathways for which no standard screening methodology exists. Fields marked with "X"s indicate incomplete pathways. Boxes with a blank field and no "X" indicate pathways irrelevant to specified exposure model.

The HQ for the trophic transfer model is:

$$HQ_{i,j} = \frac{I_j \cdot CR_{i,fish}}{TRV_{i,j}} \cdot BMF_i \qquad \text{Equation 33}$$

where,

- I_j is the daily dietary ingestion rate for piscivore *j* (kg BW/kg ingested prey/day, dry weight);
- $CR_{i,fish}$ is a concentration ratio for COPC *i* in free water to fish ([mg COPC/kg fish]/[mg COPC/kg sediment], dry weight).
- $TRV_{i,j}$ is the dietary TRV (food only) for COPC *i* and receptor *j* (mg COPC/kg body weight/day, dry weight);

- BMF_i is the free water to fish to vertebrate consumer biomagnification factor.

The determination of biomagnification factors (BMFs) may follow EPA (1995b, Appendix K) or another scientifically defensible methodology. BMFs should represent a trophic transfer of two or more. (BMFs are sometimes referred to as TTFs [trophic transfer factors].) This factor should be considered to equal unity for constituents that are not known to biomagnify.

In order to calculate an ESL for any given COPC, the HQ is set to unity and the equation is solved for the concentration of COPC *i* according to the parameters for each receptor *j*. This solution is performed as:

$$ESL_{i,j} = \frac{TRV_{i,j}}{I_j \cdot CR_{i,fish} \cdot BMF_i} \qquad \text{Equation 34}$$

where:

- $ESL_{i,j}$ is the sediment ESL for COPC *i*, receptor *j* given the conditions of biomagnification, thus the BMF factor (mg COPC/kg sediment, dry weight),

Transfer to higher trophic level (carnivores) is not accounted for by Equations 31 and 32, and should be addressed in the uncertainty analysis. Indeed, the *BAF* tends to increase in value with each trophic exchange.

Organisms that require simultaneous consideration as piscivores and carnivores for other terrestrial functional groups, should also be evaluated with a multiple pathways approach for calculation of an HI (Section 4.3.2). Thus, calculation of an HI is required for functional groups that fall within this context. The CSM should be consulted for consideration of simultaneous (multimedia) exposure models required for any of several receptor groups.

Uncertainties associated with the ESL derivation for piscivores include derivation of the TRV value for the representative piscivore (mammals, birds, herpeptile). In this context, uncertainty can be minimized by use of studies on species that are phylogenetically close to the species under consideration in the risk assessment. Additionally, uncertainty arises from selection of a $CR_{i,fish}$ that adequately describes the sequestration of a COPC from water to fish. This relationship may be poorly developed for many COPCs, and may be necessarily modeled, particularly for organic COPCs. Models required to describe the uptake of a COPC from the free water column by fish must be explicitly exposited in the risk analysis. Any uncertainties regarding ESL derivation for piscivores should be discussed fully in the uncertainty analysis.

4.4 Aquatic Benchmark Value Comparisons

4.4.1 Water Quality Benchmarks

Aquatic communities integrate contaminants from a variety of sources, including terrestrial soil and senescent receptor contamination (from suspension in runoff), direct discharge of contaminants into surface water, and the influx of contaminated groundwater. Water of potential concern to ecological receptors at and around the BMI Complex includes surface water (particularly sheetwater flow to ephemeral and permanent drinking water sources, e.g., the Kerr-McGee Seep area) and shallow groundwater accessed directly by organisms living in soil and also daylighting at the Kerr-McGee Seep (or any other seeps that may exist or emerge in the future).

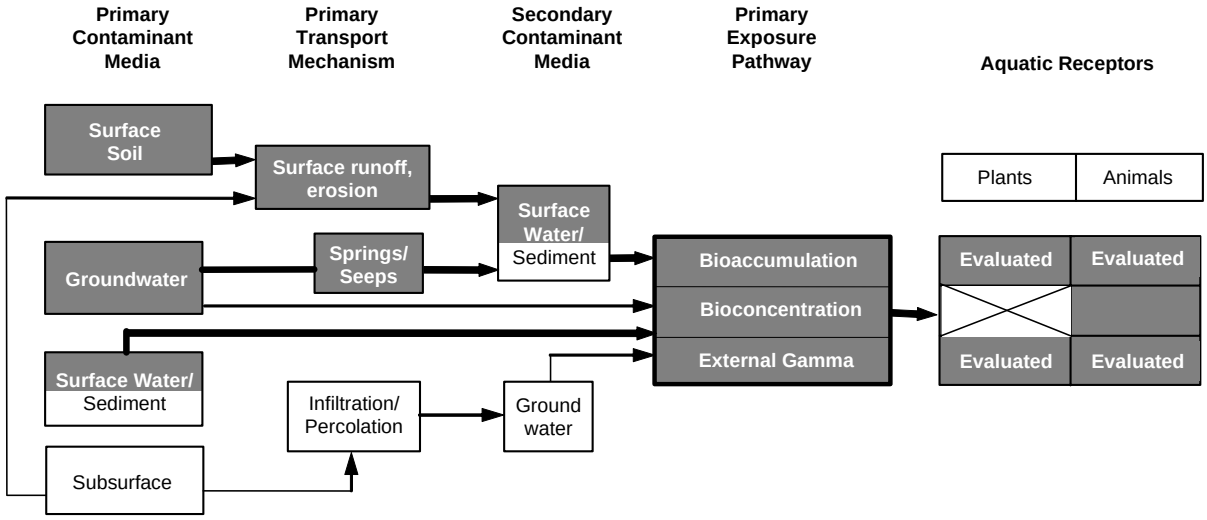
For the purposes of this portion of the screening analysis, only exposure pathways related to surface water and groundwater affecting surface water are evaluated. For portions of the BMI Complex and vicinity where terrestrial plants may access groundwater, uptake must be considered as part of the terrestrial model for plants (Section 4.3.2.1). For other aspects of contaminant transport arising from contaminated groundwater contacting soil, sediment, or surface water, consideration should be reflected in the CSM and should be fully discussed therein. Any receptor groups that may be affected by contact with media that is secondarily contaminated by tainted groundwater should be considered with the appropriate SLERA criteria. Where uncertainty arises regarding exposure pathways for ecological receptors contacting contaminated groundwater (primarily or secondarily via contaminant transport), a discussion of this exposure modality should be included in the uncertainty analysis. When the fate of COPCs in the biotic environment cannot be adequately determined, COPCs must not be dismissed from screening without explicit (weight of evidence) consideration of risk potentiated to any and all biota that may be part of complete contaminant pathway.

Water samples may be filtered (suspended solids removed) or unfiltered. Unfiltered samples have greater or equal concentrations of COPCs than filtered samples due to the presence of sediments and organic particulates. As an upper bound of potential exposure, unfiltered water should be used in screening evaluations. If unfiltered samples show no potential risk, no further evaluation of the filtered samples is needed. If unfiltered samples show potential risk to aquatic biota under consideration, water samples for chemical content should be evaluated on the basis of filtered samples, as this is considered the bioavailable fraction of these constituents in free water (EPA 1996b). If filtered water samples show no potential risk to aquatic biota but unfiltered samples do show a potential risk, then investigation should be directed toward the potential risks posed by aquatic sediments (Sections 4.3.4.1, 4.4.2).

Methods for screening water are based on exposure pathways to aquatic organisms. For aquatic organisms, the SLERA approach assumes that aquatic organisms are generally exposed to the greatest fraction of contamination by means of direct media contact, i.e., continuous bodily contact (primarily gill and osmoregulatory organs) with water. Ecological risk screening for waterborne COPCs, therefore, pertains to receptors associated with unladen (sediment-free) benthic surfaces and the free water column of both lentic and lotic systems.

The exposure model for pathways of water-born contamination to aquatic receptors is provided in Figure 13. (Aquatic foodchain modeling for concerns of bioaccumulation and biomagnification of waterborne constituents is found in Section 4.3.4.)

Figure 13. Ecological conceptual exposure model for aquatic receptors exposed to contaminants in water.



Boxes marked in gray are potentially complete pathways for the specified exposure model; those labeled "Evaluated" are directly evaluated with standard screening tools identified in this document, while fields left gray but blank represent pathways for which an alternative screening methodology exists. Fields marked with "X"s indicate incomplete pathways.

To be broadly protective of aquatic plant and animal species, EPA has developed methods (EPA 1995a, EPA 1996b) intended to protect a large fraction (roughly 95%, unless otherwise stated) of species (plants and animals) found in aquatic environments at large, and not specifically associated with water bodies of the northern Mojave Desert region. However, by protecting aquatic species in general, the particular species selected to be representative of feeding guilds in the aquatic realms of the BMI Complex are presumably also to be protected.

The EPA's water quality criteria (WQCs) offer a broad basis of protection for organisms living the free water column of lotic and lentic environments (EPA 1995a, 1996b). WQCs are used to derive water quality benchmarks (WQBs) for the SLERA that serve as aquatic ESLs. WQCs are derived from a variety of primary studies concerning toxicological testing of aquatic organisms. These criteria differ in the methods and/or rigor of their development. Consequently, WQBs must be adopted in hierarchically, similarly to SQBs, based upon the rigor of derivation and ubiquity of protection afforded aquatic species.

For any single COPC, there may be more than one WQB. WQBs are hierarchically ranked for use in screening in the following order, based upon comprehensiveness of derivation, level of protection afforded based on their derivation, and recommendation of the EPA (EPA 1995a, 1996b, 2002a) in the development of water quality criteria:

1. WQB1: Chronic National Ambient Water Quality Criteria set forth by EPA (2002a).
2. WQB2: Great Lakes methodology Tier I final chronic value (FCV) (EPA 1995a).
3. WQB3: Great Lakes methodology Tier II secondary chronic value (SCV) (EPA 1995a).
4. Other sources, including Suter and Tsao (1996) and Suter (1996).

Fundamentally, the higher the rank of the WQB in the list, the more inclusive and stringent it is of criteria for calculation. Higher ranked WQBs (as above) more broadly cover the range of conditions and taxonomic variability found in aquatic environments, and therefore have a broader applicability and utility in the protection of aquatic life. If more than one WQB exists for a given COPC, then they should be used in the above rank order. WQBs that are derived from limited data, including those derived solely from acute data, should be discussed in the uncertainty analysis for their applicability in the SLERA. When WQBs are unavailable or not calculable for a given COPC in water, then the COPC is retained as a COPEC.

Table 2 shows how WQB values are hierarchically chosen and used as a screening-level benchmark (final water ESL) for aquatic organisms. Table 2 can be expanded to accommodate sources for any number of WQBs, such that if additional sources are viewed to be acceptable, additional columns are added to the right of the "WQB3" heading. When WQBs are not available or cannot be calculated for a given COPC, then the COPC is retained as a COPEC and discussed in the uncertainty analysis.

Table 2. Method for obtaining a final non-radionuclide WQB.

COPC	Chronic NAWQC (WQB1) (mg/L)	Tier I value (WQB2) (mg/L)	Tier II value (WQB3) (µg/L)	Final Water ESL (µg/L)
U	Value	Value	Value	WQB1
V	No value	Value	Value	WQB2
W	No value	No value	Value	WQB3
X	No value	No value	No value	No WQB, retain COPC as COPEC

* Based on wildlife exposure calculation.

The following paragraphs describe the methods used to select aquatic ESLs, from the most (WQC1) to the least (WQC3) preferable. The discussion uses a question-and-answer approach to aid in the selection of WQCs. The sources cited in the discussion provide more comprehensive information on the calculations used to determine the various criteria and benchmarks. Suter (1996) presents a list of many National Ambient Water Quality Criteria (NAWQC), Tier II values, and other TRVs.

➢ ***Water Quality Benchmark 1: Chronic Ambient Water Quality Criteria***

- Are chronic NAWQCs available as set forth by EPA (2002a)?
- If "**yes**" use the NAWQC as the WQB and final aquatic ESL.
- If "**no**" go to the WQB2 section.

NAWQCs have been developed for chronic exposure of aquatic organisms to some waterborne chemicals by EPA's Office of Water (OW) under the Clean Water Act, Section 304 (EPA

2002a). The development of NAWQCs is outlined in EPA (1995a). NAWQC values are considered applicable or relevant and appropriate requirements (ARARs), therefore, should be considered foremost for final water ESL adoption (Sample et al. 1998). Metals are often water hardness-dependent and should be adjusted for site-specific conditions: see EPA (1996b) and EPA (2002a) for explanations/delineation of methods, as methods require analyte-specific information).

➢ ***Water Quality Benchmark 2: Tier I Water Quality Criteria***

- Can Tier 1 WQCs be derived? (See criteria for derivation below.)
- If "**yes"** use the methods described in this section for deriving a Tier I WQC (WQB2) as the final aquatic ESL.
- If "**no"** go to the WQB3 section.

The EPA recommends that Tier I WQCs, based on the determination of chronic values (CVs), be developed in the absence of NAWQCs. Tier I CVs can be determined using methods of the Great Lakes Water Quality Initiative, as detailed in EPA (1995a). A CV may be obtained by calculating the geometric mean of the lower and upper chronic limits from a chronic test or by analyzing chronic data using regression analysis. The lower chronic limit, as defined in EPA (1995a), corresponds to a no observed effects concentration (NOEC) and the upper chronic limit corresponds to a lowest observed effect concentration (LOEC). A discussion of the acceptability criteria and details of Tier I WQC and CV determinations can be found in EPA (1995a, pp. 15395–15399). Supporting information for derivation of Tier I WQC values may be obtained from the AQUIRE database (AQUIRE 1997). Similar information may also be available in primary literature. The fundamental requirements and methods for deriving Tier I WQCs are outlined below.

To derive Tier I WQCs, results of chronic toxicological tests must be used meeting acceptability criteria of EPA (1995a, p. 15397). In addition, at least one CV test result must follow from *each* of the following taxonomic categories:

- the family *Salmonidae* in the class *Osteichthyes*
- one other family in the class *Osteichthyes*
- a third family in the phylum *Chordata* (e.g., fish, amphibian)
- a planktonic crustacean (e.g., a cladoceran, copepod)
- a benthic crustacean (e.g., ostracod, isopod, amphipod, crayfish)
- an insect (e.g., mayfly, dragonfly, damselfly, stonefly, caddisfly, mosquito, midge)
- a family in a phylum other than *Chordata* or *Arthropod*a (e.g., *Annelida*, *Molluska*, *Rotifera*)
- a family in any order of insect or any phylum not already represented.

For each species (within the taxonomic categories listed above) for which at least one CV is available, the species mean chronic value (SMCV) is calculated as the geometric mean of the available values, given their correspondence of measurement units. For each genus for which one or more SMCVs are available, the genus mean chronic value (GMCV) is calculated as the

geometric mean of the SMCVs available for species within the genus. When these data are compiled, then complete the following algorithm:

1. Order the GMCVs from low to high.
2. Assign ranks, *r*, to the GMCVs from "1" for the lowest to "n" for the highest. If two or more GMCVs are identical, they are assigned successive ranks.
3. Calculate an empirical cumulative probability *(P)* for each GMCV as r/(n+1).
4. Select the four GMCVs with cumulative probabilities closest to 0.05.
5. Using the four selected GMCVs and *P*s, calculate a final chronic value (FCV) following EPA (1995a), outlined below.

FCVs are calculated using the following mathematical relationships:

$$S^2 = \frac{\left[\sum_1^4 (\ln GMCV)^2\right] - \frac{\left[\sum_1^4 (\ln GMCV)\right]^2}{4}}{\left[\sum_1^4 P\right] - \frac{\left[\sum_1^4 \sqrt{P}\right]^2}{4}} \qquad \text{Equation 35}$$

$$L = \frac{\left[\sum_1^4 \ln GMCV\right] - \left[S \cdot \sum_1^4 \sqrt{P}\right]}{4} \qquad \text{Equation 36}$$

$$A = L + S \cdot \sqrt{0.05} \qquad \text{Equation 37}$$

$$FCV = e^A \qquad \text{Equation 38}$$

FCVs are also calculable based on acute values. The acute value method utilizes the species mean acute value (SMAV) and genus mean acute value (GMAV) in the same manner, and calculated using the same methodology, as the SMCV and GMCV, respectively. To obtain a final acute value (FAV), Equations 35 through 38 are used substituting GMAV for GMCV, with *FAV* substituted for *FCV* in Equation 38. An FCV is, then, arrived at by dividing the FAV by the final acute-chronic ratio (FACR). The FACR is the geometric mean of at least three acute chronic ratios (ACRs), which is the ratio of an acute dose (as determined by an acute toxicity test) to the CV (as determined for the same organism in the same study); for example, LC_{50}/CV, where the LC_{50} is the lethal concentration of 50% of the experimental population. (Again, a CV may be obtained by calculating the geometric mean of the lower and upper chronic limits from a chronic test or by analyzing chronic data using regression analysis; see EPA (1995a), pp. 15395–15399 for CV acceptability criteria). Each ACR is derived from a test on <u>one</u> species. However, the FACR must be calculated from ratios derived from at least <u>three</u> different aquatic taxa as listed below:

- At least one is a fish

- At least one is an invertebrate
- At least one species that is an acutely sensitive (e.g., a daphnid) freshwater species (the other two may be saltwater species).

If these requirements are met, the methodology is as follows:

- For each species, calculate the species mean acute-chronic ratio (SMACR) as the geometric mean of the ACRs available for the species. (The requirements for meeting the ACR criteria are very specific and rigorous; see EPA (1995a), p. 15398).
- Calculate the FACR as the geometric mean of the SMACRs, following the taxonomic criteria listed above.
- Calculate an FAV following the protocol listed above and using Equations 5-8.
- Calculate the FCV by dividing the FAV by the FACR.

Generally, the final Tier I (WQC2) criterion is considered the FCV. However, if utilizing the acute value method, if one-half the FAV is lower than the calculated FCV, then one-half the FAV should be used as the Tier I (WQC2) criterion in lieu of an FAV based on the acute value method. Also, the FCV must be compared with the final plant value (FPV). The FPV (EPA 1995a, p. 15399) is defined as "the lowest plant value that was obtained with an important aquatic plant species in an acceptable toxicity test for which the concentrations of the test material were measured and the effect was biologically important." And "A plant value is the result of a 96-hour test conducted with an alga or a chronic test conducted with an aquatic vascular plant." If the lesser of the FCV or one-half of the FAV exceeds the FPV, then the FPV should be used as the Tier I WQC (i.e., WQB2) criterion.

In the end, the WQC2 for waterborne contamination is chosen as the ESL. In the case where a Tier I WQC cannot be developed, then a Tier II WQC (WQB3) may be adopted, as outlined below.

➢ ***Water Quality Benchmark 3: Tier II Water Quality Criteria***

- If no NAWQC exists, and a Tier I WQC cannot be derived, can a Tier II WQC be derived?
- If "**yes"** use the Tier II WQC (WQB3) as the final aquatic ESL.
- If "**no"** then the COPC becomes a COPEC and will be discussed in the uncertainty analysis.

If three or more experimentally determined ACRs (see above) are available for the COPC, the FACR should be determined as described above. If fewer than three ACRs can be calculated, it is assumed that each "missing" ACR value is equal to 18, so the total number of "ACRs" equals three. A secondary acute-chronic ratio (SACR) is calculated as the geometric mean of the three ACRs. The secondary chronic value (SCV) is calculated using one of the following equations:

$$SCV = \frac{FAV}{SACR} \qquad \text{Equation 39}$$

$$SCV = \frac{SAV}{FACR} \qquad \text{Equation 40}$$

$$SCV = \frac{SAV}{SACR} \qquad \text{Equation 41}$$

where, Equation 9 uses the SACR and FAV as calculated under Tier I methods, Equation 10 uses methods for calculating a secondary acute value (SAV), as outlined below, and the FACR, as outlined under Tier I methods, and Equation 41 uses the SAV, as outlined below, and SACR.

Species Acute Values (SAVs) are presented in detail in EPA (1995, p. 15400). To calculate a SAV, a minimum of one genus mean acute value (GMAV) for a daphnid (Crustacea: Cladocera) must be used. (Again, the GMAV is calculated as the geometric mean of the species mean acute values [SMAVs] available for the genus.) The lowest GMAV calculated is then divided by the secondary acute value factor (SAVF, Table 3). The requirement of at least one daphnid GMAV has been criticized for restricting the number of benchmarks that can be calculated (Suter 1996), however, Suter and Tsao (1996) provide SMAVs for calculating SAVs when no daphnid GMAVs can be calculated. These values are also presented in Table 3.

Table 3. Secondary acute value factors (SAVFs) for estimation of Tier II secondary chronic values (SCVs).

Number of GMAVs[a]	**SAV for Data Set with Daphnid Values[a]**	**SAV for Data Set without Daphnid Values[b]**
1	21.9	242
2	13.0	64.8
3	8.0	36.2
4	7.0	20.1
5	6.1	12.9
6	5.2	9.2
7	4.3	7.2

[a] Factors taken from EPA (1995a).
[b] Factors taken from Suter and Tsao (1996).

The lowest of the SCV or the FPV (final plant value, see above) is then considered the Tier II SCV. Tier II values are expected to be higher than NAWQCs in no more than 20% of all cases. The Tier II SCV is then adopted as WQB3.

- ***Other Potential Water Quality Benchmark Resources***

When an NAWQC, a Tier I, or a Tier II value is not available or cannot be calculated, other toxicologically based benchmarks are to be used from other sources, particularly primary literature. Suter (1996) and Suter and Tsao (1996) provide information on a variety of potential benchmarks and resources, although more contemporary resources may be available. When an NAWQC, a Tier I, or a Tier II value is not available or cannot be calculated for a given COPC, then the COPC is retained as a COPEC, and should be discussed in the uncertainty section.

4.4.1.1 Summary of Water ESL Derivations

Water quality benchmarks (WQBs) that are used in screening-level risk assessment for aquatic receptors may be derived from a variety of sources. More than one WQB may be available for any given constituent and are employed in a hierarchical fashion (see Table 4). Potential bioaccumulation concerns for water are considered in the wildlife exposure model (Section 4.3.4).

Table 4. Summary of sources for water ESLs.

COPC Type	Aquatic Receptors	Bioaccumulation & Biomagnification to Terrestrial Vertebrates
Organic and inorganic chemicals dissolved in water.	The following are used in order of preference: 1. WQB1: Chronic NAWQC set forth by EPA (2002a). 2. WQB2: Great Lakes methodology Tier I final chronic value (FCV) (EPA 1995a). 3. WQB3: Great Lakes methodology Tier II secondary chronic value (SCV) (EPA 1995a). 4. Other sources, including Suter and Tsao (1996) and Suter (1996).	See the wildlife exposure model, Section 4.3.4.

4.4.2 Sediment Quality Benchmarks

Sediment may be defined as unconsolidated material composed of particles deposited from suspension in air, ice, or water, or from solution in water. Particles in sediment may consist of mechanically formed fragments of rock, chemically formed precipitate from solution, or organically formed precipitate consisting of remains or secretions of plants and animals (USDA 1980). Sediments by these definitions have not been compressed or metamorphosed into solid (rock) form. This definition includes sediments in lotic (running freshwater) settings, including active channels, inactive channels, and floodplain geomorphic settings, and lentic (stillwater) settings, including lakebeds (intermittent and perennial) and shorelines, seabeds and shorelines, former lakebed and seabed settings.

Many of the settings in which sediments are found, typically have ecological communities associated with terrestrial realms. Sediments found in these conditions may be accessed as soils by terrestrial or semi-aquatic organisms, from an ecological perspective. These conditions may include vascular plants and associated organisms that live in near-shore environments, as well as "inactive" sediments, not presently vulnerable to movement by wind and water, associated with terrestrial and intermittently dry realms. Additionally, some terrestrial animals may feed on organisms that are intimately associated with sediments. Thus, evaluation of ecological risk for sediments that support terrestrial and semi-aquatic communities proceeds as part of a wildlife model (Section 4.3.4.1), and we emphasize that sampling performed for such exposure pathways must conform to standards that are appropriate for the terrestrial and semi-aquatic organisms that are being evaluated.

On the other hand, so-called "active" sediments, such as those found associated with perennial and intermittent aquatic environments, and accessed by organisms that are obligately confined to those environments, require special considerations due to concerns of bioaccumulation of

contaminants directly absorbed from the sediment and associated pore water medium. Following the exposure pathways of aquatic organisms in the evaluation of exposure to contaminated sediments is the focus for the development of sediment quality benchmarks (SQBs). Thus, evaluation of potential ecological risk for contamination found in active sediments is considered herein, and is primarily considered for protection of aquatic life.

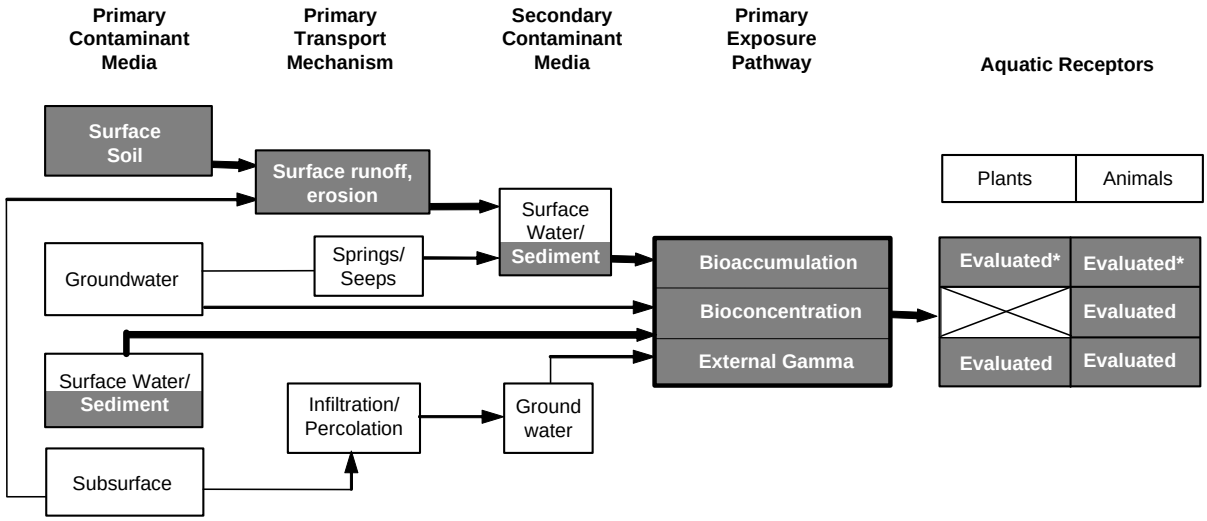
Transport of contaminated sediments to aquatic environments includes suspension in (1) discharge of effluents into perennial and intermittent water bodies, (2) surface water runoff from contaminated soils, (3) infiltration of surface water into shallow and/or deep groundwater, (4) mass wasting, and (5) wind-driven transport. Of primary concern are the first three transport mechanisms, which are included in Figure 14. In the case of the BMI Complex and surrounding areas, mass wasting is not an issue unless flooding from the Las Vegas Wash causes the loss of stream embankments that affect the property. Wind-blown soils may be influential in affecting the sediment load of a water bodies in desert regions. This is particularly true where contaminated surface soils have been exposed due to a loss or lack of vegetation, commonly seen across the BMI Complex and vicinity. This mode of transport, however, is likely minor compared to runoff from storm events, but should be identified during site-specific problem scoping. With the limited water resources in the region, the primary focus should be on pathways of sediment transport from areas adjacent to or contiguous with permanent or seasonally intermittent surface water resources that may harbor aquatic life.

Protecting sediment quality is increasingly viewed as a logical extension of water-quality protection, which helps to emphasize the interrelationship between sediment and water as exposure media. Chapman (1989) cites several reasons for the employment of sediment quality criteria (SQCs) and SQBs, including

- Various toxic contaminants, found only in trace amounts in the water column, accumulate in sediments to elevated levels.
- Sediments serve as reservoirs for and sources of contaminants affecting the water column.
- Sediments accumulate and integrate contaminant concentrations over time, whereas water column contaminant concentrations are much more variable and dynamic.
- Sediments provide habitat for benthic organisms and others, and may affect feeding, and rearing areas for many aquatic and semi-aquatic organisms that rely on the water column.

Sediment benchmarks for the protection of aquatic life are derived from information about direct effects of contaminated sediments on aquatic organisms. Similar to WQCs, the EPA has developed methods for minimizing the risks posed by contaminated sediments, and that are broadly protective of aquatic plant and animal species (EPA 1995a, EPA 1996b). These methods are intended to protect a large fraction (roughly 95%, unless otherwise stated) of species (plants and animals) found in aquatic sedimentary environments at large, and not specifically associated with water bodies of the northern Mojave Desert region. However, by protecting aquatic species in general, the particular species selected to be representative of feeding guilds in the sedimentary realms of the BMI Complex and the surrounding areas are presumably also to be protected.

Figure 14. Ecological conceptual exposure model for aquatic receptors exposed to contaminants in aquatic sediments.



Boxes marked in gray are potentially complete pathways for the specified exposure model; those labeled "Evaluated" are directly evaluated with standard screening tools identified in this document. Fields marked with "X"s indicate incomplete pathways.

Sediment ESLs result from calculating sediment quality benchmarks (SQBs) from national ambient water-quality criteria (NAWQC; EPA 1998b, 2002a), Great Lakes Tier I water-quality criteria (WQC; EPA 1995a), Great Lakes Tier II water-quality criteria (WQC; EPA 1995a), standardized tests on sediment-dwelling invertebrates (EPA 1996a), or sediment effects concentrations (SECs) (EPA 1996a). Sediment ESLs will be developed from SQBs in this section.

The SQC/SQB methodologies adopted for ecological screening of contaminated sediments conform with those proposed by the EPA for developing ecotox thresholds (EPA 1996b). These methods for screening sediments are based on the assumption that aquatic organisms are generally exposed to the greatest fraction of contamination by means of direct media (sediment) contact, i.e., continuous bodily contact (primarily gill and osmoregulatory organs) with sediments in water. Screening methods for contaminated sediments in the aquatic realm pertains primarily to organisms associated with laden benthic surfaces.

As mentioned, sediment quality benchmarks (SQBs) come from a variety of sources but are all based upon toxicological information derived from primary studies. However, not all of the benchmarks are equal, as they may be derived from different measurement endpoints and sources of variable quality. Values from studies using freshwater sediments have been assigned highest priority, and are generally values endorsed by EPA or approved authorities. SQBs are hierarchically ranked for use in screening in the following order, based upon comprehensiveness of derivation, level of protection afforded based on their derivation, and recommendation of the EPA (EPA 1995a, EPA 1996a and b, Jones et al. 1997):

1. SQBs calculated utilizing SQCs derived from national ambient water-quality criteria (NAWQC)or from Great Lakes Tier I water-quality criteria (WQC) (EPA 1995a) according to EPA (1996b);
2. SQBs calculated from Great Lakes Tier II WQC (EPA 1995a) according to EPA (1996b);
3. SQBs derived from sediment effects concentrations (SECs), described below (EPA 1996a);
4. EPA Region IV screening values (EPA 1996c);
5. Jones et al. (1997), Long et al. (1995).

For example, if an SQB can be calculated from NAWQC or Tier I WQC (EPA 1995a), then this SQC becomes the preferred criterion. If an SQB cannot be calculated from NAWQC or Tier I WQC, then it may be calculated from Great Lakes Tier II WQC, and is used preferably to an SQB arrived at based on SECs, and so on.

Table 5 shows how SQB values are used to derive a final sediment ESL. Table 5 can be expanded to accommodate sources for any number of SQBs, such that if additional sources are viewed to be acceptable, additional columns are added to the right of the "SQB3" heading. When SQBs are not available or cannot be calculated for a given COPC, then the COPC is retained as a COPEC and discussed in the uncertainty analysis.

Table 5. Method for obtaining the final sediment ESL for non-radionuclides.

Contaminant	SQB1[a] (mg/kg)	SQB2[b] (mg/kg)	SQB3[c] (mg/kg)	Final Sediment ESL (mg/kg)
U	Value[d]	Value	Value	SQB1
V	No value[e]	Value	Value	SQB2
W	No value	No value	Value	SQB3
X	No value	No value	No value	No sediment SQB available or calculable; retain COPC as COPEC for sediment.

[a] SQB1 = Primary SQB value.
[b] SQB2 = Secondary SQB value.
[c] SQB3 = Tertiary SQB value.
[d] Value = SQB value available for that COPC.
[e] No value = No SQB value available for that COPC.

Each of the SQBs are described and exposited in the subsections that follow. Jones et al. (1997) summarize the SQB methods and expand upon some of the definitions used. This discussion uses a question-and-answer approach to aid in selecting SQBs to address COPCs. The sources cited in this discussion provide more comprehensive information on the calculations used to determine the various criteria and benchmarks.

➢ ***Sediment Quality Benchmark 1***

- Is the COPC a non-ionic organic compound?
- Does a NAWQC or Tier I WQC exist for the COPC?
- If "**yes**" for both of the above, follow the directions outlined below for generating SQBs.
- If "**no**" to either of the above, go to the SQB2 section.

The calculation of an SQB1 requires the calculation of a sediment quality criterion (SQC). The preferred means of generating SQCs is the "equilibrium partitioning method" (EqP) proposed by the EPA (EPA 1996b). SQCs have been proposed by EPA's Office of Water for acenaphthene, dieldrin, endrin, fluoranthene, and phenanthrene (EPA 1996b). A number of alternative benchmarks have been proposed by EPA (1996b) and Jones et al. (1997). All these values were derived using the EqP method, which quantifies the hydrophobicity of the chemical by using the octanol/water partition coefficient (K_{ow}) and determines the sorption capacity of the sediment by the mass fraction of organic carbon (f_{oc}) of the sediment. It is important to note that the EqP method is appropriate for *non-ionic organic compounds only* (EPA 1996c). The relationship between K_{ow} and the sediment organic carbon partitioning coefficient, K_{oc}, is described by Equation 42 (Di Toro 1985):

$$\log\left(K_{oc}\right) = 0.00028 + 0.983\log\left(K_{ow}\right) \qquad \text{Equation 42}$$

The EqP method assumes that pore water is in equilibrium with sediment and that pore water must meet water quality standards to be considered nontoxic (O'Connor et al. 1998). The EqP method is favored over direct measurement of a chemical in pore water because complexation of the chemical with dissolved organic carbon can be substantial. If the colloids or suspended solids available for direct ingestion by wildlife are not considered, then only the uncomplexed chemical in pore water (in equilibrium with the organic carbon fraction of the sediment) is bioavailable to aquatic organisms. Jones et al. (1997) stated "for highly hydrophobic chemicals and where there is significant dissolved organic carbon complexing, the solid-phase chemical concentration gives a more direct estimate of the bioavailable pore water COPC concentration than do the pore water concentrations."

The EqP approach requires four major assumptions (Jones et al. 1997):

1. Partitioning of the organic compound between the sediment fraction of organic carbon and interstitial water is stable at equilibrium.
2. The sensitivities of benthic species and those that occupy the free water column (those primarily tested in the development of water quality criteria) are similar.
3. The levels of protection afforded by WQBs are appropriate for benthic organisms.
4. Exposures of water-dwelling organisms to sediment-borne contamination are similar regardless of the feeding type or habitat.

EPA has concluded that the sensitivities of benthic organisms are sufficiently similar to those of water column species to tentatively permit using WQBs to derive sediment quality benchmarks (Jones et al. 1997). Because of complexities associated with metal binding in sediments (e.g., metal binding sites other than organic carbon, such as clay surfaces), the EqP approach is inappropriate for use with metals.

The equation for the SQC1 in units of mg/kg (see * notation, below) is:

$$SQC1 = f_{oc} \times K_{oc} \times (FCV \text{ or } CCC) \qquad \text{Equation 43}$$

where:

- f_{oc} (no units of mass) is the mass fraction of organic carbon for the sediment.
- K_{oc} (no units of mass) is the sediment organic carbon partitioning coefficient.
- *FCV* (µg/L) is the "final chronic value" from chronic NAWQC (Tier I toxicity values). See Section 4.4.1 for WQBs, and EPA (1995a) for details on the calculation of FCVs.
- *CCC* (µg/L) is the "criterion continuous concentration." See EPA (1998b) for recommended CCC values and EPA (1995a) for details on the calculation of CCCs.

** Because freshwater toxicity information is considered for deriving sediment benchmarks, it is assumed that a liter of freshwater is one kilogram mass for purposes of unit conversion.*

Using the above relationships, SQCs can be derived for any number of non-ionic organic compounds. This method uses normalized calculation of the f_{oc} to 1%. With this normalization, the SQB1 can be derived from the SQC, adjusted to site-specific conditions by a simple factor of:

$$SQB1 = \left(f_{oc\,[\text{site specific}]} \Big/ f_{oc[\text{normalized}]} \right) \times SQC1 . \qquad \text{Equation 44}$$

Using the SQCs published by the EPA and other sources as a basis for calculating the SQB1 requires a knowledge of site-specific conditions because of potentially varying levels of the mass fraction of organic carbon for the sediment (f_{oc}). Under most circumstances, the f_{oc} will be greater than 1%; thus, the SQB1 will be greater than the SQC. The EqP method is not valid when the f_{oc} is less than 0.2%.

➢ ***Sediment Quality Benchmark 2***

- Is the COPC a non-ionic organic compound?
- Does a Tier II level WQC exist for the COPC?
- If "**yes"** for both of the above, follow the directions outlined below for generating SQBs in the calculation of ESLs.
- If "**no"** to either of the above, go to the Sediment Quality Criteria 3 section.

When a NAWQC or Tier I WQC are unavailable for calculating an SQC, SQBs are generated using Tier II secondary chronic values (SCVs) for water (see Section 4.4.1 for WQBs). SQB2s are calculated using the identical mathematical relationships of SQCs with the substitution of SCV for the FCV/CCC in Equation 6. The SQB2 method is also only appropriate for non-ionic organic compounds, and those with log(K_{ow}) values between 2.0 and 5.5 (EPA 1996b). Since both SQCs and SQBs are directly dependent upon K_{ow} values, reliable sources for this information are necessary (e.g., EPA 1995b, MacKay et al. 1992-1997). Criteria for ranking K_{ow} values from the primary literature are also provided in EPA (1995a).

SQB2 values are calculated as follows:

The equation for the SQC2 in units of mg/kg (see * notation, below) is:

$$SQC2 = f_{oc} \times K_{oc} \times SCV \qquad \text{Equation 45}$$

where:

- f_{oc} (no units of mass) is the mass fraction of organic carbon for the sediment.
- K_{oc} (no units of mass) is the sediment organic carbon partitioning coefficient.

** Because freshwater toxicity information is considered for deriving sediment benchmarks, it is assumed that a liter of freshwater weighs a kilogram for purposes of unit conversion.*

Similar to SQB1 (Equation 44), we may now calculate the SQB2 utilizing the relationship of the SQC2 shown in Equation 46:

$$SQB2 = \left(f_{oc\,[\text{site specific}]} \Big/ f_{oc[\text{normalized}]} \right) \times SQC2 \qquad \text{Equation 46}$$

Jones et al. (1997) provide some SQBs for ionic organic compounds. As Jones et al. (1997) indicate, ionic organic compounds have not been well studied for their equilibrium partitioning properties in the water-sediment interface. The Oak Ridge National Laboratory (ORNL) benchmarks (Jones et al. 1997) are probably conservative, as the fraction of ionic organic compounds adsorbed to the organic carbon surfaces is likely to be greater than that for non-ionic substances. Other factors may also affect the sorption capacity of sediment for ionic compounds, including pH (Jafvert 1990).

➢ ***Sediment Quality Benchmark 3***

If an SQB1 or SQB2 cannot be calculated, values from EPA's "Calculation and Evaluation of Sediment Effect Concentrations (SEC) on the Amphipod *Hyalella azteca* and the Midge *Chironomus riparius*" should be used (EPA 1996a).

- Is the COPC a non-ionic organic compound for which NAWQC, Tier I and Tier II data are lacking?
- Is the COPC a metal?
- If "**yes"** for either of the above, follow the directions outlined below for the adoption of Sediment Effects Concentrations (SECs) as the SQB3.
- If "**no"** to either of the above, go to the "Other Potential SQB Resources" section.

SECs were derived by the National Biological Service in response to the needs of EPA for the development of SQBs for the Great Lakes. SECs were derived utilizing National Ocean and Atmospheric Administration's effects range low (ERL) and effects range median (ERM) methods, Florida Department of Environmental Protection's threshold effects level (TEL) and probable effects level (PEL) methods, and the State of Washington's apparent effects threshold (AET). The calculation of SECs is considered more robust than using a single benchmark, because multiple benchmarks are used to derive a single SEC value. Using combined TELs or ERLs as SECs minimizes the possibility of incorrectly classifying a toxic constituent in sediment

as nontoxic. Notable exceptions may apply on a case-by-case basis in screening and uncertainty analysis, and particular ERM, TEL, or AET values may be better suited as an SQB3, rather than their combined values. The following definitions apply to SEC derivation (EPA 1996a):

- ERL is the sediment COPC concentration at which 10% of the test population was observed with effects (similar to a TEL below).
- ERM is the sediment COPC concentration at which 50% of the test population was observed with effects (similar to a PEL below).
- TEL is the upper limit of the range of sediment COPC concentrations dominated by no effects data.
- PEL is the lower limit of the range of COPC concentrations that are usually or always associated with adverse biological effects.
- AET is the sediment chemical concentration above which statistically significant biological effects always occur.

The test organisms used for SEC derivation are the amphipod *H. azteca* and the midge *C. riparius*. Tests on these organisms have been conducted utilizing sediment samples from a large number of freshwater sites. Measurement endpoints have historically included reduction in survival, growth, or sexual maturation of *H. azteca* in both 14-day and 28-day tests, and reduction of survival or growth of *C. riparius* in 14-day tests.

H. azteca and *C. riparius* are widespread and common benthos over much of North America, including arid portions of the southwestern United States. Each organism is broadly representative of crustacean and insect invertebrates (respectively) that dominate lentic and lotic systems. These organisms are not considered tramp species, rather, they are part of many intact aquatic systems. Because they are ubiquitous and part of healthy aquatic systems, these organisms are considered to be adequate choices for broad-based protection of aquatic organisms at large. Additionally, the SEC project has undergone close scrutiny by EPA, the Natural Resource Trustees of the Great Lakes Systems, and Great Lakes System Stakeholders, and has been found to be adequate to serve as a model for freshwater systems nationwide (Jones et al. 1997). Jones et al. (1997) recommend SECs to be adopted as SQBs for organic COPCs not covered by the SQC or SQB1&2 methods, and for metals.

➢ ***Other Potential Sediment Quality Benchmark Resources***

SQBs adopted from sources other than those recommended in SQC 1 through 3 should be discussed for deficiencies of information in the uncertainty analysis.

If an SQB1, SQB2 or SEC cannot be derived, then the EPA Region IV's sediment screening values (EPA 1996c) may suffice as SQBs for COPCs in question. These values are based on sediment toxicity work performed on marine sediments (e.g., Long et al. 1995). Although data from studies of saltwater sediments may not seem relevant to freshwater sediments, these data have been consistently recommended by EPA (e.g., EPA 1996b and c). One study performed to assess compatibility of freshwater and marine sediment toxicity data indicates that correspondence between the two is very close for a broad range of potential toxicants (Klapow and Lewis 1979). Since in many cases the sample quantitation limit (SQL) for identifying

organic compounds was greater than the benchmark values, the benchmark value defaults to the SQL. In the case where the SQL is used as a default value, a justification should be provided for using a SQL as a benchmark value. If a SQL exceeds a known low-effects range value for any aquatic organism, then the constituent should be carried forward from numerical screening to the uncertainty analysis.

In the absence of any valid SQB resources, any available information on a given constituent (e.g., from primary literature sources) should be used. The biotic system being evaluated should be considered, as well as the range of concentrations over which there may be information on no effects or observed effects. In context with one another, the biotic system being evaluated and relative effects ranges considered may provide insight into the most appropriate SQB. The Ontario Ministry of the Environment sediment quality guidelines (Persaud et al. 1993) is an additional resource, and Jones et al. (1997) also provide references for other resources.

4.4.2.1 Summary of Sediment ESL Derivations

Sediment quality benchmarks (SQBs) that are used in screening-level risk assessment for aquatic receptors may be derived from a variety of sources. More than one SQB may be available for any given constituent and are employed in a hierarchical fashion (see Table 6). Potential bioaccumulation concerns for sediment are considered in the wildlife exposure model (Section 4.3.4.1).

Table 6. Summary of sources for sediment ESLs.

COPC Type	Aquatic Receptors	Bioaccumulation & Biomagnification to Terrestrial Vertebrates
Non-ionic organic chemicals K_{ow} in range of 2 to 5.5	The following are used in order of preference: 1. SQCs calculated from national ambient water-quality criteria (NAWQC)or from Great Lakes Tier I water-quality criteria (WQC) (EPA 1995a) according to EPA (1996b); 2. SQBs calculated from Great Lakes Tier II WQC (EPA 1995a) according to EPA (1996b); 3. Sediment Effects Concentrations (SECs) derived from EPA methods described below (EPA 1996a); 4. EPA Region IV screening values (EPA 1996c); 5. Jones et al. (1997), Long et al. (1995).	See the wildlife exposure model, Section 4.3.4.1.
Inorganic chemicals	The following are used in order of preference: 1. SEC (EPA 1996a) 2. EPA Region IV (EPA 1996c) 3. Other (e.g., Jones et al. 1997, Long et al. 1995)	

4.5 Radiological Contaminants

For radionuclides, toxicity data are not radionuclide–specific when expressed as dose limits (e.g., 0.1 rad/day). These dose limits can, however, be translated into radionuclide-specific concentrations (e.g., picocuries [pCi] per gram) for a defined exposure scenario, as detailed in DOE (2004): RESRAD-BIOTA. This SLERA recommends using Biota Concentration Guides (BCGs) developed by the U.S. Department of Energy's Biota Dose Assessment Committee DOE (2002), for the purpose of evaluating radiation as a stressor to biota and ecosystems. (A BCG is defined as the environmental concentration of a given radionuclide in soil or water that, under

the assumptions of the model, would result in a dose rate less than 1 rad/d (10 mGy/d) to aquatic animals or terrestrial plants or 0.1 rad/d (1 mGy/d) to terrestrial animals.) BCGs are pertinent to SLERA in that they provide useful evaluation systems and numerical values.

Because dose from radionuclides is additive, the contribution of radionuclides identified to be COPCs will be calculated. This calculation is based on the sum-of-fractions (SOF) method, and the contributions of various radionuclides were reviewed to determine their contribution to dose.

$$SOF = Exposure_j/BCG_j \qquad \text{Equation 47}$$

where,

- SOF = sum of fractions
- $Exposure_j$ is the exposure concentration for radionuclide *j*;
- BCG_j is the biota concentration guideline for radionuclide *j*.

If the SOF is ≤1, then no radionuclide COPECs will be identified. If the SOF is >1 then those radionuclides that contribute to the SOF are retained for further evaluation as COPECs. This additional evaluation may include additional tiers of analysis as described in the BDAC guidance documents.

5.0 Uncertainty Analysis

Many of the uncertainties for the calculation of screening values in various media and for various receptor groups has been discussed in sections of this work devoted to those topics. The uncertainty analysis, however, should be a coherent analysis that is performed following all screening calculations and comparisons of on-site chemical concentrations and distributions with background data, frequencies of detection, TRV derivation and selection, screening analyses and related calculations. Interpretations of the HQ and HI for receptor groups should be made in the context of the CSM, and a thorough discussion of uncertainties regarding exposure pathways should ensue. Additionally, the potential for organisms to encounter multiple contaminants in the environment should be discussed. Each of these aspects of uncertainty will be briefly considered in this section.

5.1 Chemical Concentrations and Distributions

Representative exposure concentration may be a difficult parameter to calculate, and depends largely on the distribution (heterogeneity, homogeneity) of a contaminant, as well as the form of the contaminant in the various media of consideration. Moreover, contaminants in the environment are not static in form or distribution through time and nearly all site analyses tend to treat data as a reflection of static form and not as a consideration of a dynamic conditions. The relative distribution of contaminants and their mobility in the environment, thus their proclivity to reach ecological receptors, should have been discussed thoroughly in the CSM, but may also enter the uncertainty analysis. In particular, the dynamics of soil and sediment turnover and movement, as well as groundwater and surface water movement, due to physical and biotic processes, is of key interest to understanding the potential for exposure of organisms to contaminants that are in a dynamic flux in the environment. The proclivity for distributional fluxes and the rates at which they may occur are of key importance for understanding the long-

term well-being of a contaminated site. Put another way, contaminants do not tend to stay where they are measured in time, and consideration of the potential for risk imposed by contaminants in the environment may largely be a time-dependent, dynamical process. One time-dependent process of key interest is contaminant movement off-site. Thus, one of the key uncertainties in a SLERA is the general lack of time-dependent considerations, which should be discussed in the uncertainty analysis.

In screening, the upper 95th percentile of the mean concentration or the maximum concentration for each COPC is calculated for the entire geographical area under consideration. The uncertainty analysis should include some consideration of use of the upper 95th percentile of the mean or maximum concentration parameters, particularly how those parameters reflect what may be found in the environment. Each of these parameters may overestimate the actual exposure concentration that organisms at large may encounter at a given site, especially if contaminants are relatively immobile and unevenly concentrated. Naturally, this *potential* overestimation is desirable for screening, as it minimizes the risk of missing a COPEC. However, organisms have many mechanisms for interacting with their physical surrounds that may increase or decrease their proclivity for contaminant exposure, and such behavioral and phenological mechanisms should be discussed, particularly for COPCs where screening analyses indicate that the concentrations are close to (either above or below) threshold values.

One special condition of significant consideration in uncertainty is if a COPC is distributed saltatorially or spottily in a given medium. In such cases, the most reasonable screening comparison is made with the maximum concentration of the COPC for any given medium. The reason for this is that organisms that may form a center or locus of activity at or near the site of maximum concentration are imposed the greatest risk from the COPC. It is precisely these conditions that a SLERA should protect against. COPCs that are distributed saltatorially or spottily in the environment should be of considerable concern in the uncertainty analysis.

Uncertainty associated with the representative exposure concentration should also be discussed, including consideration of the findings of the data review (e.g., precision and bias of sample results for environmental media samples, which precedes screening) and the impact of the review on the confidence in representative concentration estimates.

5.2 Background Comparisons

The development of background or reference site values is critical to an initial screen of COPC data for naturally occurring inorganic chemicals and some ubiquitous organic chemicals. The validity of comparison to naturally occurring site conditions must be established well in advance of a SLERA. Since the natural environment contains naturally varying levels of chemical constituents, calculated background values should be seen as simplistic averages. Since the typical value of comparison is the upper 95th percentile on the mean background (or reference site) value, this means that on average, organisms encounter values higher than the sampled mean less frequently than lower values of a given constituent in their environment. This consideration makes the background comparison conservative as a screening tool. Additionally, organisms that occur indigenously to any given site have typically adapted (behaviorally, developmentally, and reproductively) to natural background site conditions, unless the site has

been identified as a sink for a given biotic population, which is rare under natural conditions. This further serves to illustrate that the comparison of measured COPC values to background (or a validated reference site) is a conservative measure for screening, particularly since the comparison is with a constituent concentration representative of the upper 95th percentile of the mean (and not the maximum background concentration). However, these points also serve to illustrate the critical nature of the background or reference site development. Uncertainties from that development should be thoroughly propagated through the background comparison, and in the case where a statistically valid background (or reference site) datum or data set cannot be calculated, a background comparison should not be included in the SLERA. Such an uncertainty should be thoroughly discussed.

Due to the nature of TRV development (discussed below), ESLs for some COPCs may be calculated below background or reference site concentrations. If the representative concentration of a COPC is within the range of background concentrations, then the uncertainty analysis should address whether the representative concentration represents an elevated risk to that of background, or if it represents an exposure similar to background across the site, particularly given the full range of background values. For a SLERA, added risk can be expected if the concentration of a COPC is found to be above the 95th percentile of the mean value of background or reference site concentration. Thus, discussion in the uncertainty analysis should also concern the issue of whether the addition of a COPC to various on-site media has elevated site concentrations to a point that there is a potential for added risk.

5.3 Frequency of Detection

The frequency of COPC detection is directly correlated with the methods for sampling a site and the distribution of a given COPC on-site. The FOD analysis should only proceed if sampling of a given site is adequately in support of the CSM and the potential exposure pathways for ecological receptors deemed appropriate for site consideration.

Uncertainties regarding the FOD analysis primarily arise from adequacy of sampling and if sampling for a given COPC is adequate to detect the nature and extent of contamination. For example, if the nature of contamination for a given COPC is saltatory (spotty), then the FOD comparison is invalid for that COPC. Moreover, if sampling cannot adequately characterize the distribution of contamination for a given site, then an FOD analysis is not valid.

Given that the FOD analysis is valid, the minimum number of validated sampling points for a given COPC at *any single site* is twenty (see Section 2.1.1 for a full explanation). Sampling concerns for the FOD comparison and the resultant outcomes according to COPC distribution and concentrations should be thoroughly discussed in the uncertainty analysis.

5.4 TRV Selection and Development

Derivation of TRVs introduces uncertainty to the SLERA that were discussed in Section 4.2.1. Uncertainties can arise due to differences in test and target organisms; differences in exposure modalities of test and wild organisms; differences of chemical form utilized in tests vs. those found in site media; differences in the potential of exposure duration and frequency for test vs.

wild organisms; differences of the manifestation of effects, selected endpoints, and the measurement of effects; differences of the desired level of effects (NOAEL, LOAEL, or other LOE), and uncertainties of bounding effects conditions. These uncertainties must be fully elucidated and discussed.

Uncertainties in TRV derivation must be adequately offset in the final calculation of the TRV for each COPC and receptor or functional group such that potential risk of adverse effects from a given dose is minimal. Uncertainty factors (UFs) are used as penalties in the final determination of a TRV, based on the disparity of experimental conditions vs. those found in wild populations. In particular, the magnitude of UFs may be substantially influenced by experimental parameters such as the test exposure duration, test period, measure of effects (e.g., NOAEL, LOAEL, or LD, and whether the study was chronic, subchronic, or acute, etc.), and critical life stage (e.g., developmental, reproductive) of test organisms. Uncertainty is also generated by the lack of taxonomic and trophic similarity of the test organism compared to the species of concern for the SLERA, particularly those representing functional groups. A TRV based on the same primary toxicity study can vary by several orders of magnitude, depending on the approach used to characterize experimental uncertainty. Consistency in the application of UFs to the derivation of TRVs is critical for reliable development of TRVs in ecological risk assessment and should be fully documented. These uncertainties should be fully disclosed in the uncertainty analysis.

5.5 Screening Analyses

Media-specific screening calculations involve a tremendous number of assumptions. These include bioavailability of chemical constituents, the development of benchmarks or toxicity reference values (TRVs), development of concentration ratios (CRs), and the simplicity or idealization of modeling contaminant uptake by organisms.

The initial assumption of screening calculations is that the chemical form of any given COPC is the same in the environment, with the same qualities of bioavailability, as the chemical form used in toxicity studies that form the basis of the calculations. In general, toxicity studies use readily bioavailable forms of chemicals. The TRVs that are a direct result of these studies may, therefore, overestimate the bioavailability of the chemical form of a COPEC in the natural environment. Conversely, there is the potential that less bioavailable forms of a given chemical may have been employed for purposes of experimental study than those found as COPCs in the natural environment. These considerations should be made evident in discussion and reference of TRV development, and should also be considered in the uncertainty analysis.

The methods of chemical delivery in toxicological studies may also differ significantly from those considered in the ecological exposure models for organisms in a SLERA. As a consequence, media-specific toxicological thresholds for COPCs considered in the SLERA may differ significantly from those of toxicological studies. The uncertainties that arise from issues of calculating toxicological thresholds based on media-specific conveyance should be discussed in the uncertainty analysis.

Uncertainty factors (UFs) introduce an additional level of uncertainty in the calculation of TRVs and subsequent screening calculations. UFs are most often used to hedge conservatively in the

calculation of TRVs from various toxicological studies, and may be based on differences of taxonomic status between experimental and wild organisms, differences in trophic relationships, feeding habits, behavioral habits, primary medium of exposure, or any of myriad differences in experimental and natural conditions that may affect contaminant exposure conditions (see Section 4.2.1). Since UFs are themselves an expression of uncertainty, the uncertainty analysis should reflect upon the specific nature of the UFs and the rationale for their application in each and every case of TRV derivation.

Concentration ratio (CR) calculations are derived from studies of partial or whole organisms that have been subjected to a particular medium, concentration, and duration of chemical exposure. What ensues is measurement of the concentration of the chemical under study in the whole organism or tissues of particular concern. Calculation of CRs involves similar concerns as those for TRVs, thus the uncertainties of their derivations are either inadvertently or overtly propagated through subsequent screening calculations. Additionally, most toxicological studies do not take biomarkers (chemical byproducts of metabolic processes in organisms) into direct consideration while measuring endpoint concentrations of a chemical constituent. This is naturally more of a concern for organic chemicals than inorganic chemicals, but the underlying principle is that chemicals or elements of one form of delivery to an organism may have differing physiochemical forms that may alter the outcome of measurement involving bioaccumulation of that chemical or element. Thus, CRs for any chemical constituent should be chosen carefully and conservatively for the SLERA, and uncertainties regarding their derivation and selection should be made explicitly clear in the uncertainty analysis.

Simplicity or idealization of modeling contaminant uptake by organisms is clearly implicit in the derivation of HQ, HI, and/or ESL calculations. These models represent a balance between the totality of mathematical description of complex environmental and physiological processes and the necessary simplification required for models to have function in the context of the SLERA. More so than not, HQ, HI, and/or ESL models and benchmark derivations are designed to be conservative in the context that a delivery of a full dose of any given COPC to a target receptor is possible in the natural environment based solely on the medium of conveyance and the concentration of the constituent in the environment. This simplistic model does not account for many possible outcomes in the complex world of physical transport and fate of any given constituent, or the interactions that organisms have with the physical world and other organisms that populate their worlds. It is for these very reasons that HQ, HI, and/or ESLs and screening benchmarks are intended to identify the lowest common thresholds of potential COPC detriment based on concentration in a given medium in the environment.

An extension of modeling simplicity is the use of the functional group approach to parameterization of screening models. To provide conservative estimates of screening calculations, maximum intake rates (ingestion, imbibition, and inhalation as scaled to body weight) were used for each model, respectively. For the soil ingestion model by wildlife, the maximum proportionate consumption of dietary soil was used for each receptor considered. This approach maximized the scaled dose to the organism with the intention that the scaling is protective of all species within a feeding guild. Naturally, this approach may overestimate the potential risk to any single species within the functional group, but is conversely intended to provide maximum protection for *all* species assigned to the group. The potential overestimation

of risk associated with identification of a COPEC from any of the wildlife or bioaccumulation models should be considered in the context of conservative model assumptions.

An important aspect of screening model simplicity is that the potential for effects of bioconcentration and biomagnification are only minimally addressed in the SLERA. Chemicals that bioconcentrate are those that tend to reach levels of concentration in the tissues of organisms that exceed the concentrations found in media of contact in the natural environment. Bioconcentration is only minimally addressed in the selection of CRs. If a CR is greater than unity (>1.0), then the implication is that target organisms bioconcentrate the constituent. However, this modality of physiological sequestration is poorly documented for most organisms and most chemical constituents. Moreover, bioconcentration of biomarkers (metabolic byproducts of biotically sequestered contaminants) is even more rarely considered. Thus, the most reasonably conservative CR is always best to err with in screening calculations, but the implications of such a selection should be thoroughly discussed in the uncertainty analysis.

Modeling of biomagnification of chemical constituents is limited in the SLERA models to calculation of ESLs for wildlife that may assume contaminated sediment-based and water-based prey exposure. These models do not address the potential of biomagnification from soils, and although such concerns are typically of lesser consequence than biomagnification from contaminated sediments and free water, they are not without meritorious concern for certain chemical constituents, e.g., organochlorines that have low mobility in soil and low solubility, but are readily available to terrestrial organisms through soil uptake pathways. Additionally, biomagnification beyond the level of secondary consumer is not the least bit considered in the SLERA. If biomagnification concerns exist for a given site, then consideration of tertiary consumers, indigenous parasites and other natural organisms that are high in the trophic order, should be addressed in the uncertainty analysis. If the potential for impact to higher trophic order organisms cannot be ruled out for a given COPC, then the COPC should be considered a COPEC and carried beyond screening in the ecological risk assessment process. These concerns extend also to biomarkers that may also be biomagnified.

5.6 HQ and HI – Single and Multiple Contaminants

The HQ ratio is a simple index that is calculated in order to provide a measure whereby the exceedence of a TRV can readily be assessed. As mentioned in Section 4.2.3, the HQ is not a calculation of *risk,* which is a probabilistic measure of effect. An HQ>1, however, indicates a likely occurrence of ill effects for organisms that comprise a population that are subject to contaminant uptake in their routines of daily life at a given site, or in their interactions with that site. Naturally, since the HQ is the ratio of *potential* dose to a *potential* effects threshold, all of the uncertainties in the measures of *actual* dose (related to assumed dose calculations for a given receptor) and *actual* effects (given the uniqueness of species in a functional group) are propagated into the HQ calculation. It is thus evident that the HQ is a conservative measure of likely threshold exceedence. Uncertainties that arise in the calculations of dose and effects may be discussed in the context of the HQ in the uncertainty analysis.

The HI is a more uncertain measure than the HQ. The HI is intended to protect against the *additive* effects of contaminants, but functions poorly to account for potential synergistic and

antagonistic effects of contaminants and their physiological byproducts. The fundamental assumption of the HI is that if an HI is >1, then the additive stress from contaminants in an organism's diet exceeds that for a potential threshold of effects. However, there are few toxicological studies that directly measure potential additive, synergistic, or antagonistic effects. Therefore, a quantitative reduction of the minimum criteria for any HQ component of an HI (to <1.0 as a potential threshold effect) is completely arbitrary, and yet highly protective of risk potentiated by additive or synergistic effects of multiple contaminants. As the number of contaminants in an organism's environment increases, this arbitrary reduction in the threshold effects level becomes increasingly uncertain, as well as increasingly protective. Moreover, if at least one COPC yields an HQ of >1, then the potential thresholds of effects for all other COPCs may be substantially reduced. This latter potential is ultimately very demanding of the risk assessor in the evaluation of uncertainty in the HI analysis. All inclusions of COPCs in the list of COPECs, and particularly any reductions in the number of COPECs in the uncertainty analysis, should be considered in the context of uncertainties regarding additive and synergistic contaminant effects on ecological receptors.

5.7 Conceptual Exposure Models

Conceptual exposure models cover the primary pathways of contaminant exposure for organisms in terrestrial and aquatic realms. These models, however, are idealizations of the many overlapping pathways between significant ecological realms. Where division of realms results in the potential exclusion of a primary exposure modality for organisms that may be found at a given site, then it is the onus of the ecological risk assessor to bridge such gaps in information with a corrected conceptual exposure model and attending description, and follow on with the appropriate assessment of potentiated risk. In the case where the conceptual exposure models cannot adequately describe exposure modalities for certain receptor categories, particularly when compared with the functional foodchain categories, then uncertainties may arise over the adequacy of screening-level risk methods for identification of COPECs. In such cases, uncertainties that arise should be thoroughly exposited in the uncertainty analysis.

Uncertainty regarding the significance of exposure pathways that were not evaluated in screening calculations should be discussed in the uncertainty analysis, particularly those for foliar uptake of contaminants by plants, fugitive dust and particulate inhalation, and dermal exposure. If the potential exists for any organism in any functional group to suffer significant effects by these exposure pathways, then the appropriate exposure model should be developed for the SLERA. Full disclosure of the uncertainties associated with alternative exposure modalities should be discussed in the uncertainty analysis.

Many sites have multiple COPCs in multiple exposure pathways. Cumulative effects and contaminant interactions (synergies and antagonisms) may alter the threshold of toxicological effects for any or all COPCs. As mentioned, however, the ESL calculations are modeled on the assumption of the simple additive effects of chemicals. This assumption may overestimate or underestimate the actual impact of multiple contaminants from synergistic or antagonistic effects. Information is sparsely available for most chemicals regarding synergistic or antagonistic effects, therefore, discussion of these uncertainties are necessitated. The necessity of considerations of synergistic and antagonistic effects arises primarily when observations of

site conditions do not jibe with expectations of species presence and absence, species assembly, functional organization of organisms on site, and/or behavioral conditions of observed species on site. The necessity of familiarity of the ecological risk assessor with the ecosystem functionality of the site under consideration is underscored by the necessity to address uncertainties associated with ecological parameters as related to site conditions and exposure modalities, including organismal responses on scales spanning individuals to the ecological community at large. Expertise on the site's functional ecology should be consulted if these conditions cannot be adequately addressed in the uncertainty analysis. These associations are fundamental to adequate development, description, and implementation of the conceptual exposure models, as well as completion of the ecological risk analysis, interpretation and discussion.

Related to an understanding of site functionality is the observation that physical and biotic disturbance of a given site may or may not be related to site contamination. Adequate discussion of the site's history and natural history can aid in identifying conditions that arose from physical disturbance attributable to site operations or otherwise, and furthermore aid in understanding the potential for identifying ecological effects of exposure to contaminants. Discussion of the site's natural history characterization in the context of ecological impacts of site operations (including physical alterations of the site) is an invaluable tool for identifying forms of uncertainty associated with present-day effects of contaminant loads and distribution. This may be elaborated in the context of conceptual exposure models and discussed extensively in the uncertainty analysis.

5.8 Uncertainty's Epilogue

Examination of uncertainty in each step of the SLERA can result in adding or deleting COPCs from the list of COPECs. The underlying principle of this final outcome of the uncertainty analysis is that the protection of each species' populations on any given site is the most considerable matter of import for the SLERA. All additions and deletions of COPECs in the uncertainty analysis should be underscored by the tenet of population-wide protection from conditions historically arisen from site operations and contamination.

6.0 Risk Interpretation

At the completion of the screening evaluation, the risk assessor communicates the results to the risk manager, with an emphasis on the uncertainty analysis. The purpose of this communication is to provide the risk manager with sufficient information to support a risk management decision in consideration of potential ecological concerns. Sufficient information to identify a risk management strategy for ecological concerns arises from detailed and lucid examination and exposition employed in implementing the SLERA methods outlined in preceding sections of this guidance.

Recommendations of interim actions and/or a path forward in risk analysis may also be appropriate to make in the final interpretation of the SLERA. Some of the recommendations and risk management strategies that could result from the screening assessment include but are not limited to:

1. There is adequate information to conclude that the ecological risks are negligible and no additional investigation of ecological risk is recommended. For example, no thresholds are exceeded if screening-level benchmarks are not exceeded and if the HQ/HI analyses show no compelling reason to consider a COPC a COPEC. Additionally, the uncertainty analysis must indicate that there is no compelling reason to consider COPCs as COPECs for any functional group.
2. Ecological risks are not negligible, but the information is insufficient to indicate that adverse ecological effects are occurring. The recommended that must attend ambivalent conclusions is to move to a baseline ecological risk assessment.
3. There are sufficient lines of evidence to document potential or actual adverse ecological effects. Although screening analyses cannot fully delineate that level of concern or risk associated with adverse ecological effects, implications of such compel the recommendation of further investigation and/or interim action.
4. There is not adequate information to make a risk management decision due to high levels of uncertainty or incomplete information. In this case, data needs are identified based on the results of the initial screening, and a plan to collect additional data should be guided by the necessities of a complete screening analysis.

The final presentation of the SLERA should be thoroughly documented from data analysis in preparation of the ecological risk evaluation through the uncertainty analysis and risk interpretation. Appropriate references cited should be complete and documentation of variation from the above protocol (which requires prior NDEP approval) should be fully exposited. Documentation summarizing the SLERA should provide a basis on which a scientifically-based management decision can be made, such that further ecological risk analysis can be pursued or decision makers can proceed with plans for site cleanup, restoration and/or attenuation.

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Appendix A

Ecological Scoping Checklist

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A-1.0 Part A - Scoping Meeting Documentation

Site ID	
1a) Form of site releases (solid, liquid, vapor). Describe all relevant known or suspected mechanisms of release (spills, dumping, material disposal, outfall, explosive testing, etc.) and describe potential areas of release. Reference locations on a map as appropriate.	
1b) Primary Impacted Media. Indicate all that apply.	Surface soil Surface water/sediment Subsurface Groundwater Other, explain
1c) Vegetation class Indicate all that apply and list approximate percentages of site. Provide maps clearly delineating these areas under current and post-closure conditions.	Grassland/shrubland (creosote-white bursage) Desert riparian Tamarisk Bare Ground/Unvegetated Developed/Industrial or slated for development Water
1d) Is T&E Habitat Present? If applicable, list species known or suspected to use the site for breeding or foraging.	Obtain written documentation from the U.S. Fish and Wildlife Service and the Nevada Natural Heritage Program of the (Nevada) Department of Conservation and Natural Resources
1e) Neighboring/ Contiguous/ Upgradient sites of concern, includes a brief summary of COPCs and the form of releases for relevant sites and reference a map as appropriate. (Use this information to evaluate the need to aggregate sites for screening.)	
1f) Surface Water Erosion Potential Summarize conditions for potential runoff and qualitatively assess and justify low, moderate and high runoff potential under current and post-closure conditions; indicate terminal point of surface water transport; slope; and surface water run-on sources.	Subarea: ▪ Vegetation class: ▪ Run-on source: ▪ Percent slope: ▪ Runoff terminus:
1g) Other Scoping Meeting Notes	

A-2.0 Part B - Site Visit Documentation

Site ID	
Date of Site Visit	
Site Visit Conducted by	

Receptor Information:

2a) Estimate cover	**Relative vegetative cover (high, medium, low, none):** • **Relative wetland cover (high, medium, low, none):** • **Relative structures/asphalt, etc. cover (high, medium, low, none):** •
2b) Field notes on vegetation class	
2c) Field notes on T&E Habitat (if applicable). Consider the need for a site visit by a T&E subject matter expert to support the use of the site by T&E receptors.	
2d) Are ecological receptors present at the site? (yes/no/uncertain) Describe the general types of receptors (terrestrial and aquatic), and make notes on the quality of habitat present at the site.	

Contaminant Transport Information:

2e) Surface water transport. Field notes on the erosion potential, including a discussion of the terminal point of surface water transport (if applicable).	
2f) Are there **Off-site transport pathways** (surface water, air, or groundwater)? (yes/no/uncertain). Provide explanation.	
2g) Interim action needed to limit off-site transport? (yes/no/uncertain) Provide explanation or recommendation for IA to project leader.	

Ecological Effects Information:

2h) Physical Disturbance. (Provide list of major types of disturbances, including erosion and construction activities, review historical aerial photos if appropriate.)	
2i) Are there **obvious ecological effects?** (yes/no/uncertain) Provide explanation and apparent cause (e.g., contamination, physical disturbance, other).	
2j) Interim action needed to	

limit apparent ecological effects? (yes/no/uncertain) Provide explanation and recommendations to project leader for IA to mitigate apparent exposure pathways.	

No Exposure/Transport Pathways:

2k) If there are **no complete exposure pathways to ecological receptors on-site and no transport pathways to off-site receptors**, the remainder of the checklist should not be completed. Stop here and provide additional explanation/justification for proposing an ecological No Further Action recommendation (if needed). At a minimum, the potential for future transport should include likelihood that future construction activities could make contamination more available for exposure or transport.

Adequacy of Site Characterization:

2l) Do existing or proposed **data** provide information on the **nature, rate and extent of contamination?** (yes/no/uncertain) Provide explanation (Consider if the maximum value was captured by existing sample data).	
2m) Do existing or proposed **data** for the site address potential **transport pathways of site contamination?** (yes/no/uncertain) Provide explanation (Consider if other sites should be aggregated to characterize potential ecological risk).	

Additional Field Notes:

2n) Provide additional field notes on the site setting and potential ecological receptors.

A-3.0 Part C - Ecological Pathways Conceptual Exposure Model

Provide answers to Questions A to V to develop the Ecological Pathways Conceptual Exposure Model

General Site Contaminant Transport Pathways (see Figures A-1 and A-2)

Question A:

Could soil contaminants reach receptors via vapors?

- Volatility of the hazardous substance (volatile chemicals generally have Henry's Law constant $>10^{-5}$ atm-me/mol and molecular weight <200 g/mol).

Answer (likely/unlikely/uncertain):

Provide explanation:

Question B:

Could the soil contaminants reach receptors through fugitive dust carried in air?

- Soil contamination would have to be on the actual surface of the soil to become available for dust.
- In the case of dust exposures to burrowing animals, the contamination would have to occur in the depth interval where these burrows occur.

Answer (likely/unlikely/uncertain):

Provide explanation:

Question C:

Can contaminated soil be transported to aquatic ecological communities? Consider potential for erosion and pathways for surface water transport.

- Consider sources of run-on and how that may affect flow across the site and leaving the site.
- If erosion is a transport pathway, evaluate the terminal point to see if aquatic receptors could be affected by contamination from this site.

Answer (likely/unlikely/uncertain):

Provide explanation:

Question D:

Is contaminated groundwater potentially available to biological receptors through seeps or springs or shallow groundwater?

Known or suspected presence of contaminants in groundwater.

- The potential for contaminants to migrate via groundwater and discharge into habitats and/or surface waters.
- Contaminants may be taken up by terrestrial and rooted aquatic plants whose roots are in contact with groundwater present within the root zone (~1 m depth).
- Terrestrial wildlife receptors generally will not contact groundwater unless it is discharged to the surface.

Answer (likely/unlikely/uncertain):

Provide explanation:

Question E:

Is infiltration/percolation from contaminated subsurface material a viable transport and exposure pathway?

- Suspected ability of contaminants to migrate to groundwater.
- The potential for contaminants to migrate via groundwater and discharge into habitats and/or surface waters.
- Contaminants may be taken up by terrestrial and rooted aquatic plants whose roots are in contact with groundwater present within the root zone (~1 m depth).
- Terrestrial wildlife receptors generally will not contact groundwater unless it is discharged to the surface.

Answer (likely/unlikely/uncertain):

Provide explanation:

Question F:

Might events that induce erosion be a potential release mechanism for contaminants from subsurface materials or perched aquifers to the surface?

- Consider aquifers daylighting to the Las Vegas wash.
- Consider the erodability of surface material and the processes of eroding stream banks and other embankments.
- Consider the release of eroded materials directly into a surface water body.

Answer (likely/unlikely/uncertain):

Provide explanation:

<u>Contaminant Pathways to Terrestrial Receptors (see Figure A-1)</u>

Question G:

Could airborne contaminants interact with receptors through respiration of vapors or by means of foliar contact?

- Contaminants must be present as volatiles or gasses in the air.
- Consider the importance of inhalation of vapors and gasses for burrowing animals.
- Foliar uptake of organic vapors is typically not a significant exposure pathway, but uptake/contact of/with inorganic gasses , e.g., ozone, may have strongly detrimental effects on plant tissues.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Plants:
Terrestrial Animals:

Provide explanation:

Question H:

Could airborne contaminants interact with plants through deposition of particulates or with animals through inhalation of fugitive dust?

- Contaminants must be present as particulates in the air or as dust for this exposure pathway to be complete.
- Some contaminants posses erosive qualities that may damage vegetation or be absorbed by foliar tissues, e.g. strongly acidic or basic compounds, or boron-based and herbicidal compounds.
- Exposure via inhalation of fugitive dust is particularly applicable to ground-dwelling species that would be exposed to dust disturbed by their foraging or burrowing activities or by wind movement.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Plants:
Terrestrial Animals:

Provide explanation:

Question I:

Could contaminants interact with plants through root uptake or rain splash from surface soils?

- Contaminants in bulk soil may partition into soil solution, making them available to roots.
- Certain volatile organics may have strongly detrimental effects on plant rooting systems, e.g., methane.
- Exposure of terrestrial plants to contaminants present in particulates deposited on leaf and stem surfaces by rain striking contaminated soils (i.e., rain splash).

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Plants:

Provide explanation:

Question J:

Could contaminants interact with receptors through food web transport from surface soils?

- The chemicals may bioaccumulate in animals.
- Animals may ingest contaminated food items.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Animals:

Provide explanation:

Question K:

Could contaminants interact with receptors via incidental ingestion of surface soils?

- Incidental ingestion of contaminated soil could occur while animals grub for food resident in the soil, feed on plant matter covered with contaminated soil or while grooming themselves clean of soil. Natural soil ingestion rates for potential receptors should be considered. (Consider ingestion rates from EPA's Wildlife Exposure Handbook or other sources for estimating ingestion rates.)

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Animals:

Provide explanation:

Question L:

Could contaminants interact with receptors through dermal contact with surface soils?

- Significant exposure via dermal contact would generally be limited to organic contaminants that are lipophilic and can cross epidermal barriers.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Animals:

Provide explanation:

Question M:

Could contaminants interact with plants or animals through external irradiation?

- External irradiation effects are most relevant for gamma emitting radionuclides.
- Burial of contamination attenuates radiological exposure.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Plants:
Terrestrial Animals:

Provide explanation:

Terrestrial Water Pathways (see Figure A-1)

Question N:

Could contaminants interact with plants through direct uptake from water and sediment or sediment rain splash?

- Contaminants may be taken-up by terrestrial plants whose roots are in contact with surface waters.
- Terrestrial plants may be exposed to particulates deposited on leaf and stem surfaces by rain striking contaminated sediments (i.e., rain splash) in an area that is only periodically inundated with water.

- Contaminants in sediment may partition into soil solution, making them available to roots.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Plants:

Provide explanation:

Question O:

Could contaminants interact with animal receptors through food web transport from water and sediment?

- The chemicals may bioconcentrate in food items.
- Animals may ingest contaminated food items.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Animals:

Provide explanation:

Question P:

Could contaminants interact with animal receptors via ingestion of water and suspended sediments?

- If sediments are present in an area that is only periodically inundated with water, terrestrial receptors may incidentally ingest sediments.
- Terrestrial receptors may ingest water-borne contaminants if contaminated surface waters are used as a drinking water source.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Animals:

Provide explanation:

Question Q:

Could contaminants interact with animal receptors through dermal contact with water and sediment?

- If sediments are present in an area that is only periodically inundated with water, terrestrial species may be dermally exposed during dry periods.
- Terrestrial organisms may be dermally exposed to water-borne contaminants as a result of wading or swimming in contaminated waters.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Animals:

Provide explanation:

Question R:

Could contaminants interact with plants or animals through external irradiation?

- External irradiation effects are most relevant for gamma emitting radionuclides.
- Burial of contamination attenuates radiological exposure.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Terrestrial Plants:
Terrestrial Animals:

Provide explanation:

Aquatic Pathways (see Figure A-2)

Question S:

Could contaminants bioconcentrate in free-floating aquatic, attached aquatic plants, or emergent vegetation?

- Aquatic plants are in direct contact with water.
- Contaminants in sediment may partition into pore water, making them available to submerged roots.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Aquatic Plants/Emergent Vegetation:

Provide explanation:

Question T:

Could contaminants bioconcentrate in sedimentary or water column animals?

- Aquatic receptors may actively or incidentally ingest sediment while foraging.
- Aquatic receptors may be directly exposed to contaminated sediments or may be exposed to contaminants through osmotic exchange, respiration, or ventilation of sediment pore waters.
- Aquatic receptors may be exposed through osmotic exchange, respiration, or ventilation of surface waters.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Aquatic Animals:

Provide explanation:

Question U:

Could contaminants bioaccumulate in sedimentary or water column animals?

- Lipophilic organic contaminants and some metals may concentrate in an organism's tissues.
- Ingestion of contaminated food items may result in contaminant bioaccumulation through the food web.

Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Aquatic Animals:

Provide explanation:

Question V:

Could contaminants interact with aquatic plants or animals through external irradiation?

External irradiation effects are most relevant for gamma emitting radionuclides. The water column acts to absorb radiation, thus external irradiation is typically more important for sediment dwelling organisms.

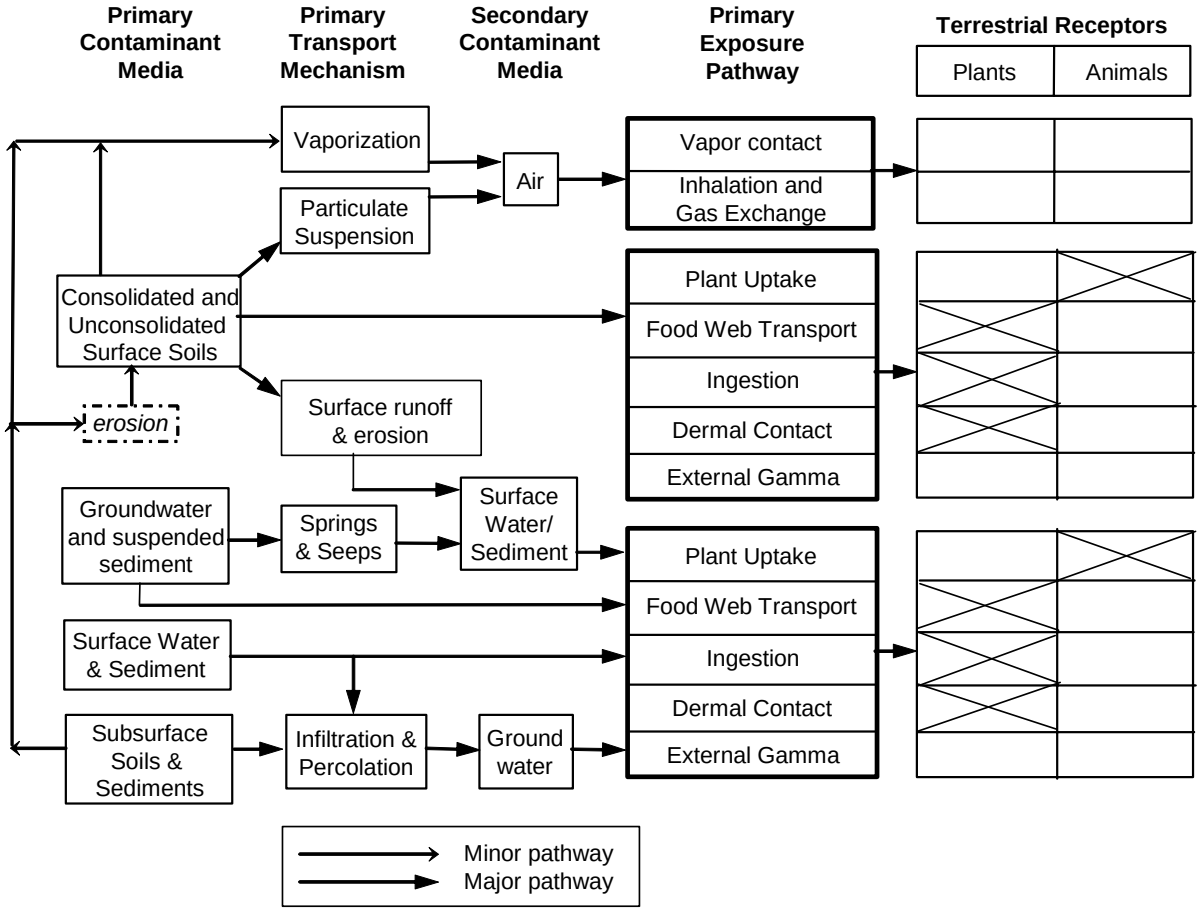
Provide quantification of exposure pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway):

Aquatic Plants:

Aquatic Animals:

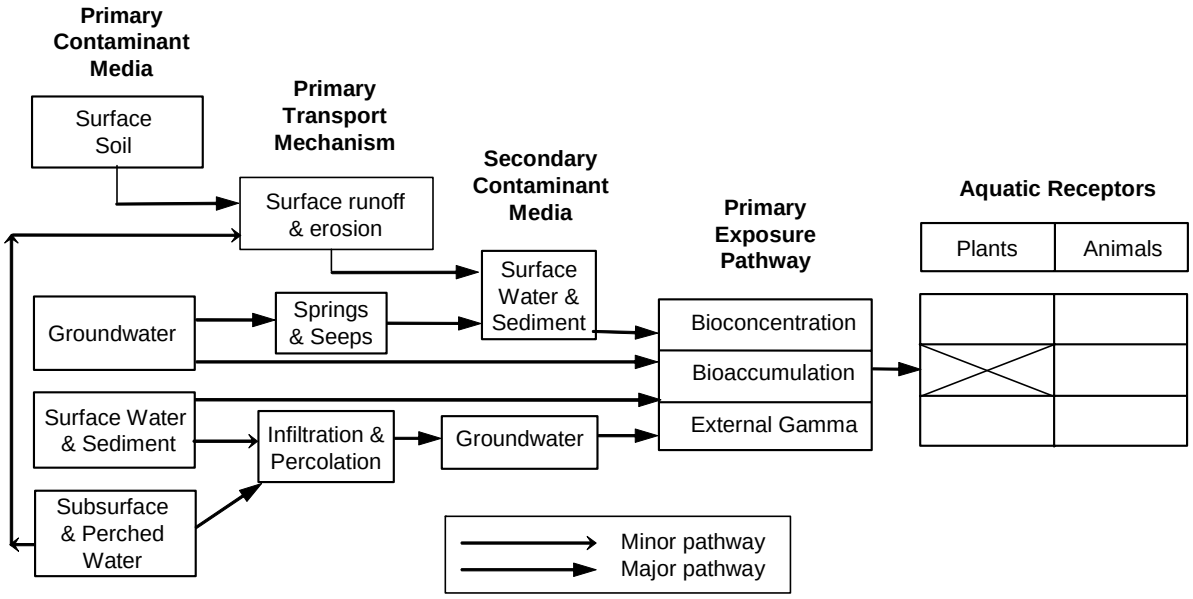
Provide explanation:

Figure A-1. Ecological pathways conceptual exposure model for terrestrial receptors.



Boxes marked with "X"s indicate incomplete pathways. Open boxes indicate potentially complete pathways.

Figure A-2. Ecological pathways conceptual exposure model for aquatic receptors.



Boxes marked with "X"s indicate incomplete pathways. Open boxes indicate potentially complete pathways.

A-4.0 Part D - Signatures and certifications

Checklist completed by:

Name (printed):	
Name (signature):	
Organization:	
Phone number:	

Date completed:	

Verification:

Name (printed):	
Name (signature):	
Organization:	
Phone number:	

Appendix B

General Assessment Endpoints

for the

BMI Complex
and
Affected Areas

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B-1.0 Introduction to GAE Process and Application

An ecological risk assessment must specify assessment endpoints in order for there to be a foundation for a risk-based decision framework. However, selecting assessment endpoints for risk assessment is often a formidable task due to the richness of the biota at any given site, and the complexity of ecological interactions on multiple geographic and temporal scales. Ultimately, assessment endpoints chosen for use in the risk assessment must be representative of the site's biota, but limited to those that provide protective coverage for all biotic organisms and processes relevant to the site. Moreover, the chosen assessment endpoints must reflect the values assigned to the site by stakeholders, including the regulatory community. Therefore, the selection of assessment endpoints is derived from sound ecological reasoning, as well as management policy, goals, and objectives that pertain to a given site. It is the intent of this Appendix to lay the general foundation for endpoint selection so that the ultimate delineation of assessment endpoints, as specified in the SLERA document for the BMI Complex and affected areas (hereinafter generally referred to as the "BMI Complex"), naturally follow.

A general foundation is provided herein that provides a framework of General Assessment Endpoints (GAEs), which reflect ecological values of broad significance to risk managers and other stakeholders. The approach of this document is based on that of Reagan et al. 1999 for "General Assessment Endpoints" and EAP (2003b) for "Generic Ecological Assessment Endpoints." This Appendix provides an introduction to the GAE process, describes the GAEs developed for the BMI Complex, and provides guidelines for identifying assessment endpoints in the context of the GAE framework.

In order to define GAEs, it is useful to begin with definitions of assessment endpoints as outlined by the EPA (1997, 1998). The EPA defines an assessment endpoint as *"an explicit expression of the environmental values that are to be protected."* EPA (1998) also remarks that assessment endpoints are *"operationally defined by an ecological entity and its attributes."* An ecological entity in this context is most often considered a biological organism, and its attributes would be measurable aspects of its survival and reproduction. As EPA (2003) states, assessment endpoints must meet "three selection criteria: ecological relevance, susceptibility (exposure plus sensitivity), and relevance to management goals."

GAEs are intended to move the ecological risk assessor from the general ecological and management relevance to the selection of assessment endpoints that meet the conditions of relevance but also provide a direct basis on which to measure implied risk. Since GAEs include managerial constructs, they are intended to encompass *ecological and human use values* at all levels of ecological organization (ecosystems, communities, and individual species). The GAE process provides a comprehensive, systematic and defensible basis for reaching consensus among risk assessors, regulators and other stakeholders in the selection of assessment endpoints for ecological risk assessments.

B-2.0 Overview of the GAE Process

The process of identifying GAEs occurs in two parts. First, ecologically relevant values are identified for the system under consideration, and second, human values associated with the

ecological resources under evaluation are identified. The GAE process is based on the assumption that the ultimate ecological value under consideration is a healthy, sustainable ecosystem. Ecological relevance, therefore, refers to the properties necessary for unimpaired ecosystem function.

The ecological evaluation begins with the identification of characteristics and processes integrally important, yet common to all ecosystems. This evaluation progresses to a consideration of the particular ecosystem present at the specific location under investigation. This progression provides a hierarchical and objective means of determining which components of the ecosystem are potentially relevant to the assessment of ecological risk. This process consists of five steps.

1. Ecological values, common to all ecosystems, are identified (Section B-3.1).
2. Functional components of the specific ecosystem are identified (Section B-3.2.1).
3. Functional food webs of the ecosystem are developed (often done concomitantly with step 2) (Section B-3.2.1.1).
4. Attributes of the functional components of the ecosystem are determined (Section B-3.2.2).
5. Ecologically relevant GAEs are described (Section B-3.3).

Once ecologically relevant GAEs have been determined, ecological values relevant to societal values and/or management goals are identified to supplement GAEs. Section B-4.0 describes societal values that are relevant to the BMI Complex and surrounds. The content of this section reflects the consensus opinion found broadly among various land management agencies. However, this consensus opinion does not reflect a promulgated position of NDEP, it merely reflects the ideas involved in the development of this document.

B-3.0 GAEs Based on Ecological Relevance

The BMI Complex and its immediate surrounds are contiguous with is the northern Mojave Desert ecosystem. This ecosystem is complex and is represented by a wide array of plant communities, soil types, and topographic features (Rundel and Gibson 1996). The BMI Complex is predominantly considered a creosote bush-white bursage habitat *(Larrea tridentata-Ambrosia dumosa)* in various stages of maturity or disturbance (established vs. emerging communities). The Complex is contiguous with a significant Mojave Desert riparian community (the Kerr-McGee Seep and Las Vegas Wash). Transitional (ecotonal) areas that connect the creosote bush-white bursage habitat with riparian areas are currently dominated by Chinese tamarisk *(Tamarix chinensis)*, which is a non-native weedy species, but with inclusions of native vegetation. In addition, the BMI Complex includes a great deal of disturbed, barren ground that has not been reclaimed from prior industrial use. Areas of urban land cover are also present.

Sustaining a healthy ecosystem is the ultimate ecological value to protect; however, to achieve this goal, a variety of ecological values must be considered and protected. The process of identifying these values, beginning at the ecosystem level and progressing to lower levels of ecological organization is described in the following sections.

B-3.1 Values Common to All Ecosystems

Recognizing that assessment endpoints are defined as values to be protected (EPA 1997, EPA 1998), the approach to developing GAEs begins by identifying values common to all ecosystems at the highest level possible: the value of preserving a healthy and sustainable ecosystem. De Leo and Levin (1997) prefer the notion of ecological integrity rather than ecological health, as they feel that integrity includes the concept of valuations that are based on human use, which they believe is the appropriate value structure for environmental management decisions. Recognizing that ecological values are ultimately human values (Harwell et al. 1994), the terms "ecological health" and 'ecological integrity" or "intactness" are employed interchangeably. A healthy ecosystem is defined to be one that contains all essential functional components and interactions, which are uncompromised by human activities and intervention, past and present, and operate at levels typical of that type of ecosystem.

There are a number of characteristics that are seminal to the healthy state and function of an ecosystem. Following the GAE approach, characteristics have been organized in this Appendix into three separate, but interrelated, attributes common to all ecosystems: (1) biological diversity, (2) functional integrity, and (3) nutrient and energy dynamics. While these attributes do not stand alone from one another and can be considered in various combinations (e.g., functional integrity can be defined to encompass both biodiversity and process dynamics), considering them one at a time allows one to look at the components, patterns of organization, and process rates somewhat independently.

In the sections that follow, the attributes common to all ecosystems are defined and discussed in the context of why they are valued and how they are related to the goal of preserving a healthy and sustainable ecosystem.

B-3.1.1 Biological Diversity (Biodiversity)

A simple definition of biological diversity is "the number of species in a community." The more species, the greater the biological diversity. However, biological diversity described in this way misses much that is relevant to why biodiversity is valued (De Leo and Levin 1997), hence why the maintenance of biological diversity is a foundational GAE. This aspect of biological diversity will be discussed further below.

Biological diversity is valued from a human perspective for multiple reasons. These include the value of extractable resources, land use, aesthetic value, value of rarity, the value of undiscovered natural products of potential benefit to human health, and the indirect value of the processes performed by diverse assemblages of species (e.g., nutrient cycling, erosion control, cleansing of water and air).

Moreover, biologically diverse systems in temperate regions of the world may be generally more resilient to natural and anthropogenic perturbations and changes than less diverse systems (De Leo and Levin 1997). Maintaining diversity can be important for maintaining the structure and function of the system. In biologically diverse systems we often find multiple species within a particular functional group, or guild. To the extent that these species perform the same ecological function, they provide functional redundancy. Functional redundancy has been shown to play an

important role in maintaining an ecosystem's ability to respond to change (De Leo and Levin 1997). The maintenance of biological diversity is recognized as an important factor that keeps the natural habitats habitable and functional for indigenous biota, as well as humans.

When attempting to measure biological diversity, it is important to carefully delineate the geographical and temporal domain prior to taking any measurements, and then accurately identify species and the variation within species that are present within these bounds. There are several broadly useful approaches to defining biological diversity, including (but not limited to) assemblage diversity, genetic diversity, and phenotypic diversity, as outlined below.

- ***Assemblage diversity***

Biodiversity is most often defined in terms of species richness (number of species) and evenness (relative abundance of species) in a given area at a given time. In order to evade confusion over the breadth of definitions for biological diversity, this form of diversity is referred to here as *assemblage diversity.* This definition has led to many attempts at the quantification and indexing of biological diversity, all of which have evident shortcomings (Magurran 1988). However, the simplest and most constructive way to consider and quantify assemblage diversity, is to simply count the number of species (species richness) in a geographically and temporally defined space (or alternately, at several scales of interest), while simultaneously measuring the relative abundance of each species (species evenness). These are perhaps the simplest measures of "biological diversity" and are applicable in many managerial practices. Assemblage diversity often forms the basis for measuring biological diversity in the common practice of defining assessment endpoints for ecological risk assessment.

Assemblage diversity changes through time and across geography. There have been many attempts to characterize assemblage diversity on landscape levels (i.e. across geographic expanses that exceed the range of one or more species in an assemblage). Most of the landscape-level measures of assemblage diversity are characterized with respect to the functional relationships (roles, niche space, and trophic position) of organisms in and among biotic communities. These measures include the assemblage diversity and the particular species that comprise the assemblage. Such measures are often useful when considering expectation for the presence or absence of particular species in a community, the replacement of species by others that provide the same function across communities, and the relative abundance of these species, given the constraints of the community dynamics. This form of assemblage diversity (often coined gamma diversity) can be used as a measure of functional redundancy between community types or between ecosystems. For example, a community in one geographic locale may have an equivalent assemblage diversity and functional redundancy within guilds, to another, very different community in a geographically distinct place (even though these may be very close to one another in terms of distance). The geographic realms of this type of diversity are arbitrary; e.g. north-facing slopes vs. south-facing bajada slopes at the edges of desert valleys, or riparian communities of the desert washes of the Northern Mojave and southern Mojave Desert. This measure may be useful for assessing biodiversity of "reference communities" (communities that serve as a benchmark for measurement).

Communities that are more diverse are not necessarily more relevant to GAE development than less diverse communities. Although many different assemblage diversity indices have been developed and used, ecologists recognize a variety of measures are needed to capture the essence of assemblage diversity (Magurran 1988).

- ***Genetic diversity***

Genetic diversity is most often measured in terms of diversity of "genotype" of a given organism in geographically and temporally bounded environs. This is a rather precise and complex measure, and is not usually considered in ecological risk assessment, unless there is a special case, e.g. an endangered species at stake or a unique population at risk. However, the maintenance of genetic diversity may be at the crux of an ecosystem's ability to sustain perturbation (e.g. influx of contamination). Often, a species or population can sustain the impact of strong selection (a strong perturbation) in the near-term only because of the genetic basis for resistance to the selective force (perturbation). If more than one perturbation impacts a population under conditions of reduced genetic basis for population resilience, then a population may not be able to recover. For example, Clements (1997) and co-researchers (NIEHS/EPA 1999) have found that communities of benthic insects in Colorado streams are no less diverse, in terms of species composition, in streams polluted by heavy metals, than in similar streams that are relatively unimpacted. These researchers have also found that the genetic diversity of the insect populations studied was far less in polluted vs. unpolluted streams. The reduced genetic diversity, observed by these researchers, may put these populations at a much greater risk to extirpation due to natural (or other) perturbation (e.g. drought, disease) than the more genetically diverse populations. Therefore, in order to minimize the implicit impact to biotic populations from anthropogenic disturbance, it is important to minimize disturbances that reduce genetic diversity, and attempt to maintain genetically diverse populations.

- ***Phenotypic diversity***

Phenotypic diversity, i.e. variation of ecological type, morph, or form, is often recognized as a morphological expression of a genetic basis of diversity within species, and can therefore be viewed as an expression of the genetic diversity, as discussed above. Phenotypic diversity is dependent on many factors, but is relevant to a species only with respect to traits that are adaptive, and therefore confer selective advantage to individuals under the biotic and abiotic conditions in which the organisms carry out phenologic (life history) events. Under appropriate circumstances, phenotypic diversity, given the attendant conditions of a biotic community, may be a useful surrogate for the measurement of genetic diversity. Therefore, in order to minimize the implicit impact to biotic populations from anthropogenic disturbance, it is important to minimize disturbances that reduce phenotypic diversity.

B-3.2.1 Functional Integrity

Ecosystem integrity was defined by Karr and Dudley (1981, as quoted by DeLeo and Levin 1997) as, "The capability of [an ecosystem] to support and maintain a balanced, integrated, adaptive, community of organisms having species composition, diversity and functional organization comparable to that of natural habitats in the region." Functional integrity may be

more narrowly defined as the pattern of interactions among components of the ecosystem. This allows one to discriminate between species composition in the ecosystem (e.g., biodiversity) and the functional interactions among components. Thus patterns in a community or ecosystem may be distinguished, such as trophic structure or habitat relationships among specific species or functional guilds, in addition to evaluating biological diversity. In practice, to assess functional integrity, factors such as food chain length, connectivity, degree of omnivory, extent of reciprocal predation (food loops), and subweb organization can be evaluated. (Pimm 1982, Reagan et al. 1996, Schindler et al. 1985, Waide 1991).

Functional integrity is a valued attribute because it connotes an intact system, i.e., one in which there is no missing link that would result in structural or functional imbalances that render the entire system more vulnerable (less resilient) to perturbation. Understanding changes in trophic structures can also elucidate the mechanism for changes in process rates. For example, the loss of functional integrity might appear as the accumulation of detritus, shifts in the relative abundance (evenness) of species, or the disappearance or replacement of species in an assemblage.

Measures of interaction among species, according to principles of organization applicable to that system, may be more subtle than the measures for assessing functional integrity, mentioned above, but may be equally important for recognizing shifts in the functional integrity of the system. For example, sub-lethal doses of contaminants can alter key ecological processes (predator prey relationships, competition, ability to take up nutrients, organismal behavior, etc.), but may go unnoticed due to the coarseness of measurement. These measures vary with scales of biotic relevance, geography and time.

B-3.1.3 Energy and Nutrient Dynamics

The flow rates and patterns of nutrient and energy processing in a given ecosystem are critical for maintaining populations of indigenous species at levels characteristic of that ecosystem. Disruption of nutrient and energy flow rates (e.g. by nutrient enrichment or chemical contamination) can lead to accumulation of detritus, reduction of primary productivity, or loss of top predators (McNaugton 1978). Each of these changes could affect ecosystem structure, function, and intactness. The qualities of biodiversity, functional integrity, and nutrient and energy dynamics are essential ecological values across all ecosystems and provide a framework for the organization of consideration of ecological values across a breadth of ecosystem types. Furthermore, these properties offer a structure for considering the intact nature of an ecosystem, at all scales of ecological organization. The values (GAEs) identified in the following sections are founded on the vision of an intact ecosystem.

B-3.2 Values Common to the Northern Mojave Desert Ecosystem

In the GAE process, ecological values common to the regional ecosystem are identified next. These values are identified through a systematic process that includes first identifying the principal functional components of the regional ecosystem. Functional components are identified using food webs based on feeding guilds. A table associating attributes with the

functional components is then developed. The attribute table provides the ecological values common to the regional ecosystem and is the basis for identifying the regional GAEs.

B-3.2.1 Functional Components of the Northern Mojave Desert Ecosystem

Because food webs provide essential structural organization of producer-consumer relationships in ecosystems (Gallopin 1972) and because all organisms in an ecosystem are part of the food web, food webs are used to identify basic functional components of an ecosystem.

Food webs are typically composed of three basic trophic categories. These categories are (1) producers, (2) consumers, and (3) decomposers (which are a special category of consumer). The following definitions aptly fit these broad categories.

1. Producers are organisms that manufacture their own food from inorganic compounds by photosynthesis or chemosynthesis (e.g., green plants). These organisms are often referred to as "autotrophs."
2. Consumers are organisms that ingest other organisms (e.g., animals that consume plants or other animals).
3. Decomposers are organisms that derive their nourishment from dead organic matter (e.g., many fungi and bacteria).

These categories are based on the broad interrelationships among groups of organisms but do not describe the many ways in which these interactions may occur.

Organisms that obtain their food in a functionally similar way constitute a "feeding guild," which is a convenient way of considering the breadth of interrelationships among groups of organisms, as it implicitly includes taxonomic and energetic interrelationships, as well as functional ecological relationships. Food webs based on feeding guilds facilitate the identification of critical ecosystem functions above the guild level, and aid in the identification of interrelationships among guilds, which may affect other ecosystem properties. Below, terrestrial and aquatic functional food webs are considered.

It is noteworthy that exotic (non-indigenous) plant and animal species, while commonly components of most ecosystems, are considered stressors for indigenous species. For the purpose of developing GAEs for the BMI Complex, *exotic organisms are not considered valued components* of the ecosystem. All functional groups identified herein include only native species.

B-3.2.1.1 Food Webs Applicable to the BMI Complex

The terrestrial and aquatic ecosystems of the BMI Complex (including the Kerr-McGee Seep and Las Vegas Wash) can be considered as a single integrated ecosystem due to their close proximity and the necessity of association of terrestrial and aquatic biota in this arid environment. Water availability in this region can be limiting for the range, foraging and migratory patterns of many

organisms in the region. Additionally, aquatic and terrestrial environs are closely linked in terms of energy and nutrient flows.

Figures B-1 and B-2 illustrate a current understanding of functional food webs for the BMI Complex and associated areas.

Figure B-1. A terrestrial food web organized by trophic categories and functional feeding guilds.

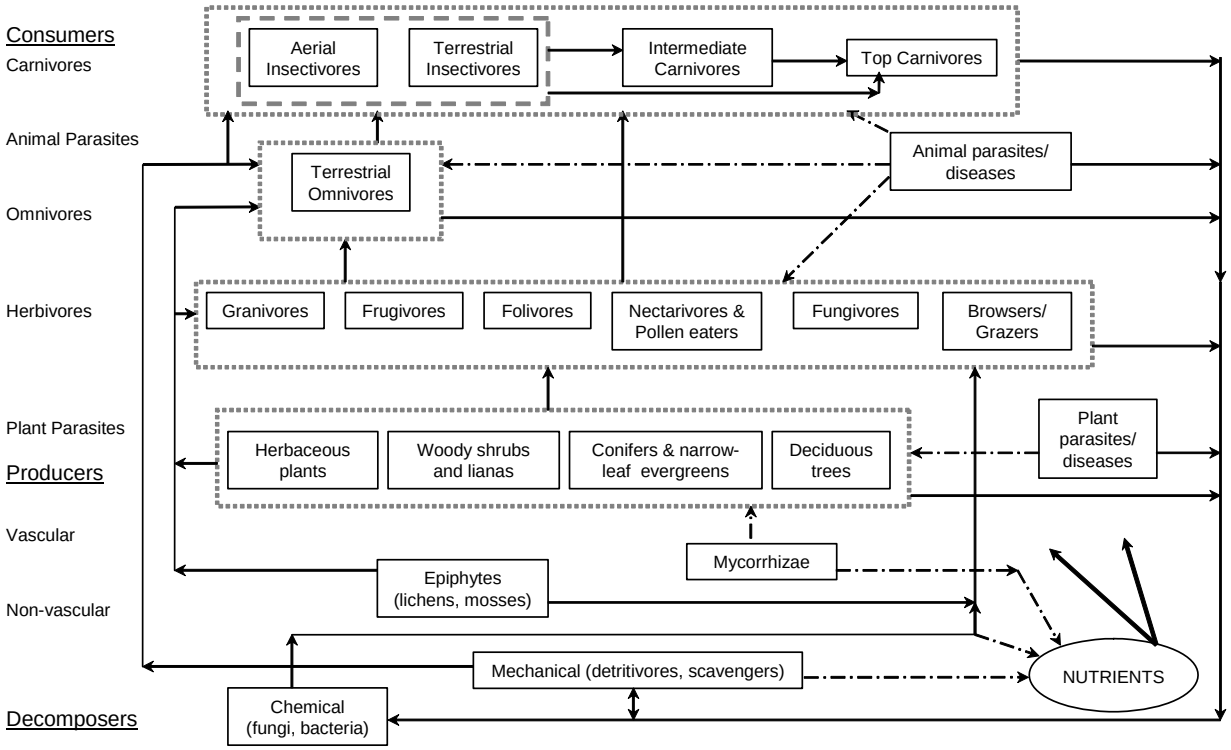
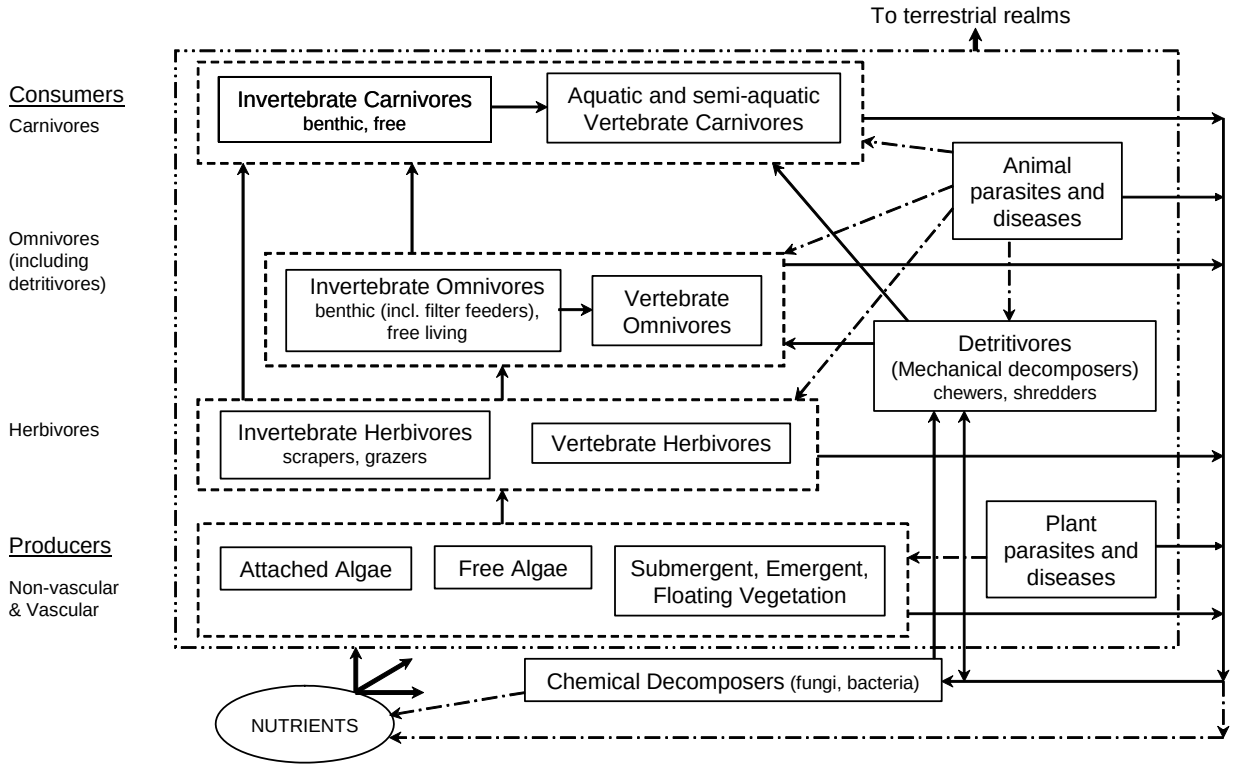


Figure B-2. An aquatic food web organized by trophic categories and functional feeding guilds.



In order to populate Figures B-1 and B-2, the reader is asked to consult Woolbright (2000) and EDAW (2002), as well as additional supporting documents that may be available thorough organizations, institutions, and agencies concerned with the northern Mojave Desert (e.g., The Las Vegas Wash Coordination Committee, Southern Nevada Water Authority, Desert Research Institute, U.S. Fish and Wildlife Service, Nevada Division of Environmental Protection), as may be applicable to the BMI Complex and area. Other, more general references may also be consulted (e.g., Rundel and Gibson [1996]) for valuable insight into northern Mojave Desert biotic communities.

B-3.2.2 Attributes of the Functional Components

The functional attributes of organisms occupying the BMI Complex and immediate surrounds are defined on the basis of their role in the food web, yet each of these components possess additional ecologically important attributes. For example, while shrubs may supply leaves and seeds for food, they also provide important structural habitat for nesting birds and ground squirrels. Nectar and pollen-feeding animals may be relatively unimportant in terms of nutrient and energy transfer through the food web, but critically important as plant pollinators. Relevant attributes of ecological components occupying the BMI Complex and surrounds are defined below (Table B-1).

Table B-1. Attributes of ecological components occupying the BMI Complex and surrounds.

ATTRIBUTE	DEFINITION
Food	Source(s) of energy and nutrients for organisms
Habitat	The biotic and abiotic structural environment in which organisms carry out their life functions.
Energy and Nutrient Fixation	The processes by which inorganic chemicals are yielded useful to living organisms.
Decomposition	The breakdown of dead organic matter by mechanical or chemical processes (both biotic and abiotic).
Propagule Dispersal	The distribution of reproductive propagules (e.g. seeds, spores, or vegetative bodies) from a parent organism into the environment.
Pollination	The sexual reproductive mechanism of flowering and seed-bearing plant species. For many plants, this process is mediated solely by symbionts (e.g. bees).
Control	The processes by which the abundance and distribution of organisms are affected by predation, herbivory and parasitism.

Attributes of each functional component of the ecosystem are presented in Table B-2. Each functional component has at least one attribute. While some attributes could be considered more important than others, the table summarizes ecological values useful for identifying GAEs. One may read GAEs from the table in sentence form: for example, “top carnivores and intermediate carnivores are valued components of the BMI Complex because of their role in control”.

Table B-2. Significant ecological attributes of functional subgroups.

	Significant Ecological Attribute						
Functional Subgroups	**Food Source**	**Habitat**	**Energy and Nutrient Fixation**	**Decomp-osition**	**Propagule Dispersal**	**Pollination**	**Control**
Predators and Omnivores							
Top Carnivores (e.g. raptors, foxes)							◎
Intermediate Carnivores (e.g. snakes, birds, invertebrates)							◎
Aquatic carnivores (e.g. fish, dragonflies)							
Terrestrial Insectivores (e.g. rodents, lizards, arachnids)	◎						◎
Aerial Insectivores (e.g. birds, bats)							◎
Terrestrial Omnivores (e.g. birds, mammals)	◎			◎			
Aquatic Omnivores (e.g. mollusks, freshwater crustaceans, caddisflies)	◎			◎			
Herbivores							
Granivores/ Frugivores (e.g. insects, rodents, birds)	◎				◎		
Folivores (e.g. herbivorous insects)	◎						
Browsers (e.g. lagomorphs)	◎						
Nectarivores/pollen eaters (e.g. insects, other invertebrates, birds)	◎					◎	
Fungivores (e.g. insects, mammals)					◎		
Aquatic Herbivores (e.g. fish, benthic scrapers, tadpoles)	◎						
Parasites (e.g. insects)							◎
Autotrophs							
Native Herbaceous Plants (e.g. grasses, forbs)	◎	◎	◎				

Table B-2 (continued). Significant ecological attributes of functional subgroups.

Native Woody Shrubs (e.g. creosote bush, white bursage)	◎	◎	◎				
Native Conifers (e.g. douglas fir, piñon)	◎	◎	◎				
Native Deciduous Trees (e.g. aspen, cottonwood)	◎	◎	◎				
Native Submergent, Emergent and Floating Plants (e.g. duckweed, watercress)	◎	◎	◎				
Submergent Aquatic Plants (e.g. algae)	◎	◎	◎				
Epiphytes (e.g. lichens, some terrestrial algae)	◎	◎	◎				
Decomposers							
Mycorrhizae (nitrogen-fixing symbiotic fungi, associated with plant roots)			◎				
Mechanical Decomposers (e.g. earthworms, detritivores, scavengers, shredders)	◎			◎			
Chemical Decomposers (e.g. fungi, bacteria)	◎			◎			

B-3.3 GAEs Based on Ecological Relevance

B-3.3.1 Globally Relevant Endpoints

The following GAEs are based on ecological values characteristic of all ecosystems:

- Biodiversity is a valued ecological attribute because of its importance to human use, contribution to resilience, and importance for maintaining structure and function.
- Functional integrity is a valued attribute because it connotes an intact system, one in which there is no missing link that would result in structural or functional imbalances that render the entire system more vulnerable (less resilient) to perturbation.
- Energy and nutrient dynamics is a valued attribute because flow rates and patterns of nutrient and energy processing are critical for maintaining populations of indigenous species at levels characteristic of the ecosystem.

B-3.3.2 Regionally Relevant Endpoints

The following regional GAEs are based on the definitions provided in Table B-1 and Table B-2.

- Top carnivores and intermediate carnivores are valued components of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their role in control.
- Terrestrial insectivores are a valued component of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their importance both in control and as a food source to higher level carnivores.
- Aquatic carnivores and intermediate carnivores are valued components of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their specialized role in control.
- Aerial insectivores are a valued component of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their importance in processes of control.
- Terrestrial and aquatic omnivores are valued components of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their roles in decomposition and as a food source to higher level carnivores.
- Granivores and frugivores are valued components of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their importance as a food source to higher level carnivores and their role as propogule dispersers.
- Folivores and browsers are a valued component of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their importance as a food source to higher level carnivores and their role as non-food chain based propogule dispersers (e.g., seeds cling to their coat).
- Nectarivores and pollen eaters are valued components of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their importance in pollination and value as a food source.
- Fungivores are a valued component of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their importance in fungal species propogule dispersal.

- Aquatic herbivores are a valued component of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their importance as a food sources and role in aquatic decomposition.
- Plant and animal parasites are valued components of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their influence on population dynamics.
- All native herbaceous and woody plants and shrubs, conifers, deciduous trees, emergent plants, epiphytes, and lianas are valued components of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their importance as food sources and habitat, as well as their role in nutrient cycling.
- Aquatic plants are a valued component of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their importance as food sources and habitat and their role in nutrient cycling.
- Mycrrohizae are a valued component of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their importance in nutrient recycling and regeneration of soils.
- Mechanical and chemical decomposers are a valued component of the BMI Complex and surrounds (northern Mojave Desert ecosystem) because of their importance in decomposition, nutrient recycling and as a food source.

B-4.0 Values and GAEs For the BMI Complex and surrounds Based On Societal Relevance

Ecological risk assessments should be conducted to identify or predict potentially adverse impacts of environmental stressors. Ultimately, however, the effectiveness of an ecological risk assessment depends on how it improves the quality of management decisions. Risk managers are more willing to use a risk assessment as the basis for making remedial decisions if the risk assessment considers ecological values that people care about (EPA 1998). Therefore, an ecological risk assessment must consider both ecological and societal values to be effective.

B-4.1 Criteria for Management Goals

Management goals are inextricably tied to the societal values of ecological resources. As the NDEP develops management goals for northern Mojave Desert habitats, they will be reflected in the GAEs developed for any given site. Values include formally recognized and protected ecological resources such as threatened and endangered species and habitats, native species and their assemblages, and recreationally important species and habitats. Identification of societal values should involve input from risk managers, risk assessors, ecologists, appropriate regulatory authorities (e.g., municipal, county, state, tribal, and federal agencies, local and regional conservation services and institutions), other experts (e.g. anthropologists), and the general public.

Other societal values for the BMI Complex and surrounds (northern Mojave Desert ecosystem) may be identified based on a review of the management goals and plans for areas potentially affected by historical and current BMI Complex activities. For example, a given area affected by historical and current BMI Complex activities may be under simultaneous management for protection of specific habitat or sensitive species, erosion control, fire suppression or protection

of archeological sites. Any of these potential connections must be thoroughly explored before, during, and after the implementation of the ecological risk assessment process.

Societal values recognized for the development of GAEs should incorporate concerns for clean water and watershed protection (both of which may fall under the scrutiny of regulatory compliance), clean soil and control of dust and erosion, etc. In order to incorporate these concerns, GAEs should be developed with an eye on neighboring systems of land use and control, as these may impact operations on the area of consideration.

B-4.2 GAEs Based on Societal Relevance

The specification of assessment endpoints with societal relevance is the last step in the process of identifying a comprehensive list of GAEs. The following GAEs are proposed for consideration as those that hold broad societal relevance and relevance to communities and stakeholders that may have interest in the BMI Complex and affected areas.

- Recreationally and anthropogenically important species are valued components of the ecosystem and are to be protected because of their importance for consumptive uses such as gathering, and for non-consumptive uses, such as bird watching.
- Threatened and endangered species, their habitats, and migratory bird nesting, roosting and lighting sites are valued components of the ecosystem to be protected because of their regulatory stature.
- The quality and quantity of water within each watershed are valued components of the ecosystem and require management of point and non-point sources of contaminants, consumptive water usage or diversion, erosion and total suspended materials to meet regulatory limits and Total Maximum Daily Loads (TMDLs)
- Certain indigenous plants and animals are valued components of the ecosystem and are to be protected because of their ethnological and other consumptive and non-consumptive uses.
- The aesthetic quality of the landscape is a valued component of the ecosystem because of its value to society.
- Wetlands, springs, and riparian areas are valued due to their unique qualities, and any applicable promulgated protection (e.g., Chapter 445A [Water Controls] of the State of Nevada Legislative Statutes), as well as their important ecological functions.

B-5.0 Application of GAEs in the Ecological Risk Assessment Process

GAEs are developed using a process based on ecological principles and knowledge of the ecological components and characteristics of an ecosystem. Additionally, GAEs reflect societal values and regulatory requirements. Development of GAEs may involve regulators, trustees, and other stakeholders. Thus, the GAE process delineates the “array of possibilities” from which the specific assessment endpoints are derived.

GAEs have been developed to ensure that values at all levels of ecological organization will be considered in the subsequent identification of site-specific assessment endpoints. The GAE process provides a framework for systematically considering how effects on particular species or other taxonomic groupings could affect functional components as well as higher levels of

ecological organization (e.g., biological diversity, functional integrity or nutrient and energy cycling). Having stated the GAEs in Sections 3.0 and 4.0, it is now appropriate to apply the third major criterion for selecting assessment endpoints, the susceptibility of receptors to known or potential environmental stressors.

Characterizing the species and habitats at a site and identifying which of these are sensitive to site contaminants are necessary first steps in the identification of site-specific assessment endpoints. Knowledge of receptor susceptibility may be used to identify site-specific assessment endpoints. The following questions should be answered in order to determine which GAEs are potentially affected by site-related contaminants:

- Which potential receptors (species representative of each functional group) and habitats are present in the area of concern?
- Which potential receptors are sensitive to which contaminants in the area of concern?
- What exposure pathways exist between contaminant sources and sensitive species (e.g., direct exposure, food chain transfer, etc.)?

Not all contaminants need to be considered simultaneously when identifying assessment endpoints. Details of the specific area under study such as contaminants, contaminant properties (e.g., bioavailability, bioaccumulation potential), ecological receptors present, sensitivity of receptors to contaminants, and exposure pathways, are evaluated by constructing conceptual site models and conducting a toxicity-based assessment. Multiple contaminants present at a site may act on various receptors through different exposure pathways, thus assessment endpoints may differ for each contaminant.

There are a number of ways that the GAE process is used to develop site-specific assessment endpoints. For instance, where aquatic insects may be adversely affected, these would be an obvious value to be protected. It follows that the biodiversity of aquatic macroinvertebrates could also be considered as an appropriate assessment endpoint. However, it is less obvious that because the "detritivore" functional component of the aquatic ecosystem is composed partially of certain insects, decomposition rates for the aquatic system could be diminished as a result of contaminant effects on them.

Variability in ecological, time, and geographic scale is important in deciding how to apply GAEs to the selection of assessment endpoints. For example, contaminated sediments in a spring may have undetectable effects on the total biodiversity of the entire BMI Complex and surrounding area, but may adversely affect the benthic biodiversity of the spring. It is important to consider geographic scale of effect (e.g. local, watershed, regional) when considering a specific assessment endpoint. It is also important to distinguish between effects on variable time scales, as this may, in turn, effect the selection of assessment endpoints. Time-dependent scales of effect may include processes that are population based (e.g. population viability measures) or community based (e.g. species exclusion based on competitive inhibition/release due to contaminant effects). For example, population-based effects from contamination may be more readily observed in short-lived organisms (e.g. rabbits) than in long-lived organisms (e.g. coyotes).

Once site-specific assessment endpoints have been identified, at least one measure of effect or exposure must be selected to evaluate the potential risk posed to each assessment endpoint. (It is beyond the scope of this Appendix to treat the development of appropriate measures in detail.) A measurement endpoint is a measurable characteristic that is related to the valued characteristic chosen as the assessment endpoint (EPA 1997). When selecting appropriate measures, it is important to consider the way in which the results will be used to contribute to the risk assessment. Typically a weight of evidence approach is used, combining multiple lines of evidence in a qualitative or quantitative fashion. Thinking ahead about which lines of evidence will be supportive during the risk characterization phase will ensure that useful measures are selected.

Most assessment endpoints are addressed by measures that include one or more of the following:

- Media-specific contaminant measurements.
- Tissue analysis of plants and lower trophic-level animals.
- Food chain modeling to higher trophic-level organisms.
- Biological toxicity testing and bioaccumulation studies conducted under controlled conditions.
- Field measurements of biodiversity and various aspects of ecosystem function and health.

In some instances biomarkers (metabolic byproducts of specific contaminants) are also useful measures, since they can be used to determine more directly whether a receptor has actually been exposed to the stressor of concern.

Each of the GAEs should be considered in a site-specific assessment endpoint, or an explanation should document why no site-specific assessment endpoint is necessary. For example, a site-specific assessment endpoint is not required if a GAE is not pertinent to an assessment, e.g. due to an incomplete exposure pathway or lack of toxic effects. These considerations must be consistent with the conceptual site model and functional food webs for the geographic area under study.

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