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Screening Level Ecological Risk Assessment Methods

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EXECUTIVE SUMMARY

This document builds on and clarifies the methods and ideas provided in the previous screening approach document, "Screening Level Ecological Risk Assessment Approach (SLERA) for the Environmental Restoration (ER) Project at Los Alamos National Laboratory" (Kelly et al. 1998, ER ID 57916). This document, as did the previous document, provides guidance for screening level assessments of potential adverse impacts to ecological resources from wastes resulting from past operations at the Los Alamos National Laboratory (LANL or the Laboratory). This document responds to the New Mexico Environment Department (NMED) comments on the previous document. Major changes made in response to those comments include the addition of guidance for sediment, water, and multi-media screening levels, the inclusion of additional receptors to account for on-site bioaccumulation and some biomagnification concerns, the emphasis on interim actions during the scoping evaluation, the inclusion of multimedia evaluations, and the simplification and clarification of multi-contaminant evaluations.

The guidance provided in this document follows the Environmental Protection Agency's (EPA's) "Ecological Risk Assessment Guidance for Superfund" (EPA 1997, ER ID 59370), and the "Guidelines for Ecological Risk Assessment" (EPA 1998, ER ID 62809). This document entirely supersedes the previous SLERA document (Kelly et al. 1998, ER ID 57916).

This methods document has two purposes. The first is to provide a basis for reaching consensus with regulators, managers, and other interested parties as to the best approach for conducting screening level ecological risk investigations at the Laboratory. The second is to provide guidance to ER ecological risk assessors. This guidance will promote consistency, rigor, and defensibility in ecological screening investigations and in the reporting of those investigation results.

Section 1, Introduction, provides a brief introduction to the document. Section 2, Ecological Screening Process Overview, provides an overview of the ER screening assessment process (including a process flow diagram). Section 3, Generic Problem Formulation for Ecological Screening Assessments, describes the Laboratory-wide information that is needed for the screening level ecological risk problem formulation, including the environmental setting, contaminant fate and transport, exposure pathways, and food webs. This Laboratory-wide information provides the basis for the specification of screening level ecological receptors (Section 3.5) and assessment endpoints (Section 3.6).

Section 4, Site-Specific Screening Level Ecological Risk Assessment, is the longest, most complex, and most revised section, and it describes in detail the two phases of the screening assessment: the scoping evaluation (Section 4.2) and the screening evaluation (Section 4.3). The scoping evaluation includes data review, which identifies the list of chemicals of potential concern (COPCs) for the area being evaluated, and the site-specific problem formulation step. The area being evaluated may be a potential release site (PRS) or a collection of PRSs in a watershed or some other aggregation unit. In this document we will use the term *site* broadly to include these different possibilities.

The basis for the problem formulation is found in the Ecological Scoping Checklist, Appendix A. The ecological scoping checklist is a unique and useful tool for organizing existing ecological information and focusing the site visit on the information needed to develop the ecological exposure site conceptual model. It guides the risk assessor through a series of questions, tied to a generic conceptual model diagram, to develop a site-specific conceptual model. The ecological scoping checklist also addresses the issue of contaminant transport and provides the basis for evaluating the adequacy of the data for ecological risk screening. During the scoping process, interim actions, which could reduce the potential

for off-site migration of contaminants to ecological receptors or could mitigate exposure of on-site receptors to contaminants, are evaluated.

The purpose of the screening evaluation is to identify contaminants of potential ecological concern (COPEC) by exposure media. The screening evaluation focuses future investigations on the important ecological concerns and identifies those sites that do not have COPECs. Sites with no COPECs do not need further ecological evaluation. The outcome of the screening is expected to be protective of potential adverse ecological effects but is not intended to be predictive of ecological risk. Thus, conservative assumptions are made throughout the screening evaluation to ensure that contaminants, exposure pathways, and sensitive species are not missed.

Section 4.3, Screening Evaluation Overview, provides an overview of the methods used for screening. The key components of the screening evaluation are ecological screening levels (ESLs). The ESLs are developed for each chemical and are media specific. They are determined so that if a site has levels of a chemical above the ESL in any medium, then this site may pose a potential risk to ecological receptors. The ESLs for non-radionuclides incorporate a factor to account for potential additive effects of COPCs. The radionuclide ESLs are adjusted by the number of radionuclide COPCs to account for the additive dose of multiple radionuclides. Because ESLs are specific to each medium (soil, sediment and water), they do not account for exposure to multiple media. A method to account for wildlife exposure to multiple media is presented. The multimedia exposure calculation results in a hazard index (HI) value for each wildlife receptor. The HI is a sum of hazard quotient (HQ) values. HQs are calculated for each screening receptor and each contaminant and may be thought of as a ratio of a receptor's exposure at the site to an acceptable effects level. If the HI is greater than 1.0, then the site may pose an ecological risk. Sections 4.3, 4.4, and 4.5 describe the HQ and HI calculations and presents in great detail the ESL calculations for various media, receptors, and chemicals.

The ESL, HQ, and HI calculations require toxicity information, including toxicity reference values (TRVs), knowledge of bioconcentration, and bioaccumulation factors for all chemicals for all receptors and media. This information is not provided in this document. NMED requires that the Laboratory document this information in detail to ensure that the best available studies are used to develop TRVs, ESLs, HQs, and HIs. The Laboratory is now in the process of updating the ECORISK Database, which provides the necessary information and supporting detailed documentation (including TRVs and ESLs) (LANL 1998, ER ID Package 186). This database will be updated, as new studies become available.

Section 4.6, Screening Evaluation/Uncertainty Analysis, describes the uncertainty analysis that follows the COPEC identification. This section describes the key sources of uncertainty in the screening assessment. Uncertainty analysis can result in adding chemical constituents to or removing them from the list of COPECs.

The purpose of the screening assessment is to provide information to the risk managers so that informed risk management decisions can be made. Section 4.7, Risk Interpretation, provides examples of recommendations and possible risk management strategies.

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ACRONYMS

ACR	acute-chronic ratio
AET	apparent effects threshold
AOC	area of concern
ARAR	applicable or relevant and appropriate requirement
AUF	area use factor
AWQC	ambient water quality criteria
BCF	bioconcentration factor
BMP	best management practice
CCC	criterion continuous concentration
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CMS	corrective measures study
CMI	corrective measure implementation
COPC	chemical of potential concern
COPEC	contaminant of potential ecological concern
CV	chronic value
DCF	dose conversion factor
DOE	Department of Energy
EqP	equilibrium partitioning method
EPA	Environmental Protection Agency
ESL	ecological screening level

ER	Environmental Restoration
ERL	effects range low
ERM	effects range median
FACR	final acute-chronic ratio
FAV	final acute value
FCV	final chronic value
FIMAD	Facility for Information Management, Analysis, and Display
FDEP	Florida Department of Environmental Protection
FPV	final plant value
GLI	Great Lakes Initiative
GMAV	genus mean acute value
GMCV	genus mean chronic value
GIS	geographical information system
GPS	global positioning system
HI	hazard index
HQ	hazard quotient
IA	interim action
IAEA	International Atomic Energy Agency
IWP	Installation Work Plan
K_{oc}	organic carbon partitioning coefficient
K_{ow}	octanol/water partition coefficient
LANL	Los Alamos National Laboratory
LC_{50}	concentration (of toxicant) which is lethal to 50% of test population
LOAEL	lowest observed adverse effect level
LOEC	lowest observed effect concentration
NAWQC	National Ambient Water Quality Criteria
NCRP	National Council on Radiation Protection
NFA	no further action
NMED	New Mexico Environment Department
NOAEL	no observed adverse effect level
NOEC	no observed effect concentration
OU	operable unit
ORNL	Oak Ridge National Lab
OW	Office of Water
P	empirical cumulative probability
PEL	probable effects level
PRS	potential release site
PQL	practical quantitation limit

RCRA	Resource Conservation and Recovery Act
RFI	[RCRA] facility investigation
R_t	retention time
SACR	secondary acute-chronic ratio
SAV	secondary acute values
SAVF	secondary acute value factor
SCM	site conceptual model
SCV	secondary chronic values
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
SOP	standard operating procedure
SQB	sediment quality benchmark
SQC	sediment quality criteria
SMDP	scientific management decision point
SSB	sediment screening benchmark
SWMU	solid waste management unit
TA	technical area
T&E	threatened and endangered
TEL	threshold effects level
TF	transfer factors
TRV	toxicity reference value
UCL	upper confidence limit
VCA	voluntary corrective action
VCM	voluntary corrective measure
WQB	water quality benchmark
WQC	water quality criteria

1.0 INTRODUCTION

This document describes the approach used by the Los Alamos National Laboratory (LANL or the Laboratory) Environmental Restoration (ER) Project for screening level assessments of potential impacts to ecological resources resulting from legacy wastes, i.e., wastes resulting from past operations at the Laboratory. This approach follows the Environmental Protection Agency's (EPA's) "Ecological Risk Assessment Guidance for Superfund" (EPA 1997, ER ID 59370), and the Guidelines for Ecological Risk Assessment" (EPA 1998, ER ID 62809). This document responds to the New Mexico Environment Department (NMED) comments on the previous document. Major changes made in response to those comments include the addition of guidance for sediment, water, and multi-media screening levels, the inclusion of additional receptors to account for on-site bioaccumulation and biomagnification concerns, the emphasis on interim actions during the scoping evaluation, the inclusion of multimedia evaluations, and the simplification and clarification of multi-contaminant evaluations.

A broad audience is anticipated for this document, including NMED regulators, Department of Energy (DOE) and Laboratory ER Project managers, ER Project staff implementing this approach, and other interested parties and practitioners. This approach document provides more detail than will be pertinent to the interests of many in this diverse audience. Sections 1, 2, and 3 should be of interest and accessible to the general audience. ER practitioners and regulators will become well acquainted with Section 4, which includes details of the methods and calculations used for screening level ecological evaluations.

The EPA guidance requires that initial screening level assessments use conservative assumptions to evaluate the potential for adverse ecological impacts. The rationale behind this requirement is to provide a high confidence that all potential adverse impacts to ecological receptors resulting from legacy wastes are identified in the initial investigations. Thus, the screening level assessment may be used to identify sites that clearly pose no threat to the environment as well as sites that need immediate corrective action. However, for the many sites that do not fall into one of these two categories, screening level evaluations must be followed by a series of progressively more in-depth and site-specific evaluations to accurately characterize risks and to provide adequate information for risk management decisions. The screening level assessment helps to focus these more detailed (and often more complex) site-specific investigations by identifying important contaminants, ecological endpoints, and spatial scales. The screening level evaluation also employs a common metric for comparing risks among different sites, thus providing a tool for prioritizing site investigations and corrective actions.

This methodology documents the ecological screening process for application to individual potential release sites (PRSs) or clusters of PRSs. Application of this methodology to larger spatial aggregates is not explicitly considered, but this process is intended to be sufficiently flexible to evaluate potential ecological risk at a variety of spatial scales. The main difference in application of the screening methodology to large spatial aggregates is the calculation of an appropriate representative exposure concentration for the screening receptors. The methods described in this document are intended for assessing current day risk at the site where contamination from legacy wastes has been investigated. However, these methods, coupled with the appropriate transport models, can be used to assess the potential for future ecological risk at areas affected by off-site transport of contaminants. The discussion and evaluation of transport models, other than to emphasize their importance, is beyond the scope of this document.

2.0 ECOLOGICAL SCREENING PROCESS OVERVIEW

The ecological screening process flow diagram is shown in Figure 2.0-1. Prior to application of the screening process is the pre-scoping site evaluation. The purpose of pre-scoping to eliminate sites based on administrative criteria. Following the pre-scoping evaluation, the screening process is composed of two parts, the scoping evaluation and the screening evaluation. The purpose of the screening is to provide information to support the risk management decision, which is based on the interpretation of the screening results.

Pre-Scoping Evaluation

The first step of the pre-scoping evaluation is to determine if the site is a candidate for an administrative no further action (NFA) decision. The administrative NFA is based on the following New Mexico Environment Department (NMED) criteria (NFA Criteria are listed in Section II.B.4.a.(4).(b), "No Further Action (NFA) Proposals Criteria," in the *NMED RCRA Permits Management Program Document Requirement Guide* (NMED 1998, ER ID 57897).

- NFA Criterion 1. The Solid Waste Management Unit/Area of Concern (SWMU/AOC) cannot be located, does not exist or is a duplicate SWMU/AOC.

- NFA Criterion 2. The SWMU/AOC has never been used for the management (i.e., generation, treatment, storage and/or disposal) of Resource Conservation and Recovery Act (RCRA) solid waste or hazardous wastes and/or constituents or other Comprehensive Environmental Response, Conservation and Liability Act (CERCLA) hazardous substances.

- NFA Criterion 3. No release to the environment has occurred or is likely to occur in the future from the SWMU/AOC. (For an administrative NFA this is generally based on process knowledge, rather than environmental sample information.)

The Environmental Restoration (ER) Project personnel provide the justification for administrative NFA recommendations. Given the fact that administrative NFAs do not affect environmental media, environmental sample information is usually not required, and ecological evaluations are unnecessary. Although administrative NFAs are considered for individual potential release sites (PRSs), in the remainder of this document, the term *site* is used broadly to represent a PRS or a cluster of PRSs. The criteria to define PRS clusters are currently under development. The main difference in the application of this methodology to individual PRSs or clusters of PRSs is the size of the area being evaluated. The size of the site typically directly impacts the complexity of the assessment, which will be documented during site scoping.

The second part of the pre-scoping evaluation is data review. During data review chemists, statisticians, data analysts, and risk assessors work together to identify chemicals of potential concern (COPCs) (LANL 1998, ER ID 58981). COPCs are typically inorganic chemicals and radionuclides measured greater than background concentrations, and detected organic chemicals. If there are no COPCs identified, then the site may be recommended for NFA. The site investigation team provides the justification for these recommendations in the Resource Conservation and Recovery Act (RCRA) facility investigation (RFI) report and further ecological evaluations are unnecessary.

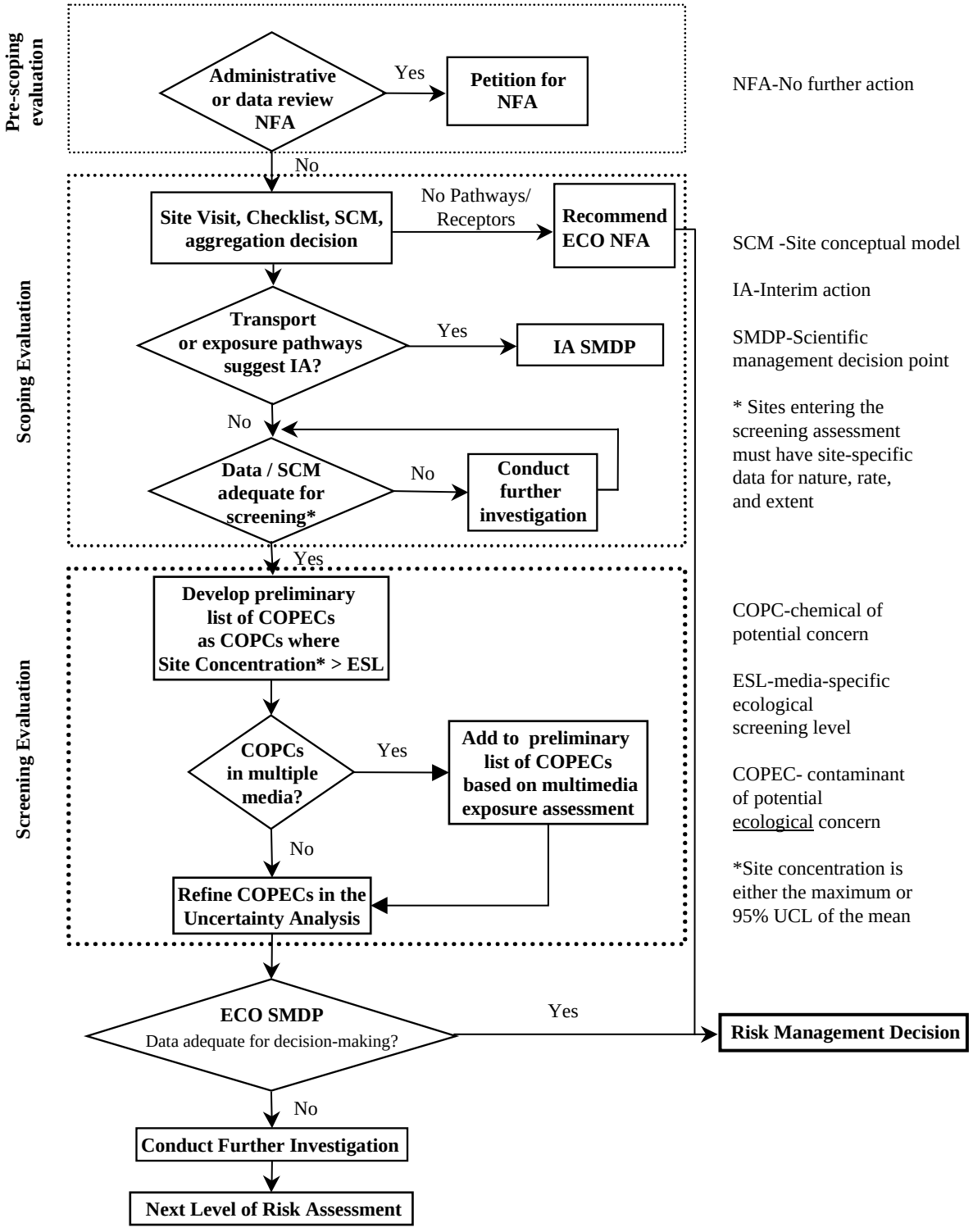


Figure 2.0-1. Process flow for ecological screening assessment.

Scoping Evaluation

Any sites that are not proposed for NFA during pre-scoping must undergo ecological scoping, including a site visit by a member of the ecological risk assessment task team and completion of the ecological scoping checklist (described in detail in Section 4.2, Scoping Evaluation, and presented in Appendix A). The ecological exposure site conceptual model is developed during scoping, using the ecological scoping checklist. Fate and transport issues relative to ecological concerns are assessed during scoping, and particular attention is paid to the need for an interim action Scientific Management Decision Point (IA SMDP) (Figure 2.0-1). In the IA SMDP, the site Project Leader involves the appropriate technical and management personnel in an evaluation of interim actions for the site. Such interim actions would be obvious actions that could reduce the potential for off-site migration of contaminants to ecological receptors or could mitigate exposure of on-site receptors to contaminants. Examples of interim actions include stabilization measures such as run-on and run-off controls, source removal actions, or controlling site access. These are often referred to as best management practices (BMPs).

The clustering of PRSs is also addressed during scoping (i.e., should other PRSs be combined with this site for the screening assessment?). After the scoping evaluation, if the ecological risk assessment team determines that the site poses no threat to the environment because there are no ecological receptors and/or there are no pathways to receptors, a recommendation for ecological NFA is made. The justification for this recommendation is documented in the Ecological Screening Assessments section of the RFI report (LANL 1998, ER ID 58981) or equivalent section of other reports.

During scoping, a decision is made about the adequacy of the data and the site conceptual model for the screening evaluation (Figure 2.0-1). At a minimum, the ecological screening evaluation must be performed for all relevant media (e.g., soil, water, or sediment) that have a complete ecological exposure pathway. Before the screening evaluation can be performed, site-specific data must be deemed adequate for characterizing the nature, rate, and extent of contamination in order to justify use of the sample maximums as reasonable estimates for the highest concentrations expected at the site. If adequate data do not exist for the site, a recommendation must be made to collect additional data. If existing data may not represent the highest contaminant levels, the benefits of collecting additional data should be evaluated against the bias in the current sample maximum values. It should be noted that when data are adequate[1] and appropriately distributed, the 95% upper confidence limit (UCL) of the mean media concentration may be used instead of the maximum value in calculations and comparisons (Dinwiddie 1998, ER ID 62741).

Screening Evaluation

Once the scoping process is complete, the screening evaluation is conducted. The goal of the screening evaluation is to identify the contaminants of potential ecological concern (COPECs). This requires toxicity information for the COPCs identified in data review. If toxicity information is not available for a particular COPC, it is identified as a COPEC and must be addressed in the uncertainty analysis. Figure 2.0-1 shows two steps may be need to make a preliminary list of COPECs. In the case of a site with a single contaminated medium (soil, sediment, or water), COPECs are identified by determining if site

[1] Considerations of data adequacy to calculate a 95% upper confidence level (UCL) include having the spatial coverage of the contaminated area and having sample results that appear to be derived from a single statistical distribution.

concentrations[2] are greater than the media and COPC-specific ecological screening levels (ESLs). (Note that the ESLs incorporate factors to account for potential additive effects of multiple COPCs.)

The COPEC identification first requires assembling ESLs for all media and all COPCs. If a site has levels of a chemical in any medium above the ESL, then this COPC is considered a COPEC for the site. Developing ESLs is a complicated process and, for receptor-specific ESLs, requires a vast amount of information about the toxicity of the COPC and the exposure parameters for the receptor. The methods used to calculate ESLs are presented in detail in this document. However, the actual ESLs and the toxicity and other parameter information required for their calculation are maintained in a separate document, the ECORISK Database (LANL 1998, ER ID Package 186). The ECORISK Database is available to anyone performing or reviewing ecological screening assessments for the Laboratory, and notices of updates to this database will be issued as new information becomes available.

If the site has multimedia contamination, the hazard index (HI) is also calculated for each screening receptor. The HI is the sum of hazard quotients (HQs) for all of the COPCs and the HQ may be thought of as a ratio of a receptor's exposure at the site to an acceptable effects level. If the HI is greater than 1.0, those COPCs contributing 0.3 or more to the HI are identified as COPECs.

The ESL comparisons and HQ/HI calculations are followed by an uncertainty analysis that focuses on key sources of uncertainty in the screening assessment and can result in the addition or deletion of COPECs. The list of COPECs is not considered to be final for screening until after the uncertainty analysis. The main components of the uncertainty analysis are described in Section 4.6, Screening Evaluation/Uncertainty Analysis.

Following the uncertainty analysis, the results of the screening assessment are provided to the risk managers. At this point an ecological SMDP is required. Out of this SMDP, a risk management strategy may be recommended by the risk assessors. Possible recommendations and risk management strategies are discussed in Section 4.7, Risk Interpretation.

[2] Guidance from NMED allows the use of the maximum or the 95% UCL of the mean where appropriate (Dinwiddie 1998, ER ID 62741).

3.0 GENERIC PROBLEM FORMULATION FOR ECOLOGICAL SCREENING ASSESSMENTS

As noted in the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA)-specific (Superfund) ecological risk guidance (EPA 1997, ER ID 59370), problem formulation is the most critical step of an ecological risk assessment. The Superfund guidance identifies (among others) the following issues for the screening level problem formulation:

- Environmental setting (physical and biological)
- Contaminant fate and transport
- Food webs
- Screening receptors
- Exposure pathways
- Assessment endpoints

Problem formulation at Los Alamos, therefore, requires understanding of the physical and biological setting of the Laboratory. The physical setting greatly influences the potential contaminant transport pathways, which also influence the potential exposure pathways for ecological receptors. The biological setting is important for receptor selection because receptors must represent the broad spectrum of plant and animal species present at the Laboratory. One key exposure pathway is expressed through the food web (see Section 3.4, Functional Food Web). An understanding of the feeding relationships among animals and plants can be organized into a food web and used to develop representative groups of ecological receptors. Receptor groupings based on feeding relationships are an efficient and effective way to represent all ecological resources (biota) of relevance. In the following sections, the general physical setting of the Laboratory and the surrounding area is summarized first and followed by descriptions of the salient biotic features.

3.1 Environmental Setting

The Laboratory is situated on the Pajarito Plateau, which consists of a series of finger-like mesas separated by deep east-to-west oriented canyons cut by intermittent streams. Mesa tops range in elevation from approximately 7800 ft on the flanks of the Jemez Mountains to about 6200 ft at their eastern termination above the Rio Grande Canyon. Climate, geographic setting, geology, hydrology, and biology of the Laboratory are described briefly below.

3.1.1 Geographic Setting

The Laboratory and residential and commercial areas of Los Alamos and White Rock are located in Los Alamos County, in north central New Mexico, approximately 60 miles north-northeast of Albuquerque and 25 miles northwest of Santa Fe (Figure 3.1.1-1). The surrounding land is largely undeveloped, with large tracts of land north, west, and south of the Laboratory held by the Santa Fe National Forest, Bureau of Land Management, Bandelier National Monument, General Services Administration, and Los Alamos County. The Pueblo of San Ildefonso borders the Laboratory to the east.

The Laboratory is divided into technical areas (TAs) that are used for building sites, experimental areas, waste disposal locations, roads, and utility rights-of-way (see Figure 3.1.1-2). However, these uses account for only a small part of the total land area. Most land provides buffer areas for security and safety and is held in reserve for future use. Thus, the majority of the Laboratory is undeveloped land that supports diverse and abundant ecological resources.

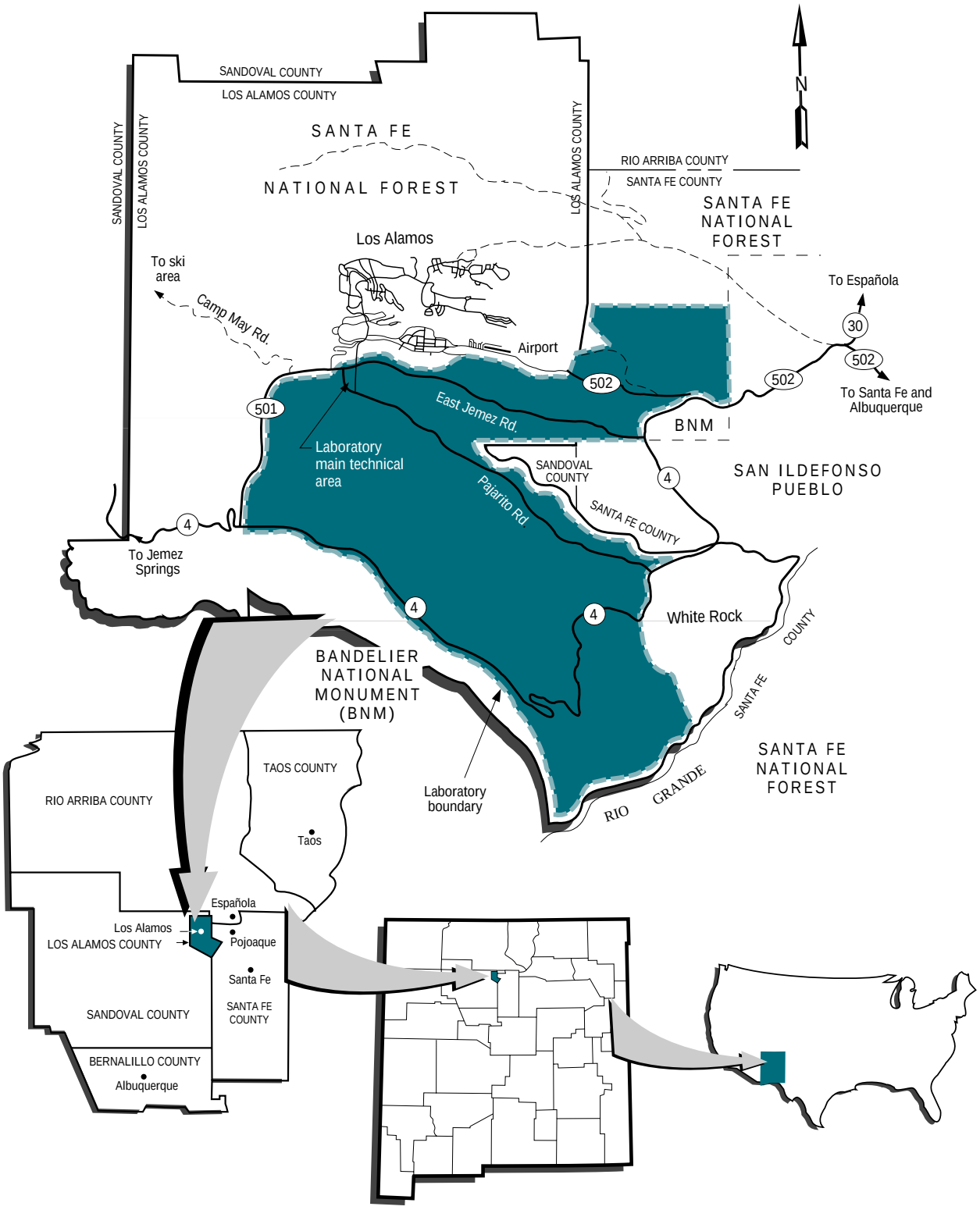


Figure 3.1.1-1. Regional location of Los Alamos National Laboratory.

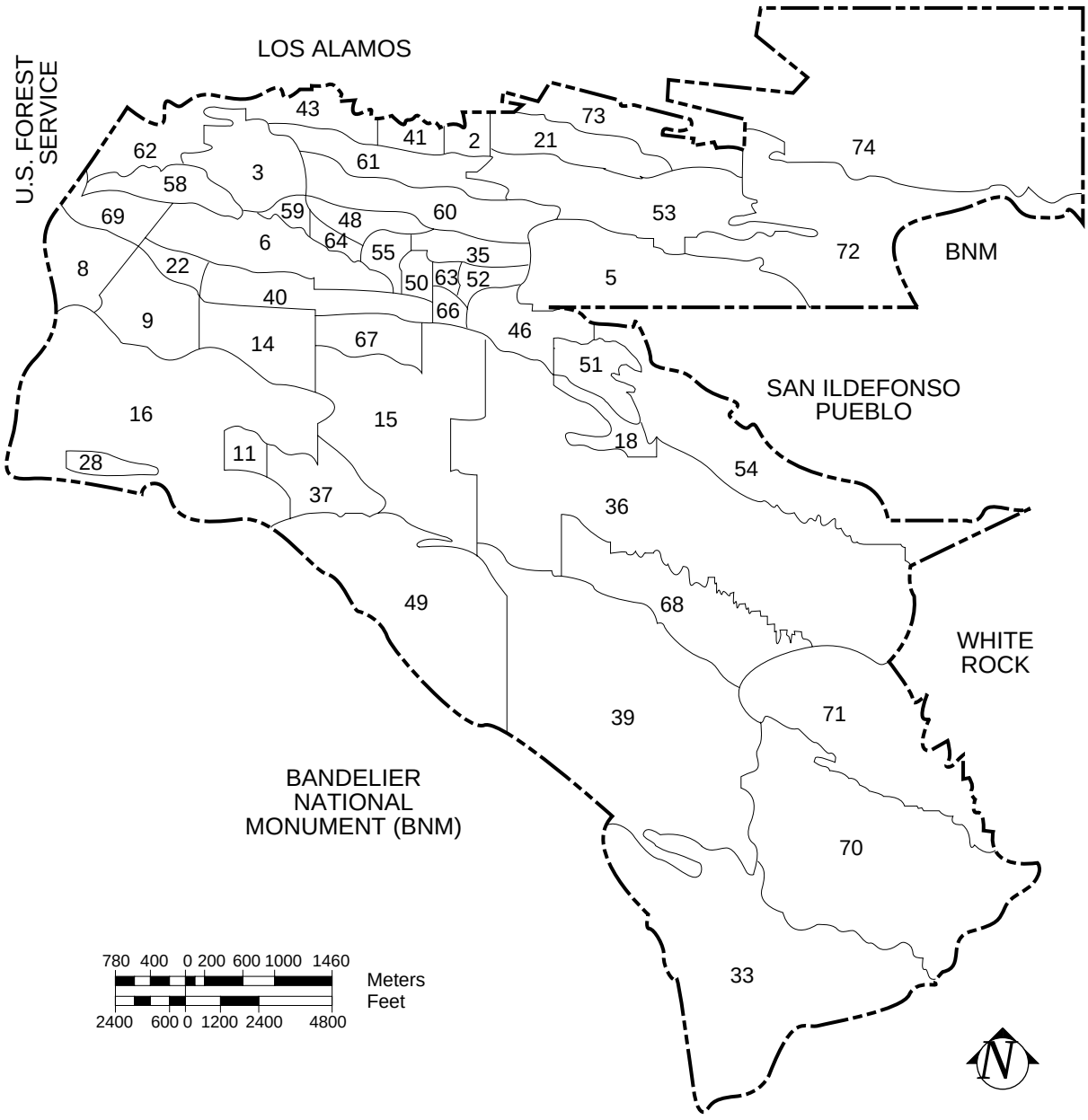


Figure 3.1.1-2. Technical areas of Los Alamos National Laboratory in relation to surrounding landholdings.

3.1.2 Climate

The average diurnal temperature at Los Alamos is 13°C (55°F). Winter temperatures range from -1°C to 10°C (30°F to 50°F) during the daytime, to -9°C to -4°C (15°F to 25°F) during the nighttime. Summer temperatures range from 21°C to 31°C (70°F to 88°F) during the daytime, to 10°C to 15°C (50°F to 59°F) during the nighttime. The average annual precipitation (including both rain and water equivalent of frozen precipitation) is 48 cm (19 in.). Details are available through the World Wide Web at http://weather.lanl.gov/ and are discussed in the "Installation Workplan for Environmental Restoration Project" or IWP (LANL 1998, ER ID 58605, p. 2-40).

The semiarid, temperate, mountain climate in Los Alamos County influences weather and soil development, as well as biotic assimilation in the region. Both weather and soil conditions influence transport of contaminants at the Laboratory and potential exposure of ecological receptors to contamination. The speed, frequency, direction, and persistency of wind can influence the airborne transport of contaminants. High winds, which are common in the spring, can result in atmospheric transport of contaminants (see discussion in Section 3.2, Contaminant Fate and Transport). Additional discussion of atmospheric pathways may be found in the IWP (LANL 1998, ER ID 58605, p. 2-42). The role of climate in the atmospheric contaminant pathway will be considered as part of the site-specific scoping evaluation (see Section 4.2, Scoping Evaluation).

High-intensity thunderstorms in the summer can cause erosion of unstable sediment or soil. The form, frequency, intensity, and evaporation potential of precipitation can strongly influence surface water runoff and infiltration of contaminants (Section 3.2, Contaminant Fate and Transport).

3.1.3 Geology and Soils

Geology

Geologic and hydrologic information provides the basis for the discussion of hydrologic transport of contaminants. The likelihood of hydrologic transport is considered in the site-specific scoping evaluation (see discussion in Section 4.2, Scoping Evaluation). The geologic and hydrologic characteristics in and around the Laboratory as they relate to the potential for contaminant transport are complex. For a better understanding of these topics, please read the detailed discussion provided in Chapter 2 of the IWP (LANL 1998, ER ID 58605, pp. 2-6–2-21). Additional literature on the hydrology and geology of the Los Alamos region may be found in an annotated bibliography of geologic, hydrogeologic, and environmental studies related to solid waste management units of the Laboratory (LANL 1990, ER ID 47588).

The Laboratory extends over the east-sloping, dissected tableland of the Pajarito Plateau, and is bounded on the west by the eastern Jemez Mountains and on the east by White Rock Canyon of the Rio Grande. The geology of the Pajarito Plateau primarily reflects ancient volcanism in the Jemez Mountains and surrounding areas. The Rio Grande rift lies to the east of the plateau, forming a series of north-south trending fault troughs from southern Colorado to southern New Mexico. Most of the finger-like mesas in the Los Alamos area (Figure 3.1.3-1) are formed in Bandelier Tuff, which includes ash fall, ash fall pumice, and rhyolite tuff. The tuff is more than 1000 ft thick in the western part of the plateau and thins to about 260 ft eastward above the Rio Grande. It was deposited as a result of major eruptions in the Jemez Mountains' volcanic center about 1.2 to 1.6 million years ago. Deep canyons are incised into the Bandelier Tuff and expose it to depths of up to several hundred feet below the upper elevation of the plateau. Some of the deeper canyons expose older lava deposits and sedimentary rocks. Permeable units in the floors that outcrop below saturated alluvium create the potential for recharge to deeper ground water zones and form a source for springs and seeps in the area. Faults, cooling joints, and fractures potentially occur throughout the Pajarito Plateau (LANL 1998, ER ID 58605, pp. 2-21–2-23).

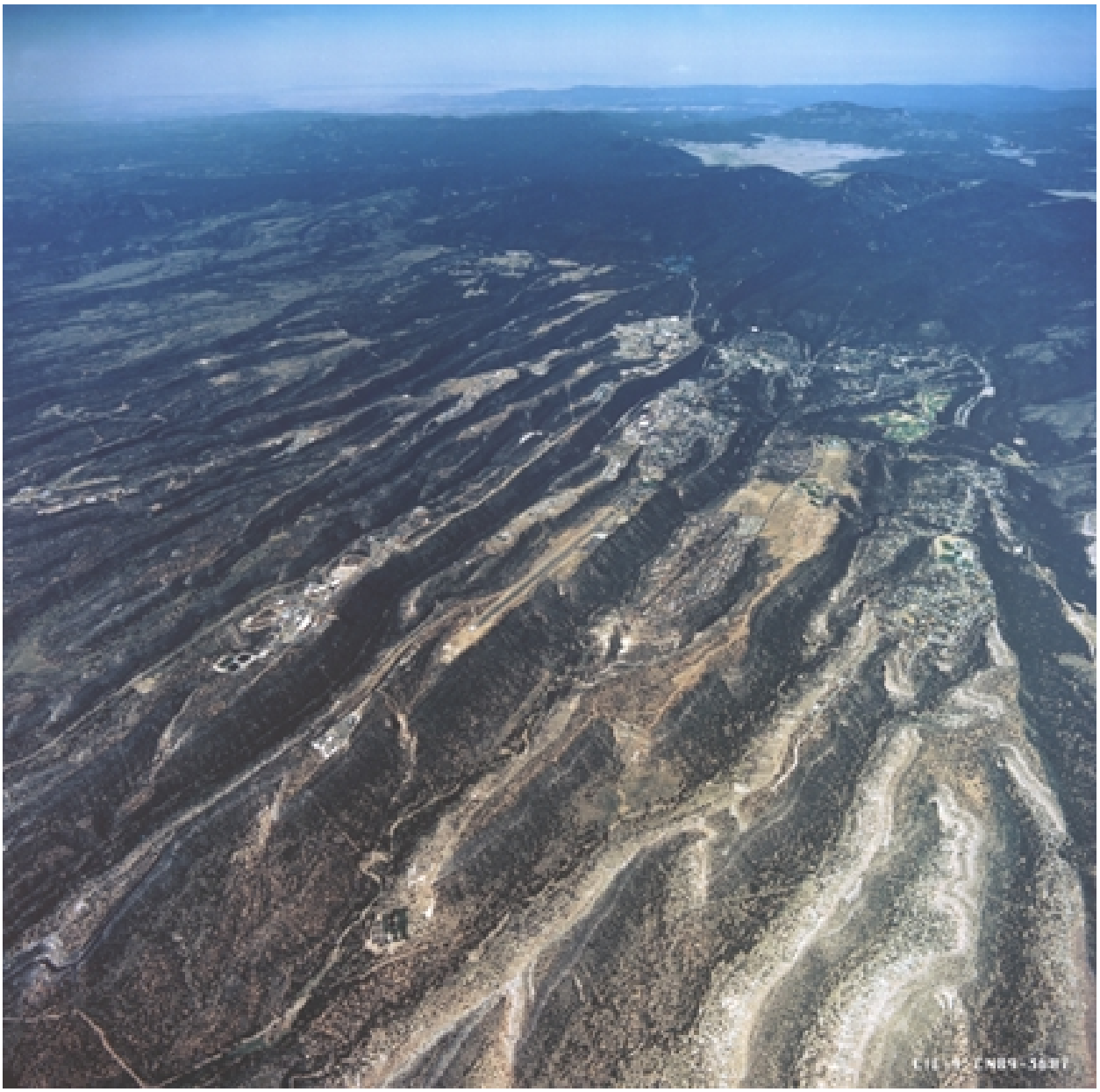


Figure 3.1.3-1. Topography of the Los Alamos area.

On the western part of the Pajarito Plateau, the Bandelier Tuff overlaps onto the Tschicoma Formation, which consists of older volcanic rock that composes most of the Jemez Mountains. The conglomerate of the Puye Formation in the central plateau and near the Rio Grande underlies the tuff. Chino Mesa basalts intertwine with the conglomerate along the river. These formations overlay the sediments of the Santa Fe Group, which extend across the Rio Grande Valley and are more than 3300 ft thick.

Most Laboratory facilities are located on tuff, which is covered by thin, discontinuous soils on mesa tops and alluvial deposits of variable thickness on canyon floors.

Soils

Soil erodability is an important consideration to help understand the potential for transport of contaminants. Therefore, a basic understanding of soil characteristics is important to the accurate completion of the "contaminant transport information" in site-specific scoping evaluation (see Section 4.2, Scoping Evaluation). Soils on the Pajarito Plateau were initially mapped and described by Nyhan et al. (1978, ER ID 05702). A large variety of soils and sediment have developed on the Pajarito Plateau as the result of interactions of the underlying bedrock, slope, and climate. Mesa tops may consist of soil derived from Bandelier Tuff, lavas, basalts, sedimentary rocks, and alluvium. Canyon floors generally contain poorly developed, deep, well-drained soils (Nyhan et al. 1978, ER ID 05702). General patterns of soil erosion rates are summarized by the following text from the IWP (LANL 1998, ER ID 58605, p. 2-23).

> Erosion rates vary considerably on the mesa tops; the highest rates occur in and near drainage channels and in areas of locally steeper slope gradient. The lowest rates occur on relatively gently sloping portions of the mesa tops removed from channels. Areas where runoff is concentrated by roads and other development are especially prone to accelerated erosion.

Mesa tops generally consist of finer textured soils and canyon bottoms consist of relatively coarse sediment. Overland runoff of contaminants is determined by erodability of soils that is influenced by primarily two factors—the amount of vegetative cover and the soil texture. Given constant vegetative cover for mesa tops and canyon bottoms, the finer textured soils of mesa tops are more subjective to overland runoff. An exception may be when intensive storm events drain canyons with large volumes of stormwater runoff.

3.1.4 Hydrology

Surface water in the Pajarito Plateau occurs as streams that are ephemeral (flowing in response to precipitation), intermittent (flowing in response to availability of snowmelt or groundwater discharge), perennial (flowing continuously), or interrupted (alternating perennial, ephemeral, and intermittent reaches). Some surface water arises from natural flows that originate in canyon heads in the upper Jemez Mountains north and west of the Laboratory. Other surface water originates from mesa top stormwater drainage and permitted Laboratory discharges. Perennial springs on the flanks of the Jemez Mountains supply base flow into the upper reaches of some canyons, but the volume is insufficient to maintain surface flows across the Laboratory site before they are depleted by evaporation, transpiration, and infiltration as discussed in the "Core Document for Canyons Investigations" (LANL 1997, ER ID 55622).

The Rio Grande is the highest order stream in north central New Mexico. Much of the surface water flow and groundwater discharge from the Pajarito Plateau canyon systems ultimately arrives at the Rio Grande through drainages that extend from the Laboratory in a southwest direction, but not as continuous flow. Only five canyons contain perennial reaches within Laboratory boundaries (Los Alamos, Pajarito Canyon, Water Canyon, Ancho Canyon, and Chaquehui Canyon). Sandia Canyon and Cañon de Valle are also suspected to have continuous flow in portions of their extent (Ralph Ford-Schmid, New Mexico DOE Oversight Bureau, personal communication).

Groundwater in the Los Alamos area occurs in three forms: (1) water in shallow alluvium in canyons, (2) perched water (a body of groundwater above a less permeable layer that is separated from the underlying regional aquifer by an unsaturated zone), and (3) the regional aquifer of the Los Alamos area. Groundwater hydrology for this region including the potential for contamination is complex. Section 2.2.2.2 of the IWP should be consulted for a detailed discussion of this subject (LANL 1998, ER ID 58605, p. 2-27).

3.1.5 Biology

The biota within the Laboratory includes approximately 500 plant species, 29 mammal species, 200 bird species, 19 reptile species, eight amphibian species, and hundreds of insect species (LANL 1998, ER ID 59904). Special consideration must be given to the protection of threatened and endangered (T&E) species and their habitat. Habitats for seven federally protected (LANL 1999, ER ID 62887) and five state-protected T&E species (Loftin and Haarmann 1998, ER ID 62881) have been identified at the Laboratory (LANL 1999, ER ID 62887). The federally listed species include the southwestern willow flycatcher (*Empidonax traillii extimus*), American peregrine falcon (*Falco peregrinus anatum*), arctic peregrine falcon (*Falco peregrinus tundrius*), whooping crane, bald eagle (*Haliaeetus leucocephalus*), black-footed ferret (*Mustela nigripes*), and Mexican spotted owl (*Strix occidentalis lucida*). Occupancy has been confirmed for only two federally listed species—the bald eagle and Mexican spotted owl (LANL 1999, ER ID 62887). The American peregrine falcon has historically longstanding aeries immediately adjacent to the Laboratory and forages on Laboratory lands. Results of preliminary risk assessments for the Mexican spotted owl, American peregrine falcon, bald eagle, and southwestern willow flycatcher are available in Gonzales et al. (1997, ER ID 62879), Gonzales (1998, ER ID 62349), and Gonzales (1998, ER ID 62350). Information on the biology and ecology of these species relevant to risk from contaminants can also be found in these references. State-listed species include the yellow lady's slipper (*Cyprepedium calceolus* var. *pubescens*), wood lily (*Lilium philadelphicum* var. *andinum*), Great Plains ladies-tresses, Jemez Mountains salamander (*Plethodon neomexicanus*), gray vireo (*Vireo vicinior*), spotted bat (*Euderma maculata*), and New Mexican meadow jumping mouse (*Zapus judsonius luteus*).More-detailed information on T&E species for the purpose of completing questions in Part B of the Scoping Checklist and for other risk assessment purposes may be found in LANL (1999, ER ID 62887) and Loftin and Haarmann (1998, ER ID 62881).

Knowledge of the vegetative communities at the Laboratory and the animal fauna found in association with these complexes is used in the ecological risk screening process for predicting the presence or absence of species at the site or in the surrounding areas. For example, areas containing mature, mixed conifer stands are important to Mexican spotted owls. Knowledge and expectations from biological assessments associated with the site are then used to identify potential pathways and exposures to ecological receptors, including T&E species.

The Laboratory has recently developed a vegetation land cover map (Figure 3.1.5-1) for the purpose of locating habitat that is suitable, or potentially suitable, for T&E species (Koch et al. 1997, ER ID 62882). The land cover map identifies areas by the dominant overstory vegetation. The map was developed using the Iterative Self-Organizing Data Analysis Technique to interpret a 1992 Landsat thematic mapper image into 30 classes. The 30 classes were then aggregated into 10 land cover types through field surveys, aerial photo interpretation, and the incorporation of topographic information. Random ground truthing was conducted using global positioning system (GPS) siting and following the National Heritage Program visual method for vegetation characterization. The resulting cover types include major vegetation zones and physiognomic types that are important to the distribution and abundance of several T&E species (Koch et al. 1997, ER ID 62882). The approximate areal extent of each cover type on Laboratory property is provided in Table 3.1.5-1. The ecologist or risk assessor who conducts scoping verifies the vegetation cover type during the site visit that supports the site-specific problem scoping (see Section 4.2, Scoping Evaluation).

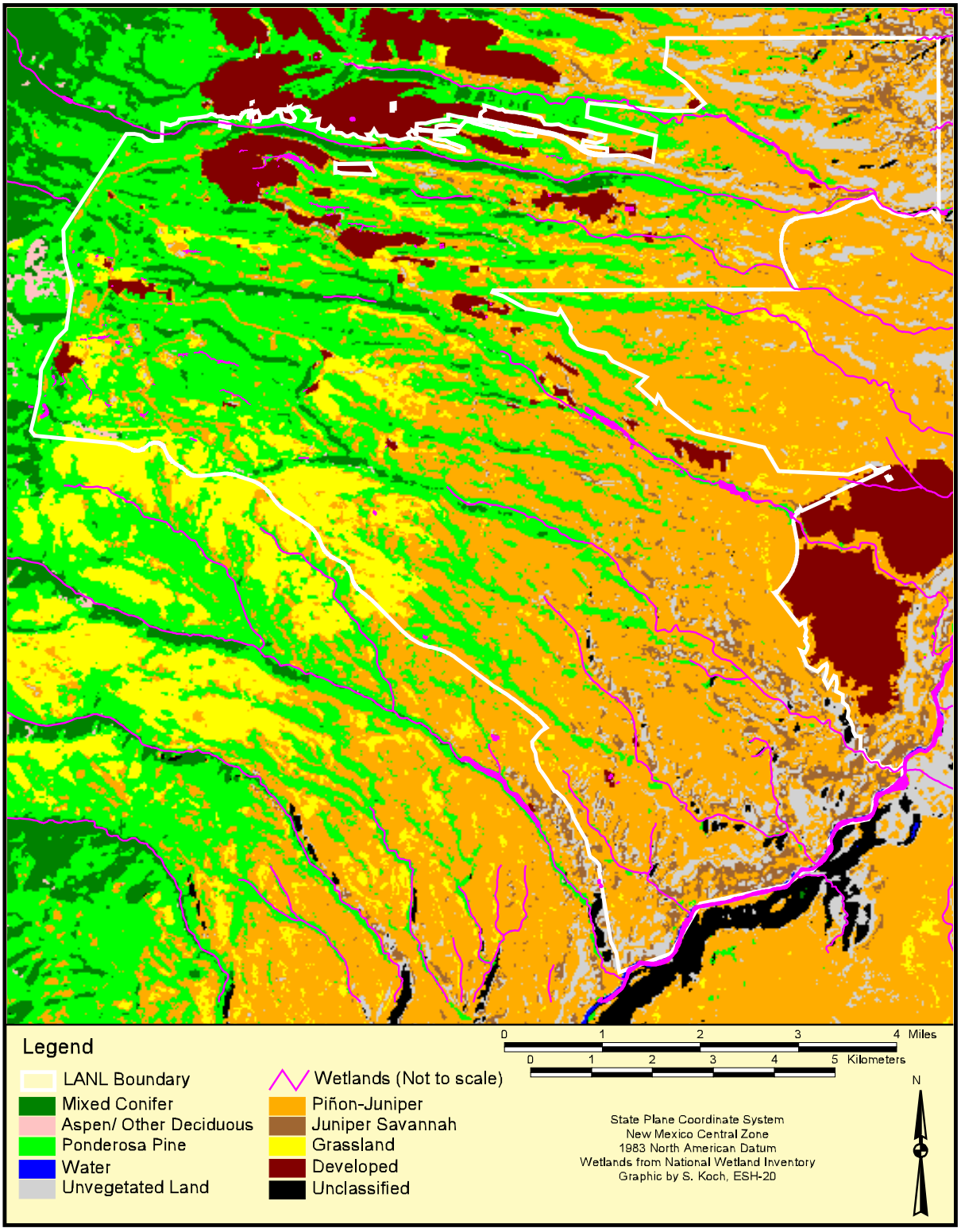


Figure 3.1.5-1. Vegetation land cover map.

Table 3.1.5-1.

Approximate areal extent of land cover types at the Los Alamos National Laboratory[a].

Cover Type	Area (mi^2)	Proportion of Total Area (%)
Mixed conifer	1.3	3.0
Aspen	0.1	0.2
Ponderosa pine	12.6	29.1
Piñon-juniper	20.0	46.2
Juniper savanna	1.6	3.7
Grasslands	2.9	6.7
Water	0.04	0.1
Wetlands	0.04	0.1
Unvegetated	2.9	6.7
Developed	1.6	3.7
Shadows	0.2	0.5
Total	43.3	100

a. Modified from Koch et al. (1997, ER ID 62882). An area of approximately 0.04 mi^2 of wetlands that were not detected by the Landsat was added.

The land cover types can be subdivided to correspond with the major elevation and climatic gradient of the region as well as edaphic, topographic, or moisture criteria (Koch et al. 1997, ER ID 62882). The elevation and climatic gradients in the region of the Laboratory most strongly influence distribution of five vegetative cover types defined by their dominant tree species and by their structural characteristics; these include juniper savannas, piñon-juniper woodlands, ponderosa pine forests, mixed conifer forests, and spruce-fir forests. In contrast, aspen forests, grasslands, shrublands, open water, and unvegetated lands are influenced less by elevation and climatic gradients. Instead, their distribution is most strongly influenced by topographic features, soils and geologic conditions, and moisture levels. Some areas are identified as shadows; steep terrain or clouds cause these shadowed areas.

Mixed conifer forests. Mixed conifer forests may be found above 2070 m (6900 ft) in elevation, blended with ponderosa pine communities, but also extend to lower elevations on north-facing slopes of canyons. These communities continue to the highest elevations of the Sierra de los Valles, 3149 m (10496 ft). Douglas fir and white fir (*Abies concolor*) are the typical overstory dominants in mixed conifer forests. At elevations above 2700 m (9000 ft), Engelmann spruce (*Picea engelmannii*) becomes more important. Ponderosa pine and aspen (*Populus tremuloides*) are also typically present. Limber pine (*Pinus flexilis*) can also be found in mixed conifer forests, especially on rocky ridgelines.

Aspen forests. Aspen (*Populus tremuloides)* communities are common at mid-elevations in the mountains, from approximately 2700 m to 3030 m (8900 ft to 9950 ft). Below 2820 m (9250 ft), aspen stands occupy north and northeast facing slopes, whereas above this elevation they are mostly found on southeast- to southwest-facing slopes. At higher elevations and on south-facing slopes, aspen typically exceeds 45% coverage and may be the only species present in the overstory. At lower elevations and on north-facing slopes, white fir, Engelmann spruce, and Douglas fir may collectively contribute up to 30% of the overstory coverage. Depending on the fire history of the specific stand, other tree species, such as ponderosa pine and limber pine, may be blended with aspen.

Grassland. Grasslands are dominated by grasses, narrow-leaf plants (e.g., yucca), and species that invade disturbed areas (colonizing species). Forbes and other non-shrubby species may be dominant

components of these communities. Shrubs and trees are absent or rare. The grassland cover type may include areas undergoing post-fire succession, abandoned homestead areas, montane meadows, and subalpine grasslands.

Open water. This cover type includes all land that is at least periodically flooded or is open water. In the wettest of these sites, the vegetative cover is limited to plant species that require or prefer permanent or seasonally mesic conditions. The Rio Grande borders the Laboratory on its eastern boundary and dominates the water component shown in Table 3.1.5-1.

Unvegetated land. This land cover type consists of all undeveloped land that is covered by less than seven percent vegetation. These land surfaces are dominated by cobbles, boulders, bedrock, or bare ground. This includes tuffaceous cliffs, basalt cliffs, felsenmeers, and basalt talus.

3.1.6 Wetlands

Wetlands are generally defined as areas of the environment containing water or moisture that support a host of aquatic plants and animals. More specifically, wetlands are defined on the basis of properties related to hydrophytes and hydrophilic plants, hydric soils, and the hydrology as described in 10 CFR 1022 (DOE 1979, ER ID 62888). In and around the Laboratory these systems occur primarily in the canyon bottoms of the Pajarito Plateau and along the banks of the Rio Grande. Wetlands may also be associated with effluent and stormwater outfalls from Laboratory and city facilities. Wetlands locations and areal coverage for 90% of the Laboratory have been determined using the GPS integrated with the geographic information system (GIS) (Bennett 1999, ER ID 62891). One hundred fifty-six wetland fragments have been identified at the Laboratory, comprising a total of approximately 49 acres (Sigler 1999, ER ID 62878), as shown in Table 3.1.5-1. The approximate locations of many of the larger wetlands are shown in Figure 3.1.5-1. Some of the larger wetlands on the Laboratory are located in upper Sandia Canyon (~6.1 acres), upper Pajarito Canyon (~13.2 acres), lower Pajarito Canyon (~2.0 acres), Mortandad Canyon, and Cañon de Valle (~1.5 acres).

The protection of wetland ecosystems at the Laboratory from the impacts of contaminants is especially important because of the diversity of associated fauna and because wetlands provide significant potential contaminant uptake pathways. These pathways include food web, direct media contact, and gamma radiation exposure pathways. Additionally, aquatic organisms occupying wetlands may experience higher exposures to contaminants because of continuous contact with water and specialized respiration mechanisms. Wetlands are of critical importance to both terrestrial and aquatic biota. Functional aspects of wetlands include food web contribution, breeding habitat, sediment retention, erosion prevention, flood and runoff storage, ground water recharge, and nutrient retention. A description of the diversity of species associated with wetlands at Laboratory and on their functional value may be found in the IWP (LANL 1998, ER ID 58605) and in DOE (1998, ER ID 62890).

3.2 Contaminant Fate and Transport

The geomorphology of the Pajarito Plateau, with its alternating mesas and canyons, determines the primary contaminant transport pathways for sources of legacy environmental contamination. Figure 3.2-1 is a schematic showing the key transport pathways:

- Hydrologic transport (e.g., surface water and groundwater)
- Physical transport (e.g., mass wasting of cliffs)
- Atmospheric transport (e.g., dust resuspension)

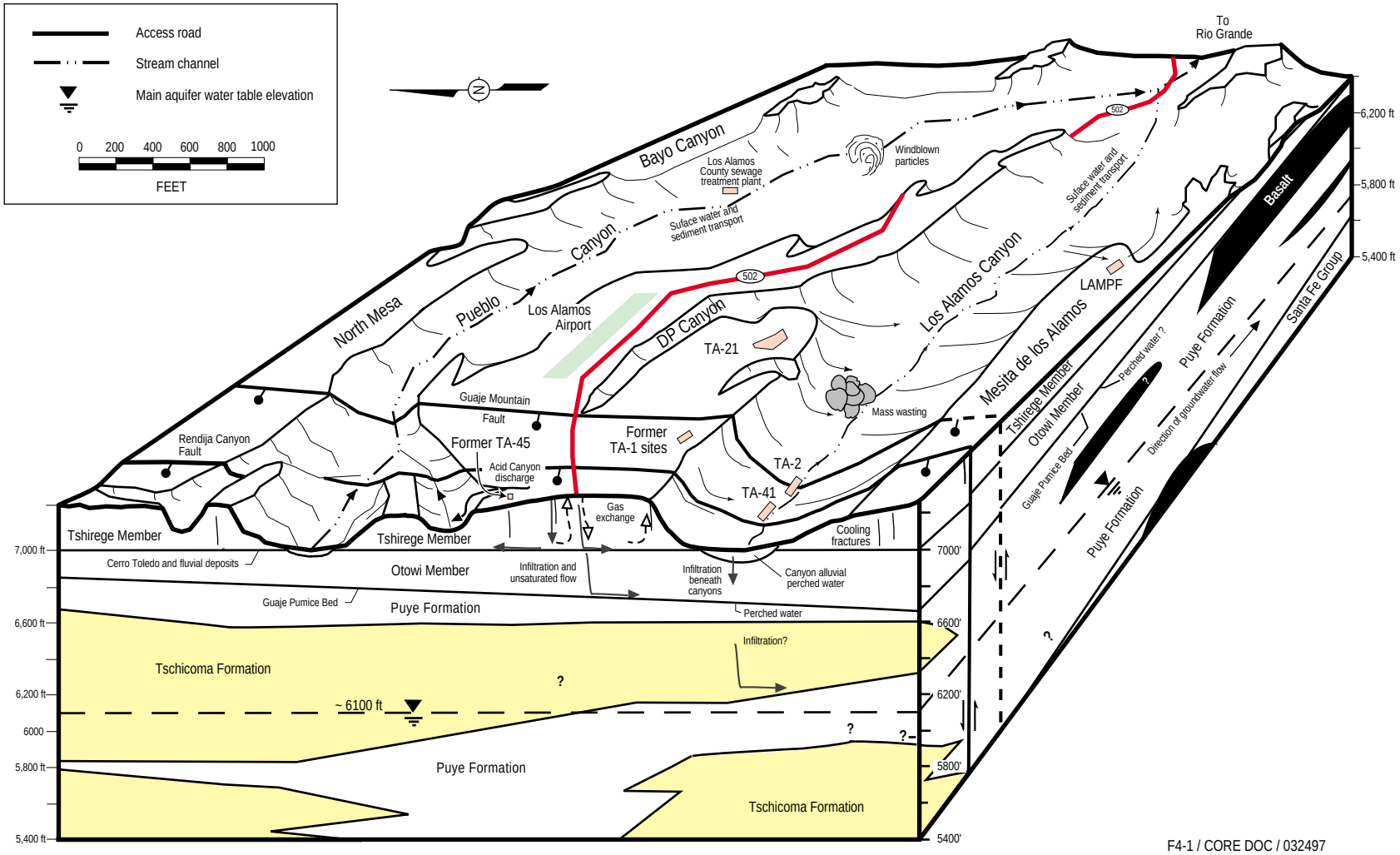


Figure 3.2-1. Key transport pathways.

These pathways are discussed briefly below. Pathways applicable to a particular site should be discussed in the applicable, site-specific reports.

3.2.1 Hydrologic Transport

3.2.1.1 Surface Water and Sediment Transport

Surface water flows provide the primary mechanism for redistributing and transporting the contaminants that remain from early Laboratory operations. The primary mechanisms affecting mobilization of contaminants within the canyons include sediment transport, contaminant dissolution and desorption, runoff, infiltration, and percolation. The water flowing through the Laboratory property, especially in canyon systems, is used by wildlife, constituting a major potential contaminant exposure pathway to these receptors.

Much of the surface water flow, including groundwater discharge from springs, from the Pajarito Plateau ultimately arrives at the Rio Grande. The Rio Grande annually transports about one million tons of suspended sediment to Cochiti Reservoir (LANL 1997, ER ID 55622). A more thorough description of canyon streams can be found in “Core Document for Canyons Investigations” (LANL 1997, ER ID 55622).

Sediment transport by surface water is believed to be the predominant mechanism for redistributing contaminants at the Laboratory. Carried by storm event runoff, contamination from mesa-top release sites could enter surface water drainages. Contaminants have also been released directly into stream channels by effluent discharges. Most environmental contaminants are adsorbed onto sediment particles, preferentially binding to particles with high surface areas and/or charged particles, such as silt and clay. The more soluble contaminants may remain in solution, which makes them available for vertical transport to perched aquifers and for later emergence in springs.

3.2.1.2 Groundwater Transport

The primary mechanism for contaminant transfer between the surface and underlying groundwater is infiltration of surface water carrying colloidal and dissolved contaminants (LANL 1997, ER ID 55622). The potential for significant infiltration from mesa-top settings is typically limited by the general lack of ponded water that might create hydraulic head. In canyon settings, however, the potential for significant infiltration exists, given the presence of perennial or intermittent surface water and coarse-grained sediments in most parts of the canyon systems and the high, vertical, hydraulic gradients beneath canyon streams.

Saturated groundwater zones beneath the Pajarito Plateau may be recharged in part by the vertical migration of water from canyon-floor alluvium. The vertical migration of alluvial groundwater may be partly directed and accelerated by faults and fractures. The role of faults and fractures as components of the hydrologic system, however, is poorly understood at this time. Unsaturated zones are considered only an occasional transport pathway.

3.2.2 Physical Transport

Physical transport of surface or subsurface materials is most dramatically possible through a mechanism termed *mass wasting.* Mass wasting is the process in which blocks of soil and rock break off the cliffs and are deposited violently into the canyons. Mass wasting is an episodic phenomenon and could be an important mechanism of contaminant transport for mesa-top sites located near canyon walls. Exposure to ecological receptors would result if subsurface contamination became surficial contamination through

mass wasting into the canyons. The transport pathways would then be similar to media subject to surface water transport. A much more gradual physical transport mechanism is surficial erosion through wind or water (Sections 3.2.1.1, Surface Water and Sediment Transport, and 3.2.3, Atmospheric Transport).

3.2.3 Atmospheric Transport

Atmospheric transport may occur through transport of windblown particles or vaporization of volatile chemicals. Transport of soil or fine sediment particles by wind can be a means of dispersing contaminants. Wind resuspension and transport of surficial contaminant-laden soil or sediment is not believed to be a significant transport pathway based on the small volume of contaminated media mobilized by this pathway. However, there may be exceptions for short periods of time at some sites where special situations exist (e.g., burrowing animals moving subsoil to the surface).

3.3 Exposure Pathways

Contaminants associated with surface soil can be available for biological receptors through the following exposure pathways:

- Rain splash or saltation-creep of contaminated soil or sediment onto plants
- Root uptake of water-soluble contaminants from surface water or shallow alluvial groundwater
- Incidental ingestion of soil
- Dermal contact with soil
- Food web transport (consumption of contaminated plants and animals)
- Direct exposure to soil containing gamma-emitting radioactive contaminants

Contaminants that are associated with sediments or surface water can be available for uptake by biota primarily through the following exposure pathways:

- Ingestion of surface water
- Root uptake of surface water
- Incidental ingestion of sediments
- Dermal contact with surface water or sediments
- Exposure to aquatic animals through respiration
- Inhalation by animals of fine sediment materials during dry periods
- Food web transport (consumption of contaminated plants and animals)
- Direct exposure to sediments containing gamma-emitting radioactive contaminants

When groundwater becomes surface water in springs or seeps, the previous exposure pathways also apply. In addition, shallow groundwater, particularly alluvial water, may be taken up by deep-rooted plants (e.g., chamisa) and enter the food web primarily through the ingestion of contaminated plants.

Contaminants present in air are available for uptake by biota through the following exposure pathways:

- Inhalation by animals during activity above ground or in burrows of contaminants present as vapors
- Respiration by plants of contaminants present as vapors
- Inhalation of particulates by animals during above ground activity or while in burrows
- Deposition of particulates on foliage
- Deposition of particulates on animals, and subsequent ingestion during grooming

3.4 Functional Food Web

A food web diagram is important for evaluating dietary exposure pathways and for specifying ecologically relevant groups of organisms for an exposure assessment. The food web structure captures functionally relevant biotic assimilation and associative relationships and is key for receptor selection. A food web diagram also shows pathways of food consumption in a biotic system by means of boxes and connecting arrows. Boxes in a food web diagram represent biota, e.g., functional assemblages or taxonomic groups, and arrows define the major direction of energy flow between biota, e.g., from prey to predators.

For the purposes of this ecological screening-level risk assessment methodology, it is more useful to design a food web where biological receptors are classified into functional groups with similar feeding roles instead of a taxonomic classification. Taxonomically based food webs use phylogenetic classification to organize species into evolutionarily related natural assemblages (genera, families, orders) and are insensitive to potentially similar feeding habits among these taxa. Figure 3.4-1 represents the functional food web for the Laboratory. The food web is organized into functional guilds based on feeding (trophic) relationships. Thus, a *feeding guild* is a collection of species sharing common food consumption roles. For example, animals that eat seeds (granivores) can be considered one feeding guild, browsers/grazers another, and top carnivores yet another. There are many ways to organize feeding guilds, from general to specific, but they are too numerous to illustrate here.

A food web organized by feeding guilds forms a basis for selection of individual species from each guild that can act as representatives of the guild as a whole. This approach formed the basis of receptor selection for this approach to ecological screening assessments at the Laboratory. The food web for the Laboratory includes three fundamental trophic positions. These are producers (vascular and non-vascular plants), consumers (herbivores, omnivores, carnivores, and parasites), and decomposers. Therefore, a minimum of three receptors is needed to represent the primary trophic associations. Within these basic trophic levels, several feeding guilds have been identified. For example, one group of consumers is herbivores, consisting of six feeding guilds: seed eaters (granivores), fruit-eaters (frugivores), foliage or leaf-eaters (folivores), nectar and pollen feeders (nectarivores/pollen eaters), fungi eaters (fungivores), and browser/grazers. Since the Laboratory food web included multiple levels of organization, it was necessary to choose receptors that were broadly representative of these levels. Note that dashed lines on Figures 3.4-1 and 3.4-2, enclosing a number of guilds in a single rectangle, represent broad categories for which a single member may suffice as a screening receptor. Receptor selection and the process are considered in more detail in Section 3.5, Screening Receptors.

Figure 3.4-1 is considered a general food web for the Laboratory, that is, it applies to both the terrestrial and aquatic environments. For the sake of completeness, we also provide Figures 3.4-2 and 3.4-3, which are specific to the terrestrial and aquatic environs, respectively, at the Laboratory. The illustration of the aquatic food web, separate from the terrestrial, will become important in understanding the discussion of receptor selection for aquatic environs in Section 3.5, Screening Receptors.

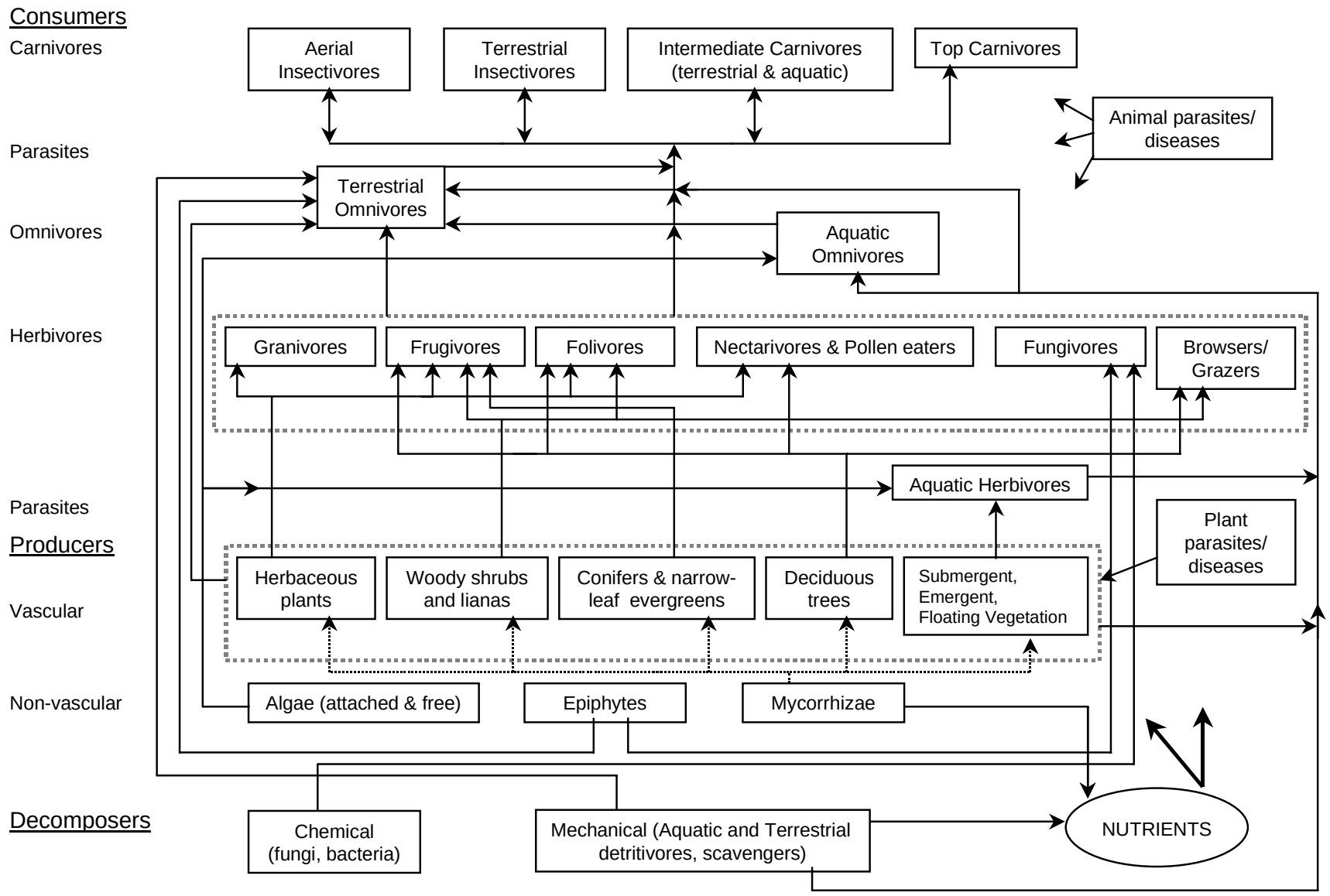


Figure 3.4-1. General food web based on feeding relationships of the biota at the Laboratory.

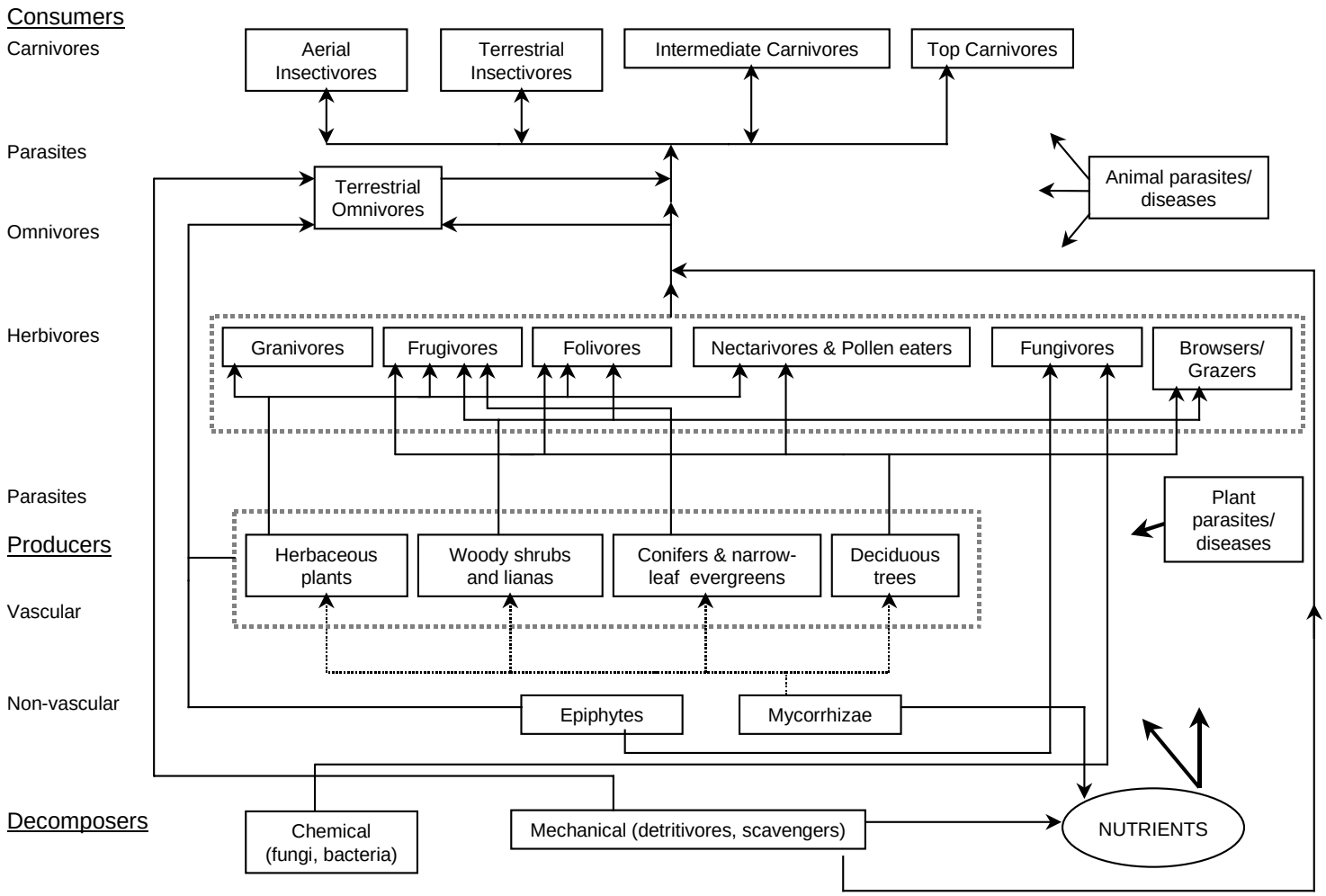


Figure 3.4-2. Terrestrial food web based on feeding relationships of the biota at the Laboratory.

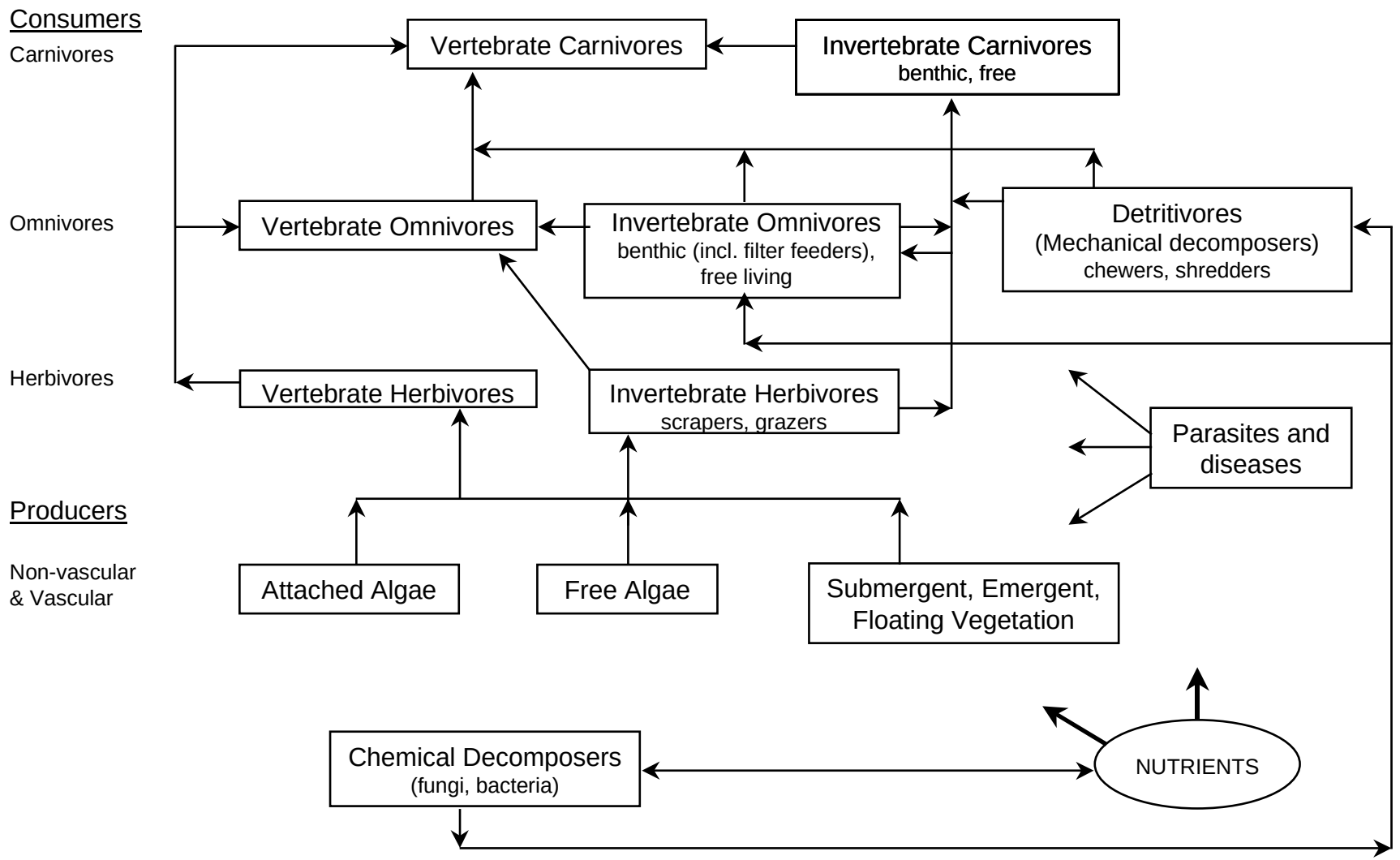


Figure 3.4-3. Aquatic food web based on feeding relationships of the biota at the Laboratory.

3.5 Screening Receptors

As described in Section 3.1, Environmental Setting, Laboratory property supports numerous habitats with a variety of plant and animal species. The selection of a set of receptors that includes representatives of every class of biota for every trophic level would result in an unwieldy number of receptors for use in ecological screening. Therefore, the rationale behind receptor selection is to select an appropriate set of receptors that address the primary feeding relationships outlined in Section 3.4, Functional Food Web. Receptor selection will facilitate the determination of potential adverse ecological impacts across the Laboratory and satisfy the following criteria (based on Fordham and Reagan 1991, ER ID 63081):

- The receptor is representative of an exposure pathway, including dietary pathways specified in the functional food web, and nondietary exposure pathways.
- The receptor is representative of a major feeding guild as defined in the functional food web.
- Protection of the receptor is protective of the integrity of ecosystem structure and function.
- The receptor is representative of potentially exposed populations or communities.
- Protection of the receptor is protective of promulgated T&E and other species of special interest or concern.
- Toxicity information is available that suggests the receptor is sensitive to contaminants from legacy waste at the Laboratory.
- Exposure information for the species is available.

Given these criteria, the selection of receptors for the Laboratory is outlined below. The selection of terrestrial receptors, including those that are considered in the aquatic food chain, follows directly from the above logic. The selection of aquatic receptors for radiological contamination is also in direct accord with the logic provided. For non-rad contaminants in aquatic environs, however, the Laboratory has chosen methods, recommended by New Mexico Environment Department (NMED) and Environmental Protection Agency (EPA) Region VI representatives at a meeting held at NMED on January 20, 1999, that are more broadly protective of aquatic ecosystems. These methods include the use of water and sediment benchmarks in ecological screening assessments for aquatic environments. The application of these benchmarks is targeted at the protection of roughly 95% of all aquatic organisms, and thus is inclusive of all trophic guilds illustrated in Figure 3.4-3. The use of benchmarks for screening aquatic environments is recommended in EPA guidance (EPA 1996, ER ID 62792). The details for the selection and application of water and sediment benchmarks are described in Sections 4.4.2, Sediment ESLs, and 4.4.3, Water ESLs. However, these benchmarks do not cover exposure to radionuclides. Specific aquatic receptors are needed to evaluate effects from radionuclides.

Terrestrial Receptors

Table 3.5-1 summarizes the factors that led to the selection of the seven terrestrial screening receptors. The use of a “generic” plant is indicative of the broad-base taxonomic concern for plants in general, rather than any particular species. The generic plant is also used to represent several plant species of special concern present at the Laboratory. Additionally, plants are primary producers and form much of the physical habitat structure used by animal species. By settling on a generic plant, we have chosen a very broadly protective view of the methods for development of ecological screening levels, discussed in Section 4, Site-Specific Screening Level Ecological Risk Assessment.

Table 3.5-1.

List of receptor species selected for screening at the Laboratory.

Receptor Species	Receptor Category	Selection Factors
Generic plant	Terrestrial autotroph (producer)	Food source for many animals Provides habitat structure and functional base for terrestrial animals Represents culturally important plants Representative of T&E plant species Direct exposure to contaminated soil Representative of all terrestrial plant species.
Earthworm	Soil-dwelling invertebrate	Represents decomposer group, which are important for nutrient cycling Large body of toxicity data Direct exposure to contaminated soil and detritus Represents a food source Representative of all soil-dwelling invertebrates
Desert cottontail	Mammalian herbivore	Food source for carnivores Ubiquitous and abundant Exposure data and toxicity data available Surrogate for economically important browsers (deer and elk)
Deer mouse	Mammalian omnivore	Food source for carnivores Ubiquitous and abundant Exposure data and toxicity data available Surrogate for T&E (New Mexico Meadow Jumping Mouse)
Vagrant shrew	Mammalian insectivore	Food source for carnivores High fraction of soil in diet relative to rabbit and deer mouse Diet is 100% invertebrates and thereby maximizes this potentially bioaccumulative exposure pathway Surrogate for T&E (Jemez Mountain Salamander)
American robin	Three diets modeled: Avian omnivore Avian herbivore Avian insectivore	Food source for some carnivores Ubiquitous and abundant Exposure data available High fraction of soil in diet
American kestrel	Two diets modeled: Intermediate Carnivore Top Carnivore	Surrogate for peregrine falcon and Mexican spotted owl by assuming 100% flesh diet Ubiquitous Exposure data available Addresses potential biomagnification from soils Conservative choice for this category, given the food intake to body weight ratio (see Section 4.3)
Red fox	Top carnivore	Exposure data available Addresses potential biomagnification from soils Conservative choice for this category, given the food intake to body weight ratio (see Section 4.3)
Algae (for radionuclides only)	Aquatic autotroph (producer)	Food source for aquatic animals Provides structure (substrate) for animals Ubiquitous and abundant Exposure and toxicity data available

Table 3.5-1. (continued)
List of receptor species selected for screening at the Laboratory.

Receptor Species	Receptor Category	Selection Factors
Daphnids (for radionuclides only)	Aquatic omnivore/herbivore	Food source for carnivores High exposure to contaminated water and sediment Ubiquitous and abundant Exposure and toxicity data available *Daphnia* and *Cerodaphnia* are typically the most sensitive aquatic organisms for a variety of contaminants
Aquatic snails (for radionuclides only)	Aquatic herbivore (grazer)	Food source for some carnivores (e.g., fish) High exposure to contaminated sediment Ubiquitous and abundant Exposure and toxicity data available
Fish (for radionuclides only)	Intermediate carnivore	Representative of potential waterborne contaminant effects in the Rio Grande High potential exposure to contaminants; potentially sensitive to persistent bioaccumulators and biomagnifiers.
Occult little brown myotis bat	Mammalian aerial insectivore	100% diet may be assumed to come from emergent aquatic insects Allows the consideration of bioaccumulation from aquatic sources to a high level mammalian receptor.
Violet-green swallow	Avian aerial insectivore	100% diet may be assumed to come from emergent aquatic insects Allows the consideration of bioaccumulation from aquatic sources to a high level avian receptor.

The earthworm (terrestrial worms of the subclass *Oligochaeta*) was selected because it represents the functional category of mechanical decomposers, which are important for nutrient cycling. In addition, earthworms have a higher exposure to contaminants than other invertebrates because of the earthworm's high soil intake and intimate soil contact. The earthworm is considered generally protective all terrestrial invertebrate species, including insects, arachnids, crustaceans, and other taxa.

The desert cottontail *(Sylvilagus audubonii)* was selected because it is a strict herbivore (browser/grazer), and can be used as a functional surrogate to evaluate potential effects on large mammalian browsers/grazers (e.g., deer and elk). The deer mouse *(Peromyscus maniculatus)* was selected because of its omnivorous food habits, and largely to represent the importance of rodents as a food source for higher consumers (carnivores and omnivores), making it important in the functional food web. The vagrant shrew *(Sorex vagrans)* was selected largely because of its high exposure to contaminants from grubbing for invertebrates in soil and because of its high-level intake of soil-dwelling invertebrates (including earthworms). The vagrant shrew also acts as a good receptor when considering a food chain model that includes bioaccumulation of contaminants from soil. The red fox *(Vulpes vulpes)* was selected because it represents a mammal with relatively high contaminant biomagnification potential due to its largely carnivorous feeding habits.

The American robin *(Turdus migratorius)* was selected because it is representative of birds that forage for ground-dwelling invertebrates, as well as fruits, with relatively high potential exposure to contaminants from its diet. The American robin can be considered in several functional roles for avian receptors: an insectivore, herbivore, and omnivore (invertebrate/plant). The American kestrel *(Falco sparverius)* was selected as a top avian carnivore because it serves well as a representative of T&E bird species at the

Laboratory, especially the peregrine falcon *(Falco peregrinus)* and the Mexican spotted owl *(Strix occidentalis lucida)*. Additionally, there is abundant information gathered for the kestrel's biology. Furthermore, the kestrel represents an organism with high susceptibility to contaminant biomagnification via terrestrial pathways.

The little brown myotis bat (*Myotis lucifugus occultus*) and the violet-green swallow (*Tachycineta thalassina lepida)* were chosen as receptors for modeling the effects of contaminants bioaccumulated from sediments to insects to aerial insectivores. The former is a species of special concern and considered rare in the Jemez mountains, although it has been trapped on Laboratory grounds. The brown myotis bat has a high fraction of its diet comprised of emergent aquatic insects, as the habitats surrounding water are favorite hunting haunts. The violet-green swallow is common on Laboratory grounds and does have some portion of its diet comprised of emergent aquatic insects, although its feeding habits are less specialized than that of the brown myotis bat. Nonetheless, both aerial insectivores may be modeled for maximum uptake of aquatic sediment borne contamination, and there is a wealth of information available for their general biology.

All terrestrial receptors were selected partially on the basis of information available regarding life history habits of the same or similar species (e.g., *Wildlife Exposure Factors Handbook*, EPA 1993, ER ID 59384).

Aquatic Receptors

No specific aquatic receptors were chosen for the screening assessment of non-radiological contaminants. Methods adopted for screening are considered by the EPA (e.g., EPA 1995, ER ID 62787; EPA 1996, ER ID 62792) and others (e.g., Jones et al. 1997, ER ID 62789) to be protective of a large fraction (roughly 95%) of aquatic organisms at large (plants, invertebrates and vertebrates). Although there are few vertebrates that reside in the aquatic realms of the Laboratory (some herpetiles), it was considered prudent to adopt methods that are otherwise considered pervasively protective and which include organisms that may be found in the Rio Grande (e.g., fish). The aquatic food web, as shown in Figure 3.4-3, is useful for the organization of the scoping portion of screening, but for contaminant-based ecological screening comparisons, methods employed broadly cover all species represented in all trophic guilds.

Four aquatic receptors were selected for screening exposure to radionuclides. Algae were selected to represent the producer functional group. Daphnids *(Crustacea)* and snails *(Gastropoda)* were selected to represent the aquatic omnivore and herbivore functional subgroups. The daphnid's diet in freshwater systems consists primarily of phytoplankton and zooplankton, while snails typically obtain food from scraping lithic and vegetative surfaces for incidental free and attached algae. Some daphnids, e.g., *Daphnia* and *Cerodaphnia*, represent the most sensitive aquatic organisms to most environmental contaminants. Lastly, a "generic" bony fish was selected to represent intermediate carnivores. There is no direct representative for the Jemez Mountain Salamander, an endangered species with both aquatic and terrestrial life stages. Juvenile salamanders are associated with water, while adults inhabit terrestrial environments. Adult Jemez Mountain Salamanders are invertebrate consumers, and can be considered functionally similar to shrews and are therefore covered by terrestrial screening procedures. We assume that juvenile salamanders or other amphibians are represented by the aquatic herbivore and omnivore receptors described above.

3.6 Assessment Endpoints

Superfund guidance states that for the screening-level assessment, assessment endpoints are any adverse effects on ecological receptors, where receptors are populations and communities, habitats, and sensitive environments (EPA 1997, ER ID 59370). Following the Superfund guidance, the Laboratory's assessment endpoints are adverse effects on receptor populations, and adverse effects on these populations can be inferred from endpoints related to impaired reproduction, growth, and survival (EPA 1997, ER ID 59370). These endpoints will be considered in the identification and evaluation of appropriate toxicity information. The toxicity information is discussed further in the next section, and readers are pointed to Section 4.3, Screening Evaluation Overview (Ecological Effects of Concern for Screening and Dose-Response Model).

4.0 SITE-SPECIFIC SCREENING LEVEL ECOLOGICAL RISK ASSESSMENT

Before conducting the screening-level ecological risk assessment, there is a pre-scoping phase where administrative no further action (NFA) proposals are documented and the chemicals of potential concern (COPCs) for the assessment are developed. Pre-scoping activities are typically reviewed for ecological risk issues.

The screening-level ecological risk assessment consists of three steps:

1. The scoping evaluation (or problem formulation phase described in Section 4.2);
2. The screening evaluation (or the screening-level risk and uncertainty analysis phase described in Sections 4.3 to 4.6); and
3. Risk interpretation (or screening-level risk characterization described in Section 4.7).

4.1 Pre-scoping Activities

4.1.1 Identify Sites for Administrative NFA

The first step of the pre-scoping evaluation is to determine if the PRS is a candidate for an administrative NFA based on the following New Mexico Environment Department (NMED) criteria (NFA Criteria are listed in Section II.B.4.a.(4).(b), "No Further Action (NFA) Proposals Criteria," in the *NMED RCRA Permits Management Program Document Requirement Guide* (NMED 1998, ER ID 57897).

- NFA Criterion 1. The Solid Waste Management Unit/Area of Concern (SWMU/AOC) cannot be located, does not exist or is a duplicate SWMU/AOC.
- NFA Criterion 2. The SWMU/AOC has never been used for the management (i.e., generation, treatment, storage and/or disposal) of Resource Conservation and Recovery Act (RCRA) solid waste or hazardous wastes and/or constituents or other Comprehensive Environmental Response, Conservation and Liability Act (CERCLA) hazardous substances.
- NFA Criterion 3. No release to the environment has occurred or is likely to occur in the future from the SWMU/AOC. (For an administrative NFA this is generally based on process knowledge, rather than environmental sample information.)

Environmental Restoration (ER) Project personnel provide the justification for these NFA recommendations. Environmental sample information is not required, and further ecological evaluations are unnecessary. If the site is not an administrative NFA, a Resource Conservation and Recovery Act (RCRA) facility investigation (RFI) is conducted and data are collected to determine if the site poses a potential threat to human health or the environment. The site visit and scoping checklist described in Section 4.2.1 can be used to guide the data collection process.

4.1.2 Identify COPCs based on Data Review

After the RFI (or equivalent investigation), the data are assessed to determine if there are COPCs in samples of environmental media collected from the site. COPCs are typically inorganic chemicals and radionuclides measured greater than background concentrations, and detected organic chemicals. Details of the data review process are provided in the "Resource Conservation and Recovery Act Facility

Investigation Report, Los Alamos National Laboratory, Annotated Outline" (LANL 1998, ER ID 58981) and are not repeated here. If there are no COPCs identified during data review and data were adequate to detect a release, then the potential release site (PRS) may be recommended for NFA and no further ecological evaluation is required. The ER Project personnel (including data analysts, chemists, statisticians, and risk assessors) are involved in the data review process. However, the conclusion that adequate data were collected to document no release (i.e., no COPCs) is provided by statisticians, chemists and other data analysts. Those sites at which COPCs are identified require ecological scoping, including completion of the scoping checklist, which requires a site visit by a member of the ecological risk assessment task team.

4.2 Scoping Evaluation

The goals of the scoping evaluation are to identify those sites that need a screening evaluation, assess the need for an aggregate assessment, identify COPCs, determine data adequacy for screening, evaluate the potential for environmental contaminant transport, and establish likely exposure pathways. The scoping evaluation is equivalent to the site-specific problem formulation step.

4.2.1 Scoping Checklist

The purpose of the scoping checklist is to provide information to

- Describe the site setting and the known form of contaminant releases;
- Confirm that complete exposure pathways to ecological receptors exist;
- Determine if the site should be combined with other sites for screening and establish the functional/operational boundaries of the assessment;
- Identify the need for an interim action through a scientific management decision point (SMDP);
- Determine if adequate quality and quantity of data exist for the screening evaluation, primarily as related to nature, rate, and extent of contamination;
- Prepare for screening evaluation by determining whether screening should encompass terrestrial and/or aquatic receptors; and
- Gather information to develop the site conceptual model (SCM) (e.g., what are the dominant/important transport pathways, exposure routes, and receptors).

Completion of the ecological scoping checklist consists of three steps, detailed in Sections 4.2.1.1 through 4.2.1.3:

1. Assembling and initially interpreting information on the nature of releases, site history and operations, potential for off-site transport, and biological receptors potentially impacted by releases.
2. Visiting the site to validate information from (1) and collecting field notes for completing the site conceptual model. The site visit can be used to document the presence or lack of receptors and off-site migration pathways. Notes are also made regarding the applicability of existing data for determining the nature, rate and extent of contamination. Specific attention is paid to the likelihood that the sample maximum represents the highest contaminant concentrations.

3. Completing the SCM diagrams identifies the complete and incomplete exposure pathways as well as the major and minor pathways.

4.2.1.1 Checklist Step 1: Assemble Existing Information

To prepare for the site visit, the following information should be obtained: (1) the most current biological assessment information for the site (typically the Biological and Floodplain Assessment for applicable operable unit [OU] and/or technical area [TA]); (2) information on site erosion potential; (3) RFI work plan or report, as applicable, that provides information on contamination source, sample locations, analytical suites, and sample results; (4) Facility for Information Management, Analysis, and Display (FIMAD) geographic information system (GIS) maps that show (if applicable) neighboring PRSs, sample locations, vegetation types, watershed name, and wetlands; and (5) historical and current aerial photographs to help document changes in site operations and conditions.

Prior to the site visit, discussion of the existing information for the site, through a structured review of history and status of relevant PRSs, is often necessary. The results of the meeting (or equivalent) will be documented in Part A of the Ecological Scoping Checklist (Appendix A). The information required for Part A of the Ecological Scoping Checklist includes: (1) site identification; (2) nature of PRS releases (solid, liquid, gaseous, or other); (3) a list of the primary impacted media (soil, water/sediment, subsurface [greater than 3 ft depth], or other); (4) specification of the applicable FIMAD vegetation classes (water, bare ground, spruce/fir/aspen/mixed conifer, ponderosa pine, piñon juniper/juniper woodland, grassland/shrubland, and developed; (5) identification of threatened and endangered (T&E) habitat, if present (list species if applicable); (6) a list and description of neighboring/contiguous/upgradient PRSs (discuss whether it is necessary to aggregate the site with additional PRSs for screening); (7) Standard Operating Procedure (LANL-ER-SOP) 2.01 information (runoff score and the terminal point of surface water transport[3]); and (8) documentation of other scoping meeting notes (as appropriate).

The project manager for the site will be responsible for arranging the scoping meeting before the site visit, if needed. Scoping meeting participants should include the project manager, ecological risk assessor, ER Project regulatory compliance interface, and other site subject matter experts as necessary (such as a soil scientist, chemist, biological resources expert, geohydrologist, and field sampling personnel).

4.2.1.2 Checklist Step 2: Site Visit

The main objective of the site visit is to affirm whether ecological receptors are present and can be exposed to site contaminant releases. A secondary objective is to evaluate whether site data provide adequate information to determine the nature, rate, and extent of contamination. The site visit should be arranged at an appropriate time of year (ideally spring or summer) to best evaluate biota at the site. If the site visit is planned for another time of year, uncertainties introduced in the initial biological assessment by such timing must be noted.

Maps showing sample locations and results and a camera are always needed for the site visit. The need for other equipment or supplies to locate and measure site features should be determined during the scoping meeting. Such additional resources may include a measuring device to roughly locate relevant

[3] LANL ER-SOP 2.01 (previously known as LANL AP-4.5) provides PRS-specific information on the potential for erosion from surface water. This document considers factors that relate to the cover (vegetative and non-vegetative), slope, surface water run-on sources, and other related factors. This document provides a total erosion score, but only the runoff score and the terminal point of surface water discharge are relevant to ecological risk scoping. If the runoff score is zero, then no erosion potential is apparent for the PRS. The terminal point of surface water discharge helps to determine if erosion from the PRS reaches aquatic settings.

biological features (measuring tape and/or rangefinder and pin flags or other markers to specify locations for surveying.

Part B of the checklist is to be completed during the site visit and includes administrative information such as the site identification, date of site visit, and personnel conducting visit. Part B also includes receptor information, primarily aimed at determining if ecological receptors are present at the site. Contaminant transport information, emphasizing surface water transport and noting if there are other modes of transport is documented in Part B. Part B also provides ecological effect information, including notes on physical disturbance and obvious ecological effects (such as dead vegetation or lack of fossorial faunal activity). An important component of Part B is the consideration of a recommendation for an IA SMDP to possibly reduce the potential for off-site migration of contaminants to ecological receptors or mitigate exposure of on-site receptors to contaminants. In the interim action (IA) SMDP the site project manager involves the appropriate technical and management personnel in an evaluation of interim actions for the site.

If there are no complete pathways to receptors and no transport pathways to off-site receptors, the remainder of the checklist (last part of Part B and Part C) is not completed. The checklist is stopped at this point and any additional explanation/justification is provided to conclude that the site poses no threat to the environment. An example of “no pathways/no receptors” is a mesa top site with buried, inaccessible contamination with no potential for offsite transport. However, a site that lacks receptors because of high levels of contamination would not qualify for the “no pathways/no receptors” stopping point.

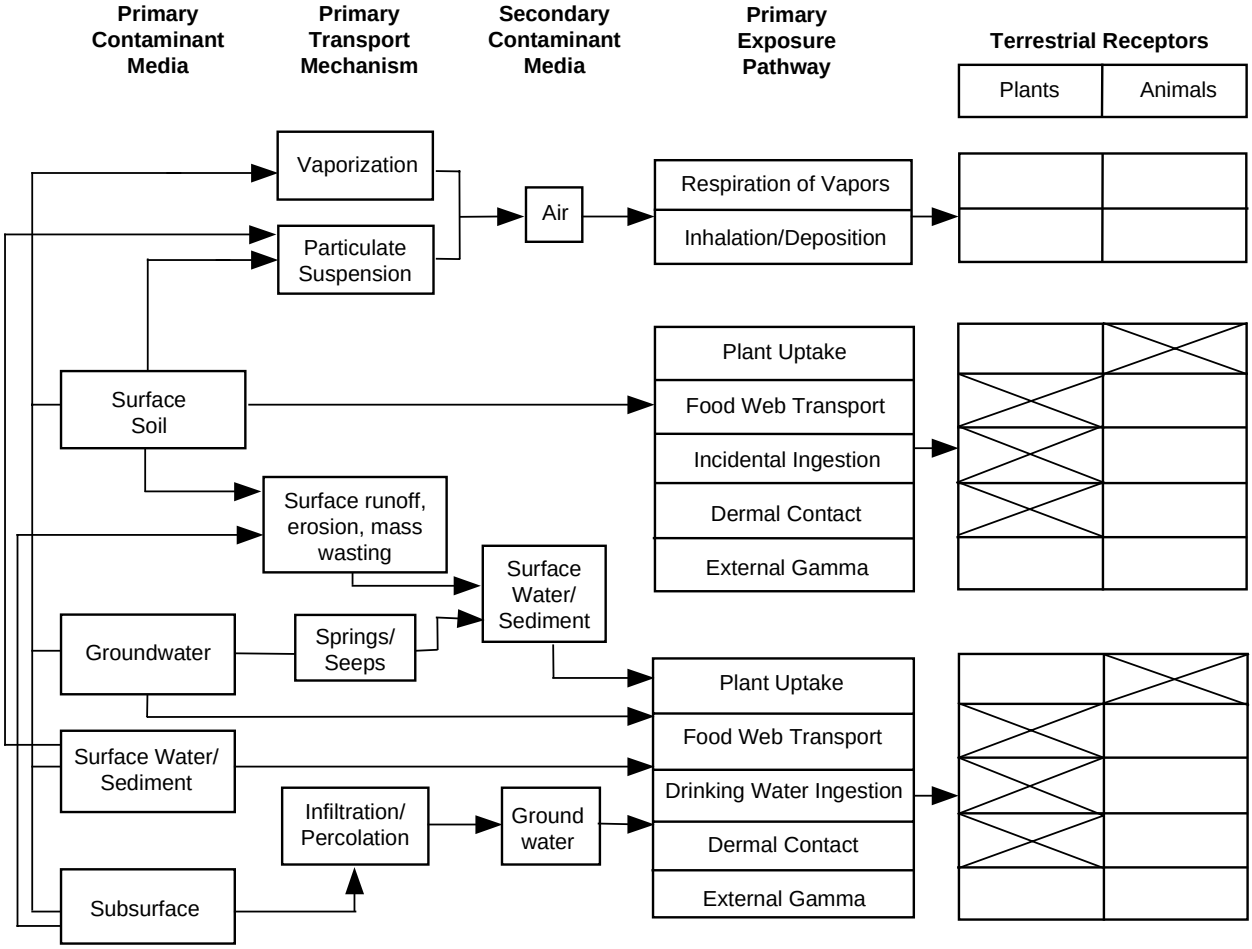
If there are receptors and pathways, then subsequent questions in Part B involving data adequacy will be addressed. Specifically, do existing data provide adequate information on the nature, rate, and extent of contamination? Also, do existing data for the site address potential pathways of site contamination and receptor exposure? Based on the ecological risk assessors’ evaluation of existing data, additional data may be required to resolve adequacy and/or quality issues. For example, if the COPCs at a site are based on elevated detection limits, the risk assessor should encourage re-sampling or re-analysis to obtain detection limits that are appropriate and usable in the ecological screening evaluation. Similarly, if vertical and/or horizontal extent of the contamination has not been adequately defined to permit an ecological assessment, a recommendation for additional sampling should be provided. Once the data issues and gaps have been resolved, the process of scoping and screening the site for potential ecological impacts should proceed.

Completion of Part B also includes additional field notes on the site setting and potential ecological receptors. The purpose of the field notes is to document other site observations considered to be relevant to the ecological screening evaluation of the site. Such information may include observations on the variability in the type and density of ecological receptors present at the site. Of particular interest are any field notes that could be used to document factors considered in the uncertainty analysis, Section 4.6.

4.2.1.3 Checklist Step 3: Ecological Site Conceptual Model

Part C of the checklist relates to the site conceptual model for ecological receptors. The ecological risk assessor should complete Part C within one or two days after the site visit. Once completed, Parts A, B, and C should be reviewed for technical accuracy by a qualified peer reviewer selected from the ecological risk task team. Part C consists of up to 22 questions related to contaminant transport and the potential for biological exposure (see Appendix A). Answers to Part C questions are used to complete the SCM. This model is used to select appropriate ecological screening receptors (terrestrial, aquatic, or both) and helps to interpret the results of the ecological screening assessment in a site-specific manner.

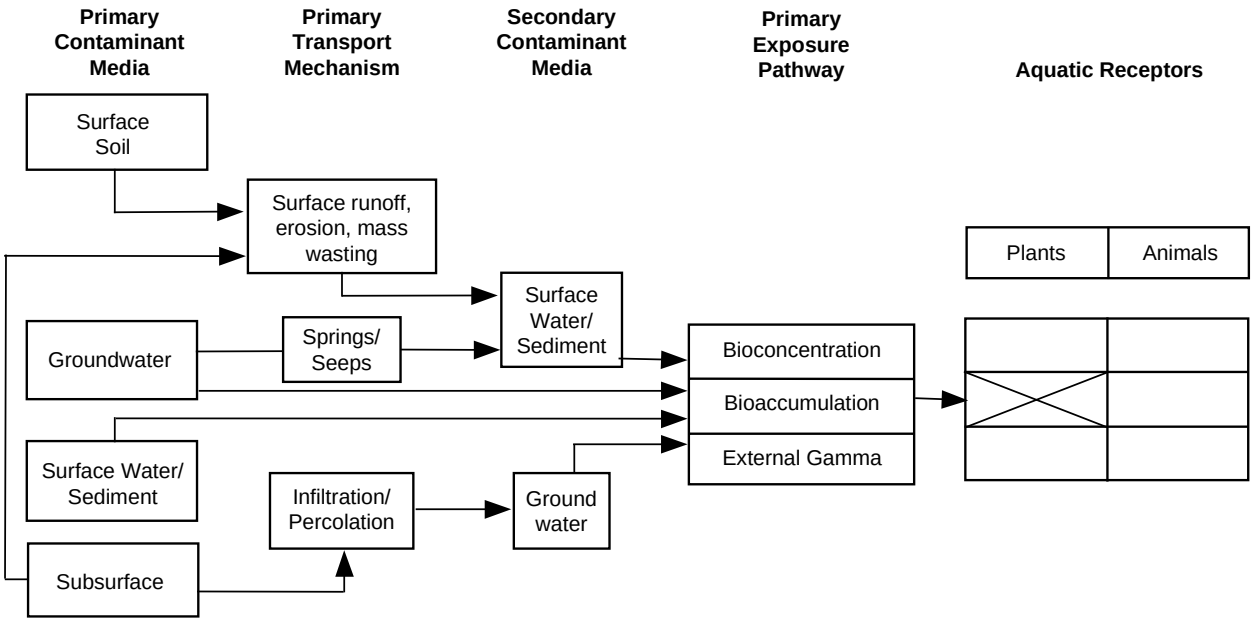
The generic terrestrial receptor conceptual model is depicted in Figure 4.2.1.3-1. The questions provided in the scoping checklist help evaluate the transport and exposure routes to terrestrial receptors. The model evaluates surface soil, groundwater, surface water/sediment, and the subsurface as potentially contaminated media. Figure 4.2.1.3-1 also illustrates the transport pathways that may lead to contaminated air, surface water/sediment, or groundwater as secondary contaminated media. There are two exposure routes to terrestrial receptors from air–respiration of vapors or inhalation/deposition of particulates. Respiration includes exposure to plants and invertebrates, and inhalation refers to exposure to wildlife. There are five possible exposure routes to terrestrial receptors from contaminated soil–plant uptake, food web transport, incidental ingestion, dermal contact, and external gamma. There are five possible exposure routes to terrestrial receptors from contaminated water/sediment–plant uptake, food web transport, drinking water ingestion, dermal contact, and external gamma. Groundwater may be an exposure medium for deep rooted plants, but is typically does not have complete exposure pathways to animals.



Boxes with Xs indicate incomplete pathways. Open boxes indicate complete pathways.

Figure 4.2.1.3-1. Terrestrial receptor conceptual exposure and transport model.

The generic aquatic receptor conceptual model is depicted in Figure 4.2.1.3-2. The questions provided in the scoping checklist help evaluate the transport and exposure routes to aquatic receptors. This model shows surface soil, groundwater, surface water/sediment, and the subsurface as possible primary contaminated media. Figure 4.2.1.3-2 also shows transport pathways that may lead to surface water/sediment or groundwater as secondary contaminated media. The aquatic model does not consider transport to air, as volatile contaminants are rapidly lost from surface water and sediment and potential for dust generation in damp sediments is unlikely. Thus, the aquatic model is most relevant to sites with perennial water. Sites with intermittent sources of water sources of water may need to evaluated as both terrestrial and aquatic conceptual models to make sure that all contaminant exposure pathway are evaluated. There are three possible exposure routes to aquatic receptors from contaminated surface water/sediment, including: bioconcentration, bioaccumulation, and external gamma. Bioconcentration covers all non-trophic exposure routes, which include respiration and dermal absorption. Bioaccumulation covers only trophic exposure routes, i.e., food web transport.



Boxes with Xs indicate incomplete pathways. Open boxes indicate complete pathways.

Figure 4.2.1.3-2. Aquatic receptor conceptual exposure and transport model.

4.3 Screening Evaluation Overview

The purpose of the screening evaluation is to identify contaminants of potential ecological concern (COPECs) by exposure media. The outcome of the screening evaluation is the determination of whether contaminants pose a potential unacceptable risk to ecological receptors. The screening evaluation is intended to be protective of the environment, not predictive of ecological risk. Thus, conservative assumptions are made throughout the screening evaluation to ensure that contaminants, exposure pathways, and sensitive species are not missed.

Screening is conducted in two steps, the first step compares site concentration data to final ecological screening levels (ESLs). The final ESLs are specific to media, and they include values for soil, sediment and water. For each medium and each COPC, there is a final ESL value. The basis for the final ESL is

discussed in Sections 4.4 and 4.5. Simply stated, the final ESL is the minimum applicable ESL value for a COPC in soil, sediment and water. The final ESL value is intended to be protective for all ecological receptors from exposure to that single media. The final ESL value incorporates a factor to account for potential additive effects of multiple contaminants. Thus, no hazard quotient/hazard index (HQ/HI) analysis to address multiple contaminants is needed for sites with only a single contaminated media. If the maximum[4] concentration of a COPC exceeds the final ESL, that COPC is considered to be a COPEC. An example result of the ESL comparison for a site with contaminated soil is provided in Table 4.3-1.

Table 4.3-1.

Example comparison of site media concentrations to ESLs.

COPC	Site Maximum Concentration (mg/kg)	Final Soil ESL (mg/kg)	COPEC in Soil
X	100	1000	No
Y	10	0.1	Yes
Z	50	10	Yes

In cases where COPCs are detected in multiple media, a second step is required where conservative estimates of exposure to wildlife species from multiple exposure media are calculated. All COPCs with potential additive effects are addressed through the calculation of the HQ/HI (hazard quotients/hazard index) for individual wildlife species receptors. If the HI for a wildlife receptor species exceeds 1.0, then those COPCs that contribute more than 0.3 to the HI for that receptor are identified as COPECs. Tables 4.3-2 and 4.3-3 provide an example summary of HI values and HQ calculations for COPCs detected in sediment and water. In this example, there are three receptors (“A”, “B”, and “C”). The HI for receptor “B” exceeds 1.0, and all other HI values are less than 1.0 (see Table 4.3-2). Thus, to determine which COPECs are identified, the COPCs that contribute to HI for receptor “B” are evaluated. This evaluation shows that COPC “Z” is identified as a COPEC in sediment (Table 4.3-3).

Table 4.3-2.

Example HI summary by wildlife receptor.

Receptor	HI
A	0.6
B	100
C	0.77

[4] However, the NMED has instructed the Laboratory that, “if the existing data set allows for a meaningful 95% UCL of the arithmetic mean media concentration to be calculated, LANL may consider substituting the 95% UCL value for the maximum media concentration” (Dinwiddie 1998, ER ID 62741).

Table 4.3-3.
Example HQ calculation for wildlife receptor "B".

COPC	Medium	Estimated Exposure to Wildlife Receptor x (mg/kg/day)	TRV (mg/kg/day)	HQ for "no effect"[a]
X	Sediment	1	100	0.01
Y		10	100	0.1
Z[b]		100	1	100
X	Water	0.1	1	0.1
Y		3	3000	0.001
Z[b]		0.01	1	0.01
				HI =100[a]

a. HI sums are rounded to three figures.
b. HQ value for this COPC identifies it as a COPEC.

Derivation of Final ESLs

Sections 4.4 and 4.5 describe the methods used to derive final ESLs for non-radiological and radiological COPCs for soil, sediment, and water. These methods are based on wildlife exposure models and, for non-radionuclides in aquatic environments, on water and sediment benchmark values from a number of data sources. Calculation of final ESLs require information derived from the primary toxicological literature, toxicologically-based numerical standards, exposure parameters for wildlife species, and NMED-approved compilations of ecological risk-based screening values. Although methods for ESL derivation are presented here, the ESLs and the supporting information are not included. They are provided in the ECORISK Database (LANL 1998, ER ID Package 186).

ESLs for radionuclides are derived from models that calculate the internal and external dose. While the radionuclide models resemble the wildlife ESL models for non-radionuclides, radionuclide ESL models are presented separately from non-radionuclides for greater clarity of presentation.

HQ/HI Calculations

Fundamental to the screening-level risk assessment is the concept of the HQ. The HQ is a ratio between exposure and an effect level, which can be used as a potential indicator of effects. The HI is a sum of HQ values for COPCs with common toxicological effects. The following equations show how the HQ and HI are calculated, and are based on EPA (1997, ER ID 59370):

$$HQ_{ij} = \frac{exposure_{ij}}{effect_{ij}} \qquad HI_i = \sum_{j=1}^{n} HQ_{ij}$$

Where: HQ_{ij} is hazard quotient for receptor i to COPC j (unitless)
$exposure_{ij}$ is exposure to COPC j for receptor i (units are mg of COPC per kg body weight per day or mg/kg/day)
$effect_{ij}$ is effect level for exposure to COPC j for receptor i (mg/kg/day)
HI_i is hazard index for receptor i to n COPCs (unitless)

Ecological Effects of Concern for Screening

In this methodology, effects of ecological concern are considered adverse at the organismal level, such as reproduction, development, growth, and survival. Table 4.3-4 shows receptors and ecological effects considered primarily relevant for screening-level ecological risk assessments. Note that this table is not intended to document all relevant effects, it only shows those effects primarily considered in evaluating the toxicological literature. Other effects will be evaluated and used on a chemical by chemical basis, and the rationale for selecting the relevant effect for each chemical will be documented in the ECORISK Database (LANL 1998, ER ID Package 186).

Table 4.3-4.

Potentially relevant ecological effects for ecological receptors.

Receptor	Effect Category		
	Reproduction/Development	Survival	Growth
Generic plant	Yield, seedling survival	Not relevant	Biomass
Earthworm	Cocoons produced	Count	Not relevant[a]
Robin	Eggs produced, hatching success, fledging survival	Adult survival	Not relevant[a]
Kestrel	Eggs produced, hatching success, fledging survival	Adult survival	Not relevant[a]
Deer mouse	Young produced, juvenile survival	Adult survival	Not relevant[a]
Desert cottontail	Young produced, juvenile survival	Adult survival	Not relevant[a]
Shrew	Young produced, juvenile survival	Adult survival	Not relevant[a]
Red fox	Young produced, juvenile survival	Adult survival	Not relevant[a]
Bat	Young produced, juvenile survival	Adult survival	Not relevant[a]
Swallow	Eggs produced, hatching success, fledging survival	Adult survival	Not relevant[a]

a. Unless growth impacts reproductive success

Effects on reproduction include measurable impacts to sexually mature adults due to exposure to a chemical. Measures may include effects on reproductive systems or the outcome of such effects, such as measures of fecundity.

Developmental effects for vertebrates include those that adversely impact organisms, in any developmental life stage, such that behavior, survival, and/or reproductive status are compromised. Effects may be morphologically and/or physiologically mediated. Effects on juveniles are associated with exposure to a chemical during pre- or post-fertilization and/or during pre- and post-embryonic development. Effects on adults are associated with exposure to a chemical during life stages when reproductive status or potential may vary (e.g., organism that reproduces over multiple years). Developmental effects for invertebrates and plants are similar to those for vertebrates. Relative to vertebrates, life stages may differ for invertebrates and plants, although the effects of chemical exposure are also morphologically and/or physiologically mediated and may directly or indirectly compromise behavior, survival, and/or reproductive status.

Growth effects include impairment of an organism's body weight, length, diameter, or other related measures resulting from chemical exposure. Survival effects include mortality and morbidity (e.g., immobility) due to chemical exposure. Growth and survival effects may be measured at any time during the life span of an organism. If exposure is multigenerational, then effects on growth and survival of the first generation and any other successive generations are considered to be developmental effects until the organism reaches maturity.

Dose-Response Model

The inherently conservative nature of the screening assessment involves a dose-response model assumed for most COPCs. For non-radionuclides the dose-response relationship is assumed to have a threshold effect, which means that low doses of a COPC have no effect on the organism. Typically, extremely high doses lead to a saturation of effects (e.g., 100% mortality). The threshold effects means that there is a maximum dose or environmental concentration that has no effect. This is the dose or concentration of interest for screening level assessments for non-radionuclides. For radionuclides, although there is not assumed to be a no-effect or threshold dose, it is assumed that there is a dose below which the risk is acceptable. This is the dose of interest for screening level assessments for radionuclides.

Most ecological screening assessments for non-radiological chemicals use the no observed adverse effect level (NOAEL) or no observed effect concentration (NOEC) as the maximum exposure value considered to be acceptable. This value is also called the toxicity reference value (TRV). The dose limits for are based radionuclides is 0.1 rads per day as discussed in Section 4.5. The Environmental Protection Agency (EPA) defines the NOAEL or NOEC 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" (EPA 1997, ER ID 59370).

To determine if wildlife receptors receive COPC doses that exceed the NOAEL (or acceptable radiological dose limits), a wildlife exposure model is developed and applied. This wildlife exposure model considers various dietary and non-dietary exposure pathways for wildlife. Modeling is not needed to evaluate exposure to non-wildlife species (e.g., plants, soil invertebrates, and aquatic organisms) because it is assumed that most of the COPC exposure to these organisms is not related to dietary pathways. Instead, it is assumed that effects on plants, soil invertebrates, and aquatic organisms are based on direct contact to and uptake from a contaminated medium. For example, root uptake for plants is considered to be the primary exposure route. If site-specific scoping suggests that foliar uptake may be a primary exposure route for a contaminant, lack of foliar uptake in the plant toxicity testing should be addressed in the uncertainty analysis.

General Wildlife Exposure Model

Wildlife exposure is derived by intake of COPCs from various sources, including the diet, incidental ingestion of contaminated media, dermal contact, and respiration. This general model is presented as Equation 4.3-1, and is based on general wildlife exposure models presented in EPA (1993, ER ID 59384).

$$E_{total} = E_{oral} + E_{dermal} + E_{respiration} \qquad \textbf{Equation 4.3-1}$$

Where: E_{total} is total exposure to a COPC (units are mg/kg/day)
E_{oral} is oral exposure (diet and direct ingestion of contaminated media, with units of mg/kg/day)
E_{dermal} is dermal exposure (with units of mg/kg/day)
$E_{respiration}$ is exposure through respiration or inhalation (with units of mg/kg/day)

For terrestrial wildlife, it has been assumed that most contaminant exposure to non-radiological chemicals is through the oral exposure pathway (Sample et al. 1997, ER ID 62807). Thus, the terrestrial wildlife exposure model for non-radionuclides simplifies to Equation 4.3-2.

$$E_{total} = E_{oral} \qquad \textbf{Equation 4.3-2}$$

Although the oral pathway is dominant in most cases, the site-specific scoping should assess the potential importance of the dermal and respiration/inhalation pathways. In cases where dermal and respiration may represent significant exposure pathways, the models presented by Hope (1995, ER ID 62783) should be used to evaluate these pathways. The oral exposure model used for terrestrial wildlife is adopted from the *Wildlife Exposure Factors Handbook* (Chapter 4 in EPA 1993, ER ID 59384), and is provided in Equation 4.3-3:

Equation 4.3-3

$$E_{oral} = C_{soil} \cdot I_{soil} \cdot AUF_{soil} + C_{water} \cdot I_{water} \cdot AUF_{water} \cdot d_{water} + C_{food} \cdot I_{food} \cdot AUF_{food}$$

Where: E_{oral} is the estimated oral daily dose for a COPC (mg/kg/day),
C_{soil} is the concentration of chemical constituent *x* in soil (mg/kg dry weight)
I_{soil} is the normalized daily soil ingestion rate (kg of soil / [kg of body weight • day], simplified to kg/kg/day in subsequent equations)
AUF_{soil} is the area use factor that represents the fraction of soil ingested from a contaminated area
C_{water} is the concentration of chemical constituent *x* in water (mg/L)
I_{water} is the normalized daily water ingestion rate (kg of water / [kg of body weight • day], simplified to kg/kg/day in subsequent equations)
AUF_{water} is the fraction of water ingested from a contaminated area
d_{water} is the density of water (1 kg/L)
C_{food} is the concentration of COPC in food (mg/kg)
I_{food} is the normalized daily dietary ingestion rate (kg of food [dry weight]/[kg of body weight • day], simplified to kg/kg/day in subsequent equations)
AUF_{food} is the fraction of the diet derived from a contaminated area

This model provides an estimate of the oral exposure associated with a concentration of an inorganic or organic chemical toxicant in soil, food and water, given an organism's normalized daily ingestion rate. This model considers incidental ingestion of soil and ingestion of contaminated water. Soil ingestion is calculated from a fraction of the dietary intake that is soil (see Chapter 4 in EPA 1993, ER ID 59384). As a conservative assumption for ecological risk screening, the area use factor (AUF) values are set to 1, suggesting that the animal receives all of its exposure from the contaminated site. An additional conservatism occurs when the maximum value is used to represent concentrations in contaminated media and food. The implications of these assumptions should be addressed in the uncertainty analysis.

An implicit assumption of this model is that the bioavailability of the COPC from the environmental media is comparable to the bioavailability of the contaminant in the toxicological experiment. Because there is currently little information on bioavailability conversions, a bioavailability term was not included in the general wildlife exposure model. If bioavailability of a COPC is known and site-specific adjustments to bioavailability are possible, this should be addressed in the site-specific uncertainty analysis, as discussed in Section 4.6.

The above model requires that all measures of ingestion (except water) are on a dry weight basis. Because EPA (1993, ER ID 59384) presents most normalized food ingestion rates on a wet weight basis, these dietary constituents must undergo wet-to-dry weight conversions. Food intakes rates are all provided in units of dry weight, and any conversion factors used in this calculation are provided. Metrics required for calculations of the general wildlife exposure model, conversions and other elements of the model are provided for terrestrial vertebrate receptors in Table 4.3-5. Note that the information provided in

Table 4.3-5 is for the screening receptors adopted by the Laboratory. It is important to note that exposure parameters provided in Table 4.3-5 represent conservative upper estimates of potential exposure. For example, the value provided for water intake represent the total daily water intake requirement and much of that water is obtained by these receptors in their diet and not from surface water sources. More realistic exposure information may be considered in the uncertainty analysis (see Section 4.6). Information about body weight and inhalation rates, which are not required by Equation 4.3-3, are provided to assist with alternate forms of the wildlife exposure model. For example, the exposure models discussed by Hope (1995, ER ID 62783) require these additional parameters.

Table 4.3-5.

Measures required for the wildlife exposure model.

Species	Parameter	Value	Units	Reference (page)	Notes
American kestrel	Body weight	0.103	kg	EPA (1993, ER ID 59384) p. 2-112	Smallest male average weight was 103 g
	Food intake[a]	0.099	kg/kg/day	EPA (1993, ER ID 59384) p. 2-112	Used higher of 2 empirical fresh weight food intake values, 0.31 kg/kg/day, multiplied by (100-68)% to account for food moisture content
	Food moisture content	0.68	Proportional	EPA (1993, ER ID 59384) p. 4-13	Diet includes insects, birds, mammals, other (see p. 2-113) [value assumes mammals, birds]
	Water intake	0.12	L/kg/day	EPA (1993, ER ID 59384) p. 2-112	Higher of 2 estimated values
	Inhalation rate	0.089	m^3/day	EPA (1993, ER ID 59384) p. 2-113	Higher of 2 estimated values
	Fraction soil in diet	0.02	Unitless	none	Default value
	Soil invertebrate diet[b]	0.5 (0)	Unitless	EPA (1993, ER ID 59384) p. 2-113	Rounded EPA value to 50% to equally expose receptor to potentially contaminated invertebrates and flesh
	Flesh diet[b]	0.5 (1)	Unitless	EPA (1993, ER ID 59384) p. 2-113	Rounded EPA value to 50% to equally expose receptor to potentially contaminated invertebrates and flesh
American robin	Body weight	0.077	kg	EPA (1993, ER ID 59384) p. 2-197	Smallest weight was 77 g
	Food intake[a]	0.35	kg/kg/day	EPA (1993, ER ID 59384) p. 2-197	Higher of 2 empirical values fresh weight food intake rate for robins feeding primarily on fruits, 1.52 kg/kg/day, multiplied by (100-77)% to account for food moisture content

a. Normalized ingestion rates are presented in units of kg of food (dry weight)/[kg of body weight • day].

b. There are two variants on the American kestrel, one more realistically models its actual diet (half invertebrate and half flesh), and the strict flesh-eater is used to mimic the diet of the Mexican spotted owl or peregrine falcon.

Table 4.3-5. (continued)

Measures required for the wildlife exposure model.

Species	Parameter	Value	Units	Reference (page)	Notes
American robin (continued)	Food moisture content	0.77	Proportional	EPA (1993, ER ID 59384) p. 4-13,14	Diet includes: invert, plants (fruits), assumed fruit
	Water intake	0.14	L/kg/day	EPA (1993, ER ID 59384) p. 2-197	Estimated from allometric equations
	Inhalation rate	N.A.[a]	m^3/day	N.A.	N.A.
	Fraction soil in diet	0.1	Unitless	Beyer et al. (1994, ER ID 62785) Table 1	Used Woodcock value
	Plant diet[b]	0, 0.5, or 1	Unitless	None	Modeled with three diets, herbivore, omnivore, insectivore
	Soil invertebrate diet[b]	1, 0.5, or 0	Unitless	None	Modeled with three diets, herbivore, omnivore, insectivore
Deer mouse	Body weight	0.020	kg	EPA (1993, ER ID 59384) p. 2-295	For females
	Food intake[c]	0.20	kg/kg/day	EPA (1993, ER ID 59384) p. 2-296	Based on empirical fresh weight food intake of 0.22 kg/kg/day (diet of lab chow, 8–10% moisture), multiplied by (100-10)% to account for food moisture
	Food moisture content	0.1	Proportional		See note on line above
	Water intake	0.19	L/kg/day	EPA (1993, ER ID 59384) p. 2-296	Adult male or female
	Inhalation rate	0.025	m^3/day	EPA (1993, ER ID 59384) p. 2-296	Higher of 2 values, estimated
	Fraction soil in diet	0.02	Unitless	Beyer et al. (1994, ER ID 62785) Table 1	For white-footed mouse
	Plant diet	0.5	Unitless	EPA (1993, ER ID 59384) p. 2-297	Rounded EPA value to 50% to equally expose receptor to potentially contaminated plants and invertebrates
	Soil invertebrate diet	0.5	Unitless	EPA (1993, ER ID 59384) p. 2-297	Rounded EPA value to 50% to equally expose receptor to potentially contaminated plants and invertebrates
Eastern cottontail as a surrogate for desert cottontail	Body weight	0.800	kg	EPA (1993, ER ID 59384) p. 2-355	Lower 95th percentile of mean weight of males. Chosen based on reported body weight of smaller desert cottontail
	Food intake[c]	0.093	kg/kg/day	Nagy (1987, ER ID 62782)	Estimated as 95% upper CI using Nagy (1987) allometric scaling formula for herbivores

a. N.A. = not available

b. There are three variants on the American robin, one modeled as a strict herbivore, an omnivore eating 50% plants and 50% invertebrates, and lastly as a strict insectivore.

c. Normalized ingestion rates are presented in units of kg of food (dry weight)/[kg of body weight • day].

Table 4.3-5. (continued)

Measures required for the wildlife exposure model.

Species	Parameter	Value	Units	Reference (page)	Notes
Eastern cottontail as a surrogate for desert cottontail (continued)	Water intake	0.097	L/kg/day	EPA (1993, ER ID 59384) p. 2-356	Estimated from allometric equations
	Inhalation rate	0.63	m^3/day	EPA (1993, ER ID 59384) p. 2-356	Estimated from allometric equations
	Fraction soil in diet	0.024	Unitless	Beyer et al. (1994, ER ID 62785) Table 1	For meadow vole
	Plant diet	1	Unitless	EPA (1993, ER ID 59384) p. 2-356	Assume strict herbivore diet
Short-tailed shrew as a surrogate for vagrant shrew	Body weight	0.015	kg	EPA (1993, ER ID 59384) p. 2-213	Smallest weight was 15 g
	Food intake[a]	0.198	kg/kg/day	EPA (1993, ER ID 59384) p. 2-213	Higher of 2 empirical fresh weight food intakes, 0.62 mg/kg/day, multiplied by (100-68)% to account for food moisture in diet of beef liver
	Food moisture content	0.68	Proportional	EPA (1993, ER ID 59384) p. 4-13	Laboratory feeding study used beef liver
	Water intake	0.223	L/kg/day	EPA (1993, ER ID 59384) p. 2-213	One value reported
	Inhalation rate	0.026	m^3/day	EPA (1993, ER ID 59384) p. 2-213	One value reported
	Fraction soil in diet	0.1	Unitless	Beyer et al. (1994, ER ID 62785) Table 1	Used woodcock
	Soil invertebrate diet	1	Unitless	EPA (1993, ER ID 59384) p. 2-214	Assume strict insectivore diet
Gray fox as a surrogate for red fox	Body weight	3 940	g	EPA (1993, ER ID 59384) p. 2-224	Lowest of 4 values
	Food intake[a]	0.045	kg/kg/day	EPA (1993, ER ID 59384) p. 2-224	Female after whelping, empirical fresh weight food intake is 0.14 kg/kg/day for an unknown diet, multiplied by assumed food moisture content (100-68)%
	Food moisture content	0.68	Proportional	EPA (1993, ER ID 59384) p. 4-13	Mostly mammals, some birds [assumed mammals]
	Water intake	0.086	L/kg/day	EPA (1993, ER ID 59384) p. 2-224	Higher of 2 values, estimated
	Inhalation rate	2	m^3/day	EPA (1993, ER ID 59384) p. 2-224	Higher of 2 values, estimated
	Fraction soil in diet	0.03	Unitless	Beyer et al. (1994, ER ID 62785) Table 1	For red fox
	Flesh diet	1	Unitless	EPA (1993, ER ID 59384) p. 2-224	Rounded diet to 100% flesh

a. Normalized ingestion rates are presented in units of kg of food (dry weight)/[kg of body weight • day].

Table 4.3-5. (continued)

Measures required for the wildlife exposure model.

Species	Parameter	Value	Units	Reference (page)	Notes
Violet-green swallow	Body weight	0.0139	kg	Dunning (1993, ER ID 62886)	Average body weight of females for *Tachycineta thalassina*
	Food intake[a]	0.268	kg/kg/day	Nagy (1987, ER ID 62782)	Estimated as 95% upper CI using Nagy (1987) allometric scaling formula for passerines
	Water intake	0.242	L/kg/day	EPA (1993, ER ID 59384) p. 3-10	Based on allometric scaling formula for birds
	Inhalation rate	N.A. [b]	m^3/day	None	None
	Invertebrate diet	1	Unitless	None	Assume 100% invertebrate diet
Occult little brown myotis bat	Body weight	0.0088	kg	Whitaker (1980, ER ID 62889)	Used mid-point of reported body weight range for *Myotis lucifugus* (3.1 to 14.4 g)
	Food intake[a]	0.159	kg/kg/day	Nagy (1987, ER ID 62782)	Estimated as 95% upper CI using Nagy(1987) allometric scaling formula for all mammals
	Food moisture content	0.69	Proportional	EPA (1993, ER ID 59384) p. 4-13	Used value for grasshoppers and crickets as surrogate for emergent aquatic insects
	Water intake	0.159	kg/kg/day	EPA (1993, ER ID 59384) p. 3-8	Based on allometric scaling formula for mammals
	Inhalation rate	0.012	m^3/day	EPA (1993, ER ID 59384) p. 3-12	Based on allometric scaling formula for mammals
	Fraction soil in diet	0	Unitless	None	Assume no soil exposure for aerial insectivores
	Invertebrate diet	1	Unitless	None	Assume 100% invertebrate diet

a. Normalized ingestion rates are presented in units of kg of food (dry weight)/[kg of body weight • day].

4.4 ESLs for Nonradiological COPCs

This section will provide an overview of the approach used to develop ESLs for non-radionuclides for soil, sediment and water. Table 4.4-1 summarizes the receptors and diet compositions used for each exposure medium.

Table 4.4-1.
ESL media and screening receptors.

ESL Medium	Receptor Group	Receptor Name	Diet Composition
Soil	Bird	American kestrel	50% invertebrate/ 50% flesh
		American kestrel	100% flesh
		American robin	100% invertebrate
		American robin	50% invertebrate/ 50% plant
		American robin	100% plant
	Mammal	Desert cottontail	100% plant
		Deer mouse	50% invertebrate/ 50% plant
		Red fox	100% flesh
		Vagrant shrew	100% invertebrate
	Plant	Generic Plant	Not applicable
	Invertebrate	Earthworm	Not applicable
Water [a]	Bird	American kestrel	No food, water only[b]
		American robin	No food, water only[b]
		Swallow	No food, water only[b]
	Mammal	Desert cottontail	No food, water only[b]
		Deer mouse	No food, water only[b]
		Red fox	No food, water only[b]
		Vagrant shrew	No food, water only[b]
		Bat	No food, water only[b]
	Aquatic	Multiple aquatic receptors that represent most aquatic organisms	Not applicable
Sediment [a]	Bird	Swallow	100% invertebrate
	Mammal	Bat	100% invertebrate
	Aquatic	Multiple aquatic receptors that represent most aquatic organisms	Not applicable

a. Water and sediment ESLs are only used to help evaluate significant exposure pathways and COPCs for those media. In all cases where a site has one of these media contaminated, a multimedia assessment is expected.

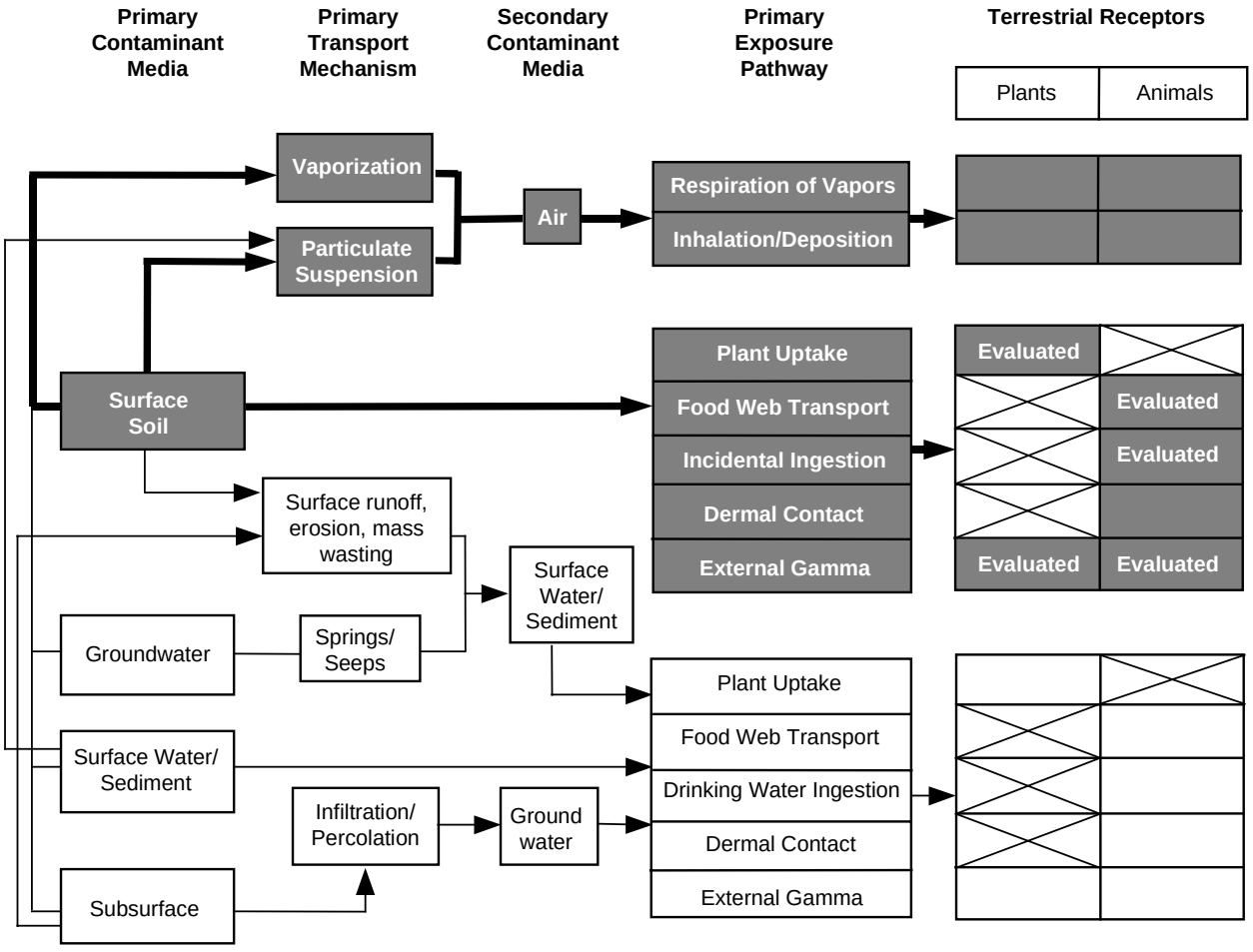
b. The water ESL for these terrestrial receptors only reflects the exposure from contaminated water from the site. Therefore, a multimedia exposure assessment may be required to address the potential cumulative effects from soil (or sediment) and water for these receptors.

4.4.1 Soil ESLs

As described in the Laboratory background document for soil, sediment, and Bandelier Tuff, “soil” is defined as material overlaying intact bedrock that has been modified by the addition of organic material or by movement of clay sizes particles and by development of ferric hydroxides (Ryti et al. 1998, ER ID 59730). For the purposes of ecological risk screening, imported fill or disturbed soils are evaluated similarly to well-developed soils, because they all have common exposure and transport pathways.

Although soil ESLs are based on exposure to terrestrial receptors—plants, invertebrates (earthworms), and wildlife—they are determined differently for each receptor. The different approaches are required because of the different ways that toxicological experiments are performed for these organisms. For plants, earthworms and other soil-dwelling invertebrates, effects are based on the concentration of a

COPC in soil. Therefore, ESL values are directly based on effects concentrations and modeling is not required. Exposure to wildlife, however, is dependent on exposure of the organism to a chemical constituent from a given medium (such as soil or foodstuff) through direct and indirect means (i.e., ingestion, inhalation, and dermal) and serves as the model for terrestrial dose exposure calculation (EPA 1993, ER ID 59384). The transport and exposure pathways that are likely to be complete for sites with soil contamination are depicted in Figure 4.4.1-1. For wildlife receptors, ESL values will be based on the dietary regimen of the receptor, including consumption of plants, invertebrates, and vertebrate flesh, with some incidental soil ingestion.



Complete pathways for soil exposure are gray; evaluated pathways are included in the soil ESL calculations

Figure 4.4.1-1. Ecological conceptual model for soil pathways.

Contaminant transport from surface soil or transport from subsurface media (soil or bedrock) are not evaluated under the soil conceptual model for ESL derivation. However, ESLs combined with transport models can used to evaluate these pathways. Surface soil is generally assumed to represent the 0–5 ft interval, but the site-specific scoping should present a rationale and justification for the depth interval assumed to represent surface soils.

The final soil ESL for each COPC will be the lowest receptor-specific soil ESL value among plants, invertebrates, robin, kestrel, shrew, mouse, cottontail, and fox. The strategy for developing the final soil ESL is presented in Table 4.4.1-1.

Table 4.4.1-1.
Strategy for obtaining the final non-radionuclide soil ESL.

COPC	Plant ESL[a] (mg/kg)	Invertebrate ESL[b] (mg/kg)	Minimum Wildlife[c] ESL (mg/kg)	Final Soil ESL (mg/kg)
U	Value[d]	Value	Value	Minimum of plant ESL, invertebrate ESL and wildlife ESL
V	No value[e]	Value	Value	Minimum of invertebrate ESL and wildlife ESL, address uncertainty of no plant toxicity information
X	No value	No value	Value	Equals minimum wildlife ESL, address uncertainty of no plant or invertebrate toxicity information
Y	No value	No value	No value	No soil ESL available or calculable; retain COPC as COPEC
Z	Value	Value	No value	Equals minimum of plant ESL and invertebrate ESL, address uncertainty of no wildlife ESL

a. Plant ESL, which is a directly measured soil concentration value (no modeling needed)

b. Invertebrate ESL, which is a directly measured soil concentration value (no modeling needed)

c. Minimum ESL of six wildlife receptors (robin, kestrel, cottontail, mouse, shrew and fox), comments on the final soil ESL assume that there are ESL values for all wildlife receptors, uncertainty of missing ESLs for some receptors should be addressed (Note that wildlife ESLs are calculated from an exposure model)

d. Value = value available for that COPC

e. No value = no value available for that COPC

For plants and invertebrates the soil ESL will be the 0.3·NOEC. Per the instructions of NMED, the factor, 0.3, is used to account for potential additive effects of COPCs. Information supporting the NOECs is provided in the ECORISK Database (LANL 1998, ER ID Package 186). For wildlife, the soil ESL is the soil concentration of the COPC, where the exposure calculated in Equation 4.4.1-2 equals the 0.3·NOAEL. Per the instructions of NMED the factor, 0.3, is used to account for potential additive effects of COPCs.

The conversion of soil concentration to dose requires a simple inversion of the wildlife exposure model (with the intake of contaminated water assumed to be zero) discussed below. This inversion is possible because the food intake value can be related to concentration in soil. The general basis for this relationship is shown in Equation 4.4.1-1.

$$C_{food} = C_{soil} \cdot TF_{food} \qquad \textbf{Equation 4.4.1-1}$$

Where: C_{food} is the concentration of the COPC in food (units are mg/kg)
C_{soil} is the concentration in soil (mg/kg)
TF_{food} is a unitless transfer factor from soil to food

Thus, the general wildlife exposure model can be re-written in the following form (Equation 4.4.1-2), after setting the AUF parameters to 1 and using the relationship between C_{soil} and C_{food} shown in Equation 4.4.1-1.

$$E_{oral} = C_{soil} \cdot I_{soil} + C_{soil} \cdot TF_{food} \cdot I_{food}$$ **Equation 4.4.1-2**

Where: E_{oral} is the estimated oral daily dose for a COPC (mg/kg/day),
C_{soil} is the concentration of chemical constituent *x* in soil (mg/kg dry weight)
I_{soil} is the normalized daily soil ingestion rate (kg/kg/day)
I_{food} is the normalized daily dietary ingestion rate (kg/kg/day)
TF_{food} is a unitless transfer factor from soil to food

Because the intake of soil can be related to the intake of food, Equation 4.4.1-2 can be further simplified to Equation 4.4.1-3. This manner of modeling soil intake rate is conservative as it assumes incidental soil intake is in addition to food intake. An alternate model would be based on a total oral intake, and in this alternate model soil and food intake would add to 100% of the total intake.

$$E_{oral} = C_{soil} \cdot I_{food} \cdot [fs + TF_{food}]$$ **Equation 4.4.1-3**

Where: E_{oral} is the estimated oral daily dose for a COPC (mg/kg/day),
C_{soil} is the concentration of chemical constituent *x* in soil (mg/kg dry weight)
fs is the fraction of soil ingested, expressed as a fraction of the dietary intake
I_{food} is the normalized daily dietary ingestion rate (kg/kg/day)
TF_{food} is a unitless transfer factor from soil to food

Solving Equation 4.4.1-3 for the COPC and wildlife receptor-specific ESL, yields Equation 4.4.1-4.

$$ESL_{ij} = \frac{0.3 \cdot NOAEL_{ij}}{I_i \cdot [fs_i + TF_{ij}]}$$ **Equation 4.4.1-4**

Where: ESL_{ij} is the soil ESL for wildlife receptor i and COPC j (mg/kg),
$NOAEL_{ij}$ is the NOAEL for wildlife receptor i and COPC j (mg/kg/day),
fs_i is the fraction of soil ingested by wildlife receptor i, expressed as a fraction of the dietary intake
I_i is the normalized daily dietary ingestion rate for wildlife receptor i (kg/kg/day)
TF_{ij} is a unitless transfer factor from soil to food for wildlife receptor i and COPC j

Equation 4.4.1-4 assumes a single food type is ingested and must be specific for herbivores, omnivores, insectivores and carnivores. Equations for these functional groups of wildlife receptors are shown in Equations 4.4.1-5 through 4.4.1-8.

$$ESL_{ij} = \frac{0.3 \cdot NOAEL_{ij}}{I_i \cdot [fs_i + TF_{plant,j}]}$$ **Equation 4.4.1-5**

Where: ESL_{ij} is the soil ESL for herbivore i and COPC j (mg/kg),
$NOAEL_{ij}$ is the NOAEL for herbivore i and COPC j (mg/kg/day),
fs_i is the fraction of soil ingested by herbivore i, expressed as a fraction of the dietary intake
I_i is the normalized daily dietary ingestion rate for herbivore i (kg/kg/day])
$TF_{plant,j}$ is a unitless transfer factor from soil to plants for COPC j

$$ESL_{ij} = \frac{0.3 \cdot NOAEL_{ij}}{I_i \cdot [fs_i + fp_i \cdot TF_{plant,j} + fi_i \cdot TF_{invert,j}]}$$ **Equation 4.4.1-6**

Where: ESL_{ij} is the soil ESL for omnivore i and COPC j (mg/kg),
$NOAEL_{ij}$ is the NOAEL for omnivore i and COPC j (mg/kg/day),
fs_i is the fraction of soil ingested by omnivore i, expressed as a fraction of the dietary intake
I_i is the normalized daily dietary ingestion rate for omnivore i (kg/kg/day)
fp_i is the fraction of plants in diet for omnivore i
$TF_{plant,j}$ is a unitless transfer factor from soil to plants for COPC j
fi_i is the fraction of invertebrates in diet for omnivore i
$TF_{insect,j}$ is a unitless transfer factor from soil to insects for COPC j

$$ESL_{ij} = \frac{0.3 \cdot NOAEL_{ij}}{I_i \cdot [fs_i + TF_{invert,j}]}$$ **Equation 4.4.1-7**

Where: ESL_{ij} is the soil ESL for insectivore i and COPC j (mg/kg),
$NOAEL_{ij}$ is the NOAEL for insectivore i and COPC j (mg/kg/day),
fs_i is the fraction of soil ingested by insectivore i, expressed as a fraction of the dietary intake
I_i is the normalized daily dietary ingestion rate for insectivore i (kg/kg/day)
$TF_{insect,j}$ is a unitless transfer factor from soil to invertebrates for COPC j

$$ESL_{ij} = \frac{0.3 \cdot NOAEL_{ij}}{I_i \cdot [fs_i + TF_{flesh,j}]}$$ **Equation 4.4.1-8**

Where: ESL_{ij} is the soil ESL for carnivore i and COPC j (mg/kg),
$NOAEL_{ij}$ is the NOAEL for carnivore i and COPC j (mg/kg/day),
fs_i is the fraction of soil ingested by carnivore i, expressed as a fraction of the dietary intake
I_i is the normalized daily dietary ingestion rate for carnivore i (kg/kg/day)
$TF_{flesh,j}$ is a unitless transfer factor from soil to flesh for COPC j

Inspection of the wildlife ESL model (Equation 4.4.1-5 and the functional group-specific Equations 4.4.1-6 through 4.4.1-8) shows that the ESL is proportional to the NOAEL. Thus, larger values of the NOAEL lead to larger ESL values, which suggests that the receptor may be more tolerant of the COPC. The opposite relationship holds for the variables in the denominator of the wildlife ESL model. Thus, a receptor with higher feeding rates or one that eats more contaminated prey, has a lower ESL. Thus, animals with higher exposure will have lower ESLs, suggesting that they are less tolerant of the COPC. Table 4.4.1-2 summarizes the input variables for the wildlife exposure models, and indicates the general sources used for these variables.

Table 4.4.1-2.

Summary of variables used in the non-radionuclide wildlife ESL models.

Variable	Source
NOAEL	Receptor and COPC specific values are obtained from reviewing primary ecotoxicity literature. Values for specific receptors and COPCs are provided in the ECORISK Database. To provide bounding information on effects a lowest observed adverse effect level (LOAEL) will also be developed for wildlife receptors. Information on the LOAEL for specific receptors will be provided in the ECORISK Database.
fs	Receptor-specific values are provided in Table 4.3-5.
I	Body weight normalized food intake for wildlife receptors, see values provided in Table 4.3-5. Thus, body weight is an implicit component of this variable. For this reason Table 4.3-5 provides body weight for each receptor. Note that intake can also be expressed as a gross daily amount (in units of kg of food ingested per day). This alternate formulation of the model requires body weight to be an explicit variable.
fp	Fraction of plants in diet is provided in Table 4.3-5.
fi	Fraction of invertebrates in diet is provided in Table 4.3-5.
TF_{plant}	The transfer from soil to plants is a COPC-specific value that is derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database. The ECORISK Database must be reviewed to determine if the soil to plant transfer factor accounts for all complete plant exposure pathways. In particular, many plant uptake factors do not include foliar uptake. If foliar uptake represents a complete pathway for site, then the effect of not including this pathway in the plant uptake factor should be evaluated in the site-specific uncertainty analysis.
TF_{invert}	The transfer from soil to invertebrates is a COPC-specific value that is derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database.
TF_{flesh}	The transfer from soil to flesh is a COPC-specific value that is derived from two other factors. The first factor is a feed to muscle transfer factor (TF_{beef}) derived from studies of beef cattle. The second factor is the maximum of either the TF_{plant} or TF_{invert}, which is used to model the prey with the most contaminated diet. It is assumed that the prey being consumed are mammalian. Thus, TF_{flesh} = maximum(TF_{plant}, TF_{invert})*TF_{beef}. Values for specific COPCs are provided in the ECORISK Database.

Uncertainties associated with soil ESLs fall into two main categories. The first group of uncertainties is associated with COPCs, including toxicity and bioavailability (or transfer factors between soil and food). The second group of uncertainties relates to receptors, including feeding rates, the amount of incidental soil ingestion and diets. These uncertainties are addressed by selecting inputs to the soil ESL calculations that represent *worst-case* conditions. For example, carnivores could have mammalian and avian prey, but this would tend to reduce exposure because of the lower fat content of birds versus mammals[5]. Uncertainties are also addressed by using the lowest receptor-specific soil ESL as the final soil ESL value for each chemical. This ensures that the screening evaluation is protective and inclusive of all potential contaminants. Soil ESLs only screen individual COPCs, and Section 4.4.4, Multimedia Screening Calculations, describes how multiple exposure media are evaluated.

One important factor not considered in the development of wildlife ESLs is the potential for biomagnification of COPCs in higher trophic levels. The carnivore is modeled as eating herbivore or

[5] The typical way to adjust the TF_{beef} for bird flesh is to apply a multiplier to this parameter to account for the relative fat content of birds and mammals. For example, if the fat content of beef is 19% and chicken is 15%, then a 0.8 factor could be used to account for the relative transfer into birds versus mammals. Because the factor is likely to be less than one, it is conservative to assume that TF_{beef} applies to any vertebrate flesh.

insectivore prey, which have consumed potentially contaminated plants or insects. However, this model does not account for top carnivores that may be eating prey with more complex diets (e.g., a raptor that eats a snake that preys on lizards that eat predaceous insects that eat herbivorous insects). Developing models to account for multiple trophic level transfers is extremely complex, and is considered to be beyond the realm of screening. The potential for biomagnification for top carnivores depends on factors relating to the spatial distribution of the COPC and its biological retention time. This should be viewed as an uncertainty and discussed on a site-specific basis where potentially biomagnifying COPCs are identified.

Body weight is the main covariate for many of the parameters in the wildlife soil ESL models. Body weight has an allometric relationship to gross food intake rates (Nagy 1987, ER ID 62782), and is also used as a normalizing factor for food intake and the NOAEL values. Some studies also show relationships between body size and toxicity (for example, Newman et al. 1994, ER ID 62788). The energy value of the food consumed by the animal also shows a relationship to food intake (Nagy 1987, ER ID 62782). For example, an animal consuming a low energy food source must consume a greater quantity to support its basal metabolism. Thus, there are interrelationships between diet composition, body weight and food intake. There are also relationships between body weight and home range because small animals tend to have smaller home ranges (Cotgreave 1993, ER ID 62905). Thus, screening receptors were selected to be relatively small species within a feeding guild, which will tend to have smaller home ranges and greater food intake per unit body mass.

As noted above, one of the goals of the approach to calculating soil ESLs is to ensure that COPECs or pathways are not eliminated prematurely. Thus, more realistic modeling, including the application of non-linear TF relationships, is viewed as unnecessary for the purposes of screening.

4.4.2 Sediment ESLs

Geomorphologists define sediments as young alluvium occurring within or near stream channels, which would be generally classified as A or C genetic horizons in soil nomenclature (Ryti et al. 1998, ER ID 59730). This definition includes sediments in active channels, inactive channels, and floodplain geomorphic settings. Sediments can also be found in lentic systems (ponds or lakes), but there are no lakes and few ponds found on Laboratory property. Inactive channel and floodplain sediments typically have associated terrestrial ecological communities, and therefore are more akin to soils from an ecological risk evaluation perspective. Thus, soil ESLs apply to inactive channel and floodplain sediments. Aquatic ecological communities are often associated with perennial and seasonally intermittent aquatic environments, thus sediment-based ESLs are developed to apply to active channel and pond geomorphic settings with developed aquatic communities.

Because of the typical association of sediments with water, application of sediment ESLs leads to an incomplete evaluation of the potential ecological effects associated with contaminated sediment/water settings. Thus, a surface water and multimedia exposure assessment will be required in all cases where contaminated sediment is identified. The intent of developing sediment ESLs is to assist in determining the sensitive receptors and major and minor exposure pathways from contaminated sediments, which in turn assists in developing an appropriate multimedia exposure model.

Sediment ESLs for the protection of aquatic life are derived from the wealth of information on direct effects of contaminated sediments on aquatic organisms. Only limited modeling is needed to develop sediment ESLs. Modeling is used to evaluate potential effects of contaminated sediments on terrestrial

receptors through accumulation of COPCs in emergent insects. Thus sediment ESLs incorporate bioaccumulation issues and trophic transfer concerns to the level of insectivores.

General discussion of the transport and exposure pathways considered in the development of sediment ESLs is needed to evaluate the applicability of sediment screening values to the results of site-specific scoping. Pathways of sediment transport to aquatic environs include water as a primary contaminated media through discharge of effluents, directly or indirectly into perennial and intermittent water bodies; surface water runoff from contaminated soils; infiltration of surface water into shallow and/or deep groundwater; mass wasting; and wind-driven transport of soil-borne COPCs into water courses/bodies (Figures 4.4.2-1 and 4.4.2-2). Of primary concern are the first three of the aforementioned transport mechanistic pathways, which are included in Figures 4.4.2-1 and 4.4.2-2. In the rare instances where mass wasting or wind-blown soils may significantly influence the sediment load of a water body, this is identified during site-specific problem scoping. With the limited water resources in the region, primary focus should be on pathways of sediment transport from areas adjacent to, or contiguous with, permanent or seasonally intermittent surface water resources.

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, ER ID 62902) cites several reasons for the requirement of sediment quality criteria/benchmarks (SQC/SQBs), including:

1. Various toxic contaminants found only in trace amounts in the water column accumulate in sediments to elevated levels;

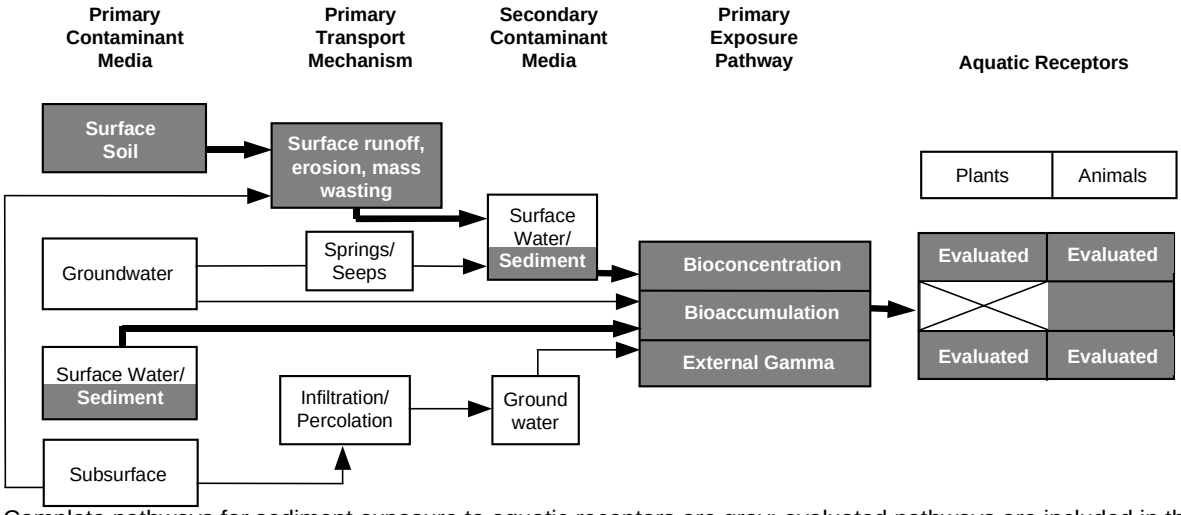
2. Sediments serve as both a reservoir and a source of contaminants to the water column;

3. Sediments integrate contaminant concentrations over time, whereas water column contaminant concentrations are much more variable and dynamic;

4. Sediment contaminants, in addition to water column contaminants, affect benthic and other sediment-associated organisms; and,

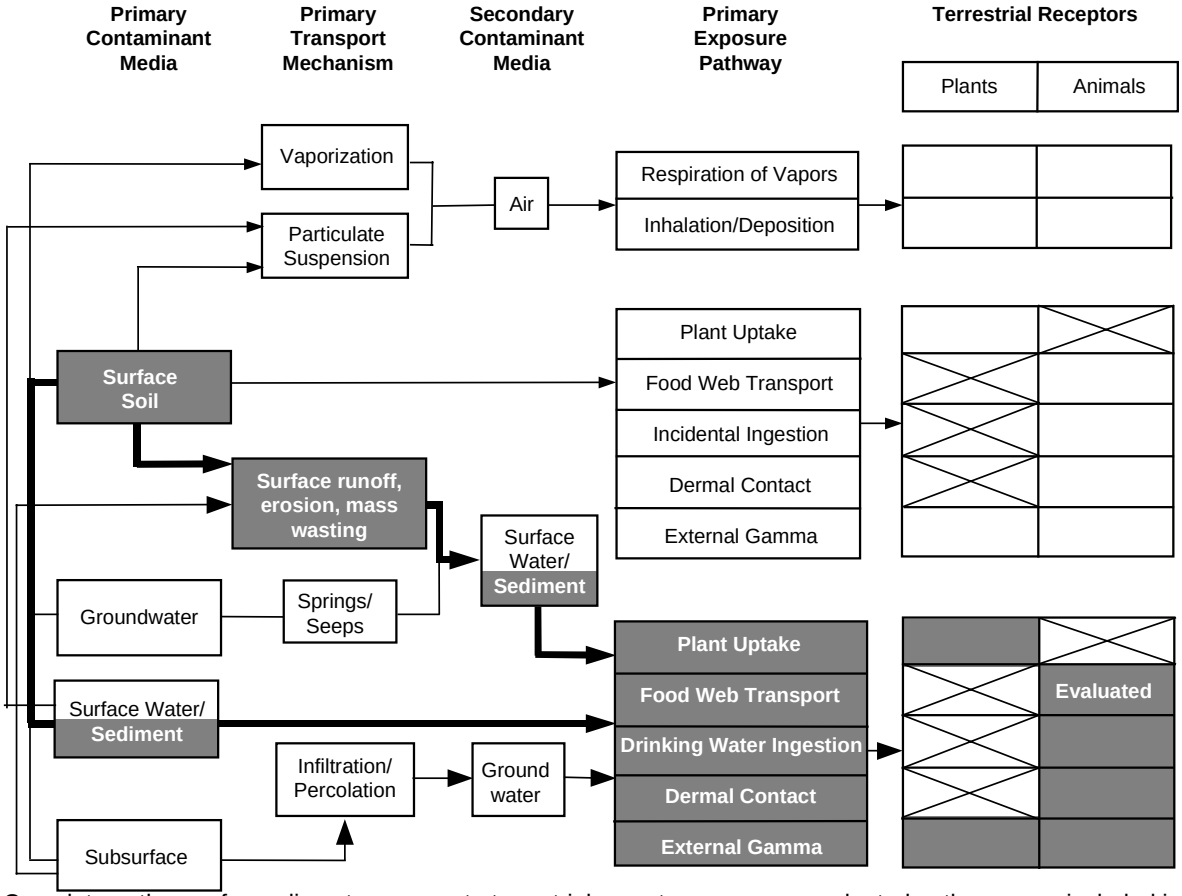
5. Sediments are an integral part of the aquatic environment, providing habitat, feeding, and rearing areas for many aquatic organisms.

The general methodologies adopted for screening aquatic receptors to contaminated sediments are in conformity with those proposed by the EPA for the development of *ecotox thresholds* (EPA 1996, ER ID 62792). Methods for screening sediments are founded on the basis that aquatic organisms are generally exposed to the greatest fraction of contamination by means of direct media contact, i.e., continuous bodily contact with sediments. Thus, the exposure pathways evaluated to aquatic receptors (using EPA methods) include bioconcentration and external gamma exposure (Figure 4.4.2-1). Aquatic ecological screening pertains to receptors that are generally associated with benthic surfaces. In order to be generally protective of aquatic plant and animal species, however, the EPA methods used in this document have been derived with the intent of protecting a large fraction (roughly 95%, unless otherwise stated) of species found in aquatic environs at large. By protecting most aquatic species, the particular species selected to be representative of feeding guilds in the aquatic realms of the Laboratory is also presumed to be protected.



Complete pathways for sediment exposure to aquatic receptors are gray; evaluated pathways are included in the sediment ESL calculations.

Figure 4.4.2-1. Aquatic conceptual model for sediment pathways.



Complete pathways for sediment exposure to terrestrial receptors are gray; evaluated pathways are included in the sediment ESL calculations to account for bioaccumulation to insectivores.

Figure 4.4.2-2. Terrestrial conceptual model for sediment pathways (to account for bioaccumulation concerns).

Although sediment ESLs are primarily developed to protect against potential effects on aquatic receptors, pathways from sediment to terrestrial receptors are also evaluated to ensure that bioaccumulation concerns have been addressed. A simple wildlife exposure model is developed in order to evaluate bioaccumulation potential of COPCs in sediments to aerial insectivores (bat and swallow) via emergent insects. The terrestrial receptor exposure model for sediment pathways is provided in Figure 4.4.2-2. This conceptual model indicates that several exposure pathways are complete, but only the food web transport pathway is evaluated, as other pathways are considered minor. Additionally, the uptake of COPCs from sediments is considered much more severe for aquatic plants and animals in direct contact with the sediment medium, which is covered by the sediment pathways model (Figure 4.4.2-1) and screening methods.

Sediment ESLs, for the protection of aquatic life, are derived from the wealth of information regarding direct effects of contaminated sediments on aquatic organisms. The adoption of the final sediment ESLs involves the consideration of multiple methods for the calculation of sediment screening benchmarks (SSBs). SSBs 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 differing measurement endpoints. Values from studies using freshwater sediments have been assigned highest priority, and only values endorsed by various EPA entities have been considered. Based upon EPA and other methods (EPA 1995, ER ID 62787; EPA 1996, ER ID 62792; EPA 1996, ER ID 62877; EPA 1996 ER ID 62794; Jones et al. 1997, ER ID 62789), the Laboratory ranks SSBs in the following order:

1. SQCs calculated from national ambient water quality criteria or from Great Lakes Tier I water quality criteria (EPA 1995, ER ID 62787) according to EPA (1996, ER ID 62792);

2. SQBs calculated from Great Lakes Tier II water quality criteria (EPA 1995, ER ID 62787) according to EPA (1996, ER ID 62792);

3. Sediment Effects Concentrations (SECs) derived from EPA methods described below (EPA 1996, ER ID 62877);

4. EPA Region IV screening values (EPA 1996, ER ID 62794) and,

5. Other [e.g., Jones et al. (1997, ER ID 62789)]

Sediment will be screened utilizing SSBs in the above rank order. For example, if an SQC is calculable from ambient water quality criteria or Tier I water quality criteria (EPA 1995, ER ID 62787), then this will become the preferred criterion; if no SQC is calculable, then an SQB will be used, also if calculable, and so on. Table 4.4.2-1 shows the way that SSB values and wildlife ESLs are used to derive a final sediment ESL. Note that the approach outlined in Table 4.4.2-1 holds for any number of SSBs; i.e., if more than three sources are viewed to be acceptable, additional columns are added after SSB3. When SSBs and wildlife ESLs are unavailable or not calculable for a given COPC, then the COPC will be retained as a COPEC and discussed in the uncertainty analysis section. The uncertainty analysis should benefit from knowledge of all acceptable SSBs and the wildlife sediment ESLs as they are available and applicable to the site-specific conditions.

In the paragraphs to follow, we summarize the salient information from a vast array of information for the selection of SSBs and describe the method used to calculate the wildlife ESL. The SSB methods are summarized, and some of the definitions are expanded upon, in Jones et al. (1997, ER ID 62789). In addition, we have provided a question and answer logic for aiding an individual in choosing a method on which to base selection of a SSB for each COPC. For a complete exposition of calculations used to derive the various criteria/benchmarks, please refer to the source documents cited herein.

Table 4.4.2-1.
Strategy for obtaining the final non-radionuclide sediment ESL.

Contaminant	SSB1[a] (mg/kg)	SSB2[b] (mg/kg)	SSB3[c] (mg/kg)	Minimum Wildlife[d] ESL (mg/kg)	Final Sediment ESL (mg/kg)
U	Value[e]	Value	Value	Value	Minimum of SSB1 and wildlife ESL
V	No value[f]	Value	Value	Value	Minimum of SSB2 and wildlife ESL
W	No value	No value	Value	Value	Minimum of SSB3 and wildlife ESL
X	No value	No value	No value	Value	Equals minimum wildlife ESL, address uncertainty of no aquatic organism toxicity information
Y	No value	No value	No value	No value	No sediment ESL available or calculable; retain COPC as COPEC
Z	Value	Value	Value	No value	Equals SSB1, address uncertainty of no wildlife ESL

a. SSB1 = primary SSB value
b. SSB2 = secondary SSB value
c. SSB3 = tertiary SSB value
d. Minimum ESL of two aerial insectivores (bat and swallow)
e. Value = value available for that COPC
f. No value = no value available for that COPC

SSB1: Sediment Quality Criteria

- Is the COPC a nonionic organic substance?
- Do National Ambient Water Quality Criteria (NAWQCs) exist for the COPC?
- If YES, for both of the above, follow the directions outlined below for generating SQCs in the calculation of ESLs.
- If NO, to either of the above, go to the SSB2 section.

The preferred means of generating SQCs is the "equilibrium partitioning method" (EqP) proposed by the EPA (EPA 1996, ER ID 62792). SQCs have been proposed by EPA's Office of Water for acenaphthene, dieldrin, endrin, fluoranthene and phenanthrene (EPA 1996, ER ID 62792). A number of other SQCs have been proposed by EPA (1996, ER ID 62792) and Jones et al. (1997, ER ID 62789). 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 *nonionic organics only*. The relationship between K_{ow} and the sediment organic carbon partitioning coefficient, K_{oc}, is described by the following Equation 4.4.2-1 (Di Toro 1985, ER ID 62876):

$$\log_{10} K_{oc} = 0.00028 + 0.983 \log_{10} K_{ow} \qquad \textbf{Equation 4.4.2-1}$$

The EqP method assumes that pore water is in equilibrium with sediment and that pore water must meet water quality standards in order to be considered nontoxic (O'Connor et al. 1998, ER ID 62790). 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 one ignores the colloids or suspended solids available for direct ingestion to wildlife, 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, ER ID 62789) state that "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, ER ID 62789):

1. Partitioning of the organic chemical 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 benchmarks) are similar;

3. The levels of protection afforded by water quality benchmarks (WQBs) are appropriate for benthic organisms; and,

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 the use of water quality benchmarks for the derivation of sediment quality benchmarks (Jones et al. 1997, ER ID 62789). Because of complexities associated with metal binding in sediments (e.g., metal binding sites other than organic carbon – e.g., clay surfaces), the EqP approach is inappropriate for use with metals.

The equation for the SQC (mg/kg*) is:

$$SQC = f_{oc} \times K_{oc} \times (FCV \text{ or } CCC) \qquad \textbf{Equation 4.4.2-2}$$

Where: f_{oc} (unitless) is the mass fraction of organic carbon for the sediment.
K_{oc} (unitless) is the sediment organic carbon partitioning coefficient.
FCV (µg/L) is the "final chronic value" from chronic ambient water quality criteria (AWQC) (AWQC; Tier I toxicity values). See Section 4.4.3, Water ESLs, and EPA (1995, ER ID 62787) for details on the calculation of FCVs.
CCC (µg/L) is the "criterion continuous concentration". See EPA (1998, ER ID 62791) for recommended CCC values and EPA (1995, ER ID 62787) for details on the calculation of CCCs.

**Because we are considering freshwater toxicity information for the derivation of sediment TRVs, we can make the assumption that a liter of freshwater weighs a kilogram for purposes of unit conversion.*

Utilizing the above relationships, SQCs can be derived for any number of organic substances. The method for SQC calculation utilizes normalized calculation of the f_{oc} to 1%. With this normalization, the SSB1 can be derived from the SQC, adjusted to site-specific conditions by a simple factor of:

$$SSB1 = \left(f_{oc\,[\text{site specific}]} \Big/ f_{oc\,[\text{normalized}]} \right) \times SQC\ .$$ **Equation 4.4.2-3**

Obviously, utilizing the SQCs published by the EPA and other sources as a basis for SSB1 adoption, requires a knowledge of site-specific conditions at the Laboratory due to varying levels of the f_{oc}. Under most circumstances, the f_{oc} will be greater than 1%, thus the SSB1 will be greater than the SQC. The EqP method is not valid when the f_{oc} is less than 0.2%.

SSB2: Sediment Quality Benchmarks

- Is the COPC a nonionic organic substance?
- Do Tier II level water quality criteria 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 SSB3 section.

When Tier I AWQCs are unavailable for calculating SQCs, SQBs are generated utilizing Tier II secondary chronic values (SCVs) for water (see Section 4.4.3, Water ESLs). SQBs are calculated utilizing the identical mathematical relationships of SQCs with the substitution of SCV for the FCV/CCC in Equation 4.4.2-2. The SQB method is also only appropriate for nonionic organic compounds, and those with log K_{ow} values between 2.0 and 5.5 (EPA 1996, ER ID 62792). Since both SQCs and SQBs are directly dependent upon K_{ow} values, reliable sources for this information are necessary (e.g., EPA 1995, ER ID 62806; MacKay et al. 1992-1997, ER ID 62903). Criteria for ranking K_{ow} values from the primary literature are also provided in the Final Water Quality Guidance for the Great Lakes System (EPA 1995, ER ID 62787).

SSB2 values are calculated as follows:

$$SSB2 = \left(f_{oc\,[\text{site specific}]} \Big/ f_{oc\,[\text{normalized}]} \right) \times SQB$$ **Equation 4.4.2-4**

Jones et al. (1997, ER ID 62789) provides some SQBs for ionic organic compounds. As Jones et al. (1997, ER ID 62789) indicate, ionic organic compounds have not been well studied for their equilibrium partitioning properties in the water-sediment interface. It is likely that the Oak Ridge National Laboratory (ORNL) benchmarks (Jones et al. 1997, ER ID 62789) are conservative, as the fraction of ionic substances sorbed to the organic carbon surfaces is likely to be greater than that for nonionic substances. Thus, the ORNL benchmarks may be used as a tentative reference for these compounds. It is noteworthy that other factors may affect the sorption capacity of sediment for ionic compounds, including pH (Jafvert 1990, ER ID 62904).

SSB3: Great Lakes Water Quality Criteria Sediment Effects Concentrations

- Is the COPC a nonionic organic substance for which 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 SSB3.

- If **NO**, to either of the above, go to the SSB4 section.

If a Tier-I level SQC or a Tier-II level SQB cannot be calculated, values from EPA's "Calculation and Evaluation of Sediment Effect Concentrations on the Amphipod *Hyalella azteca* and the Midge *Chironomus riparius*" should be used (EPA 1996, ER ID 62877). Sediment effects concentrations (SECs) were derived by the National Biological Service in response to the needs of EPA for the development of sediment benchmarks 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 (FDEP) 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 utilizing a single benchmark, as multiple benchmarks are employed for the derivation of a single SEC value. The Laboratory generally recommends using TELs or ERLs as SECs, in order to minimize 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 the ERM, TEL, or AET may be better suited for adoption as an SSB. For example, Jones et al. (1997, ER ID 62789) recommend the use of an ERM or AET for aluminum SEC, as the variability of these values was minimal compared to others. In brief, the following definitions apply (EPA 1996, ER ID 62877):

- ERL – the sediment COPC concentration at which 10% of the test population was observed with effects (similar to a TEL, below).

- ERM – the sediment COPC concentration at which 50% of the test population was observed with effects (similar to a PEL, below).

- TEL – the upper limit of the range of sediment COPC concentrations dominated by no effects data.

- PEL – the lower limit of the range of COPC concentrations that are usually or always associated with adverse biological effects.

- AET – the sediment chemical concentration above which statistically significant biological effects always occur.

In short, the test organisms used were the amphipod *H. azteca* and the midge *C. riparius*. Tests on these organisms were conducted utilizing sediment samples from a large number of freshwater sites. Measurement endpoints 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 common widespread benthos over much of North America, including New Mexico. Each organism is broadly representative of crustacean and insect invertebrates, respectively, that dominate lentic and lotic systems, such as those at the Laboratory. These organisms are not considered tramp species, rather they are part and parcel of many intact aquatic systems. As a consequence of their ubiquity and their place in healthy aquatic systems, these organisms are considered to be adequate choices for broad-based protection of aquatic organisms at large, including the Laboratory. Additionally, the SEC project has undergone close scrutiny by EPA, the Natural Resource Trustees of the Great Lakes Systems, and by Great Lakes System Stakeholders, and has been found to

be of adequate rigor to serve as a nominal model for freshwater systems nationwide (Jones et al. 1997, ER ID 62789). Jones et al. (1997, ER ID 62789) recommend SECs to be adopted as SSBs for organic COPCs not covered by the SQC or SQB methods, and for metals.

SSB4: EPA Region IV Sediment Screening Values

- Is the COPC a nonionic organic substance for which Tier I and Tier II and EPA (1996, ER ID 62877) (SEC) data are lacking?

- Is the COPC a metal for which EPA (1996, ER ID 62877) (SEC) data are lacking?

- If **YES,** *for either of the above*, follow the directions below for the adoption of EPA Region IV sediment screening values as a SSB.

- If **NO**, *to either of the above*, research literature, if available.

If data in the above resources are lacking for a particular toxicant, then the Laboratory will refer to EPA Region IV's sediment screening values (EPA 1996, ER ID 62794) as a source for the adoption of SSBs. These values are based on sediment toxicity work performed on marine sediments (e.g., Long et al. 1995, ER ID 62793). 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 1996, ER ID 62794; EPA 1996, ER ID 62792). 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, ER ID 62796). In many cases, because the practical quantitation limit (PQL) for identification of organic compounds was greater than the benchmark values, the benchmark value defaults to the PQL. Such methods need to be regarded cautiously, and in the case where the PQL is used as a default value, sound chemistry-related justification should be provided for the use of a PQL as a benchmark value. Note that if a PQL 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.

Other SSB Resources

In the absence of any of the above criteria, one should compare whatever available information there may be on any given constituent (e.g., from primary sources). 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 SSB. The Ontario Ministry of the Environment sediment quality guidelines (Persaud et al. 1993, ER ID 62875) is potentially an additional resource. Jones et al. (1997, ER ID 62789) also provide references for other resources. Measurement endpoints mentioned above may be insensitive (e.g., species absence), even for lowest effects levels (the level at which toxic effects become apparent).Consequently, SSBs adopted from sources other than recommended in SSB1-SSB4 should be discussed in the uncertainty analysis portion of screening (see Section 4.6, Screening Evaluation/Uncertainty Analysis).

Sediment Exposure to Terrestrial Receptors

To address transport of COPCs from sediment through the food chain, a wildlife ESL model has been developed (the methods described above do not explicitly account for trophic transfer concerns). This

model is based on Equation 4.4.1-7, which is the insectivore soil ESL model described in Section 4.4.1, Soil ESLs. The model shown in Equation 4.4.2-5 is based on the transfer of contamination from sediments to benthic insects, and the subsequent ingestion of the insects (by an insectivore) as contaminated food. The insectivores under consideration for this model are the bat and the swallow. The exposure information for these receptors is provided in Table 4.3-5. Transfer to higher level carnivores is not accounted for by these ESLs and should be addressed in the uncertainty analysis, Section 4.6.

$$ESL_{ij} = \frac{0.3 \cdot NOAEL_{ij}}{I_i \cdot TF_{invert,j}}$$ **Equation 4.4.2-5**

Where: ESL_{ij} is the sediment ESL for insectivore i and COPC j (mg/kg),
$NOAEL_{ij}$ is the NOAEL for insectivore i and COPC j (mg/kg/day),
I_i is the normalized daily dietary ingestion rate for insectivore i (kg/kg/day])
$TF_{invert,j}$ is a unitless transfer factor from sediment to invertebrate for COPC j

Summary of Sediment ESLs Derivation

Sediment ESLs may be derived from a variety of sources (Table 4.4.2-2), and there may be more than one SSB available for any given constituent (see Table 4.4.2-1). Additionally, ESLs are developed for aerial insectivores based on models that differ from those used to derive SSBs. In screening, the SSB chosen for an ESL for a given constituent is compared with the ESL derived for the aerial insectivore model, The lowest of the values is used as the sediment ESL. This ensures that bioaccumulation concerns are addressed by the final ESL.

The main parameters introduced in Section 4.4.2 are summarized in Table 4.4.2-3. Jones et al. (1997, ER ID 62789) provides an excellent reference for consideration of uncertainties and limitations associated with the basis and generation of sediment ESLs. Of particular importance to LANL is the fraction of organic carbon in LANL sediments. This is an environmentally variable parameter but it can be measured directly, and fairly easily, on a site-specific basis.

Table 4.4.2-2.
Summary of sources for sediment ESLs.

COPC Type	Aquatic Receptors	Aerial Insectivores
Nonionic organic chemicals K_{ow} in range of 2 to 5.5	The following are used in order of preference: 1. SQC (EPA [1996, ER ID 62792] using Tier I AWQC from EPA [1998, ER ID 62791] or EPA [1995, ER ID 62787]) 2. SQB (EPA [1996, ER ID 62792] a using Tier II SCVs from EPA [1995, ER ID 62787]) 3. SEC (EPA 1996, ER ID 62877) 4. EPA Region IV (EPA 1996, ER ID 62794) 5. Other (e.g., Jones et al. 1997, ER ID 62789, Long et al. 1995, ER ID 62793)	Eq. 4.4.2-5 with NOAEL – primary toxicity literature for each COPC TF – based on site-specific information, literature values, or simple models I – as provided in Table 4.3-5
Inorganics	The following are used in order of preference: 1. SEC (EPA 1996, ER ID 62877) 2. EPA Region IV (EPA 1996, ER ID 62794) 3. Other (e.g., Jones et al. 1997, ER ID 62789, Long et al. 1995, ER ID 62793)	

Table 4.4.2-3.

Summary of variables used in the sediment ESL models.

Variable	Source
SSB	Sediment Screening Benchmark (μg/kg). SSBs are generic and may be derived from variable sources (e.g., SQC from EPA's (1996, ER ID 62792) methods) ranked according to criteria outlined above.
SQC	Sediment Quality Criteria (μg/kg). SQCs are based on NAWQC or their equivalent (Tier I AWQC). SQCs are intended to prevent significant toxic effects in most chronic exposures.
SQB	Sediment Quality Benchmark (μg/kg). SQBs are based on Tier II Ambient Water Quality Criteria. Tier II values were developed so that sediment quality criteria (in general) could be established with fewer data than are required for NAWQC. It is expected that Tier II values would be higher than NAWQC in no more than 20% of the cases.
K_{ow}	Octanol/water partition coefficient (dimensionless). This can be an empirically determined parameter (most desirable) or, alternatively, it can be obtained by mathematical modeling (least desirable).
f_{oc}	Mass fraction of organic carbon in the sediment. This is a site-specific parameter but is commonly normalized to 1% in the absence of site-specific information.
K_{oc}	Sediment organic carbon partitioning coefficient.
FCV	Final chronic value. This is a calculated estimate of the concentration of a test material such that 95% of the genera have (on average) higher chronic values.
CCC	Criterion Continuous Concentration. This is the NAWQC for chronic exposure. The CCC is an estimate of the highest concentration of a material in the water column to which an aquatic community can be exposed indefinitely, without resulting in detrimental effect.
NOAEL	Receptor and COPC specific values are obtained from reviewing primary ecotoxicity literature. Values for specific receptors and COPCs are provided in the ECORISK Database. To provide bounding information on effects a lowest observed adverse effect level (LOAEL) is also developed for wildlife receptors. Information on the LOAEL for specific receptors is provided in the ECORISK Database.
I_i	Body weight normalized food intake for wildlife receptors, see values provided in Table 4.3-5. Body weight is an implicit component of this variable. For this reason Table 4.3-5 provides body weight for each receptor. Note that intake can also be expressed as a gross daily amount (in units of kg of food ingested per day); however this alternative formulation of the model requires body weight to be an explicit variable.
TF_{invert}	The transfer from soil to invertebrates is a COPC-specific value that is derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database

4.4.3 Water ESLs

Water of potential concern to ecological receptors at the Laboratory includes surface water and shallow groundwater. For the purposes of ecological screening, only exposure pathways related to surface water are evaluated. For those sites where exposure to shallow groundwater is an issue, a discussion of this exposure medium should be included in the uncertainty evaluation.

Water samples may be either filtered (suspended solids removed) or unfiltered. Unfiltered samples will have greater or equal concentrations of COPCs than filtered samples. As a conservative measure of potential exposure, the unfiltered water can be used in screening evaluations. If the unfiltered samples show no potential risk, no further evaluation of the filtered samples is needed. If the unfiltered samples show a potential problems, water samples for inorganic constituent content should be evaluated on the basis of filtered samples, as these are considered the bioavailable fraction of these constituents in water (EPA 1996, ER ID 62792).

Methods for screening water are based on exposure pathways to aquatic and terrestrial organisms. For aquatic organisms, the screening 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 with water. Ecological screening for waterborne COPCs pertains to receptors that are associated with benthic surfaces and the free water column of both lentic and lotic systems. In order to be broadly protective of aquatic plant and animal species methods have been derived by the EPA (EPA 1995, ER ID 62787; EPA 1996, ER ID 62792) with the intent of protecting a large fraction (roughly 95%, unless otherwise stated) of species found in aquatic environs. By using the EPA methods, it is assumed that any particular species selected to be representative of feeding guilds in the aquatic realms of the Laboratory will be protected. The exposure model for water pathways to aquatic receptors is provided in Figure 4.4.3-1. To evaluate potential effects of contaminated water on terrestrial receptors, a simple wildlife exposure model is developed (Figure 4.4.3-2). The terrestrial conceptual model is based on exposure to contaminated drinking water. Inclusion of this model addresses bioaccumulation concerns that are not addressed directly by the EPA methods.

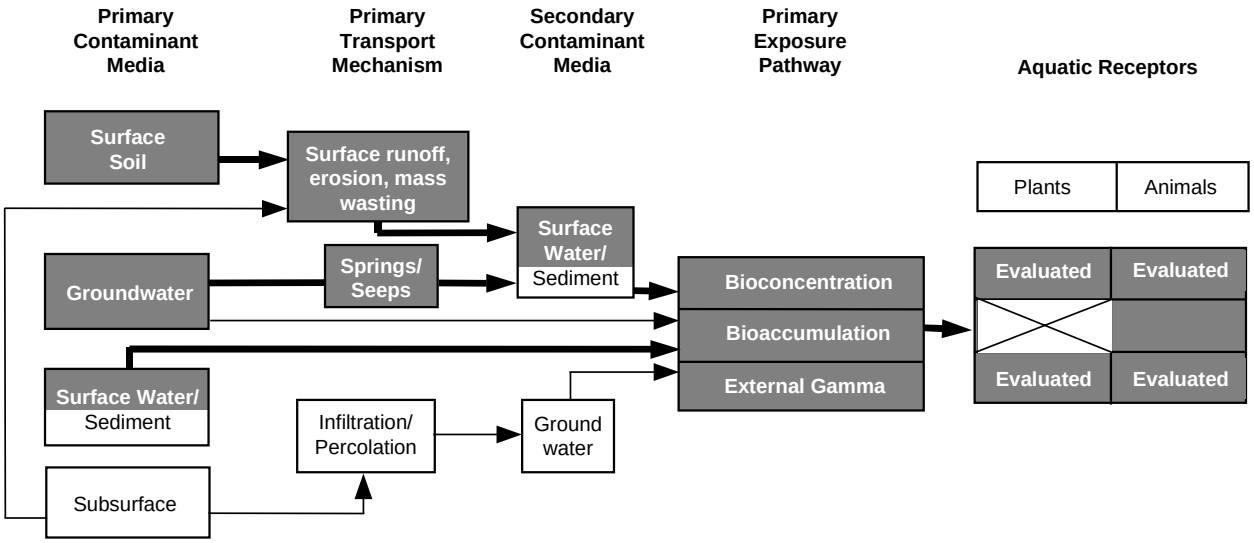
The consideration of impacts from waterborne contamination to aquatic receptors requires the evaluation of a number of water quality criteria (WQCs). WQCs come from a variety of sources, but are all based upon toxicological information from primary studies; only values endorsed by various EPA approved sources have been considered herein. These criteria differ in the methods for their development and/or in the rigor of their development. Consequently, WQCs must be evaluated in a hierarchical fashion, based upon evaluation of their conservatism or certainty for the protection of approximately 95% of aquatic species.

For any single COPC, there may be more than one WQC, but for screening purposes, they will be chosen in the following order:

1. Chronic AWQC set forth by EPA (1998, ER ID 62791) and Sections 3101.J and 3101.N of the New Mexico water standards (NMED 1994, ER ID 62874);

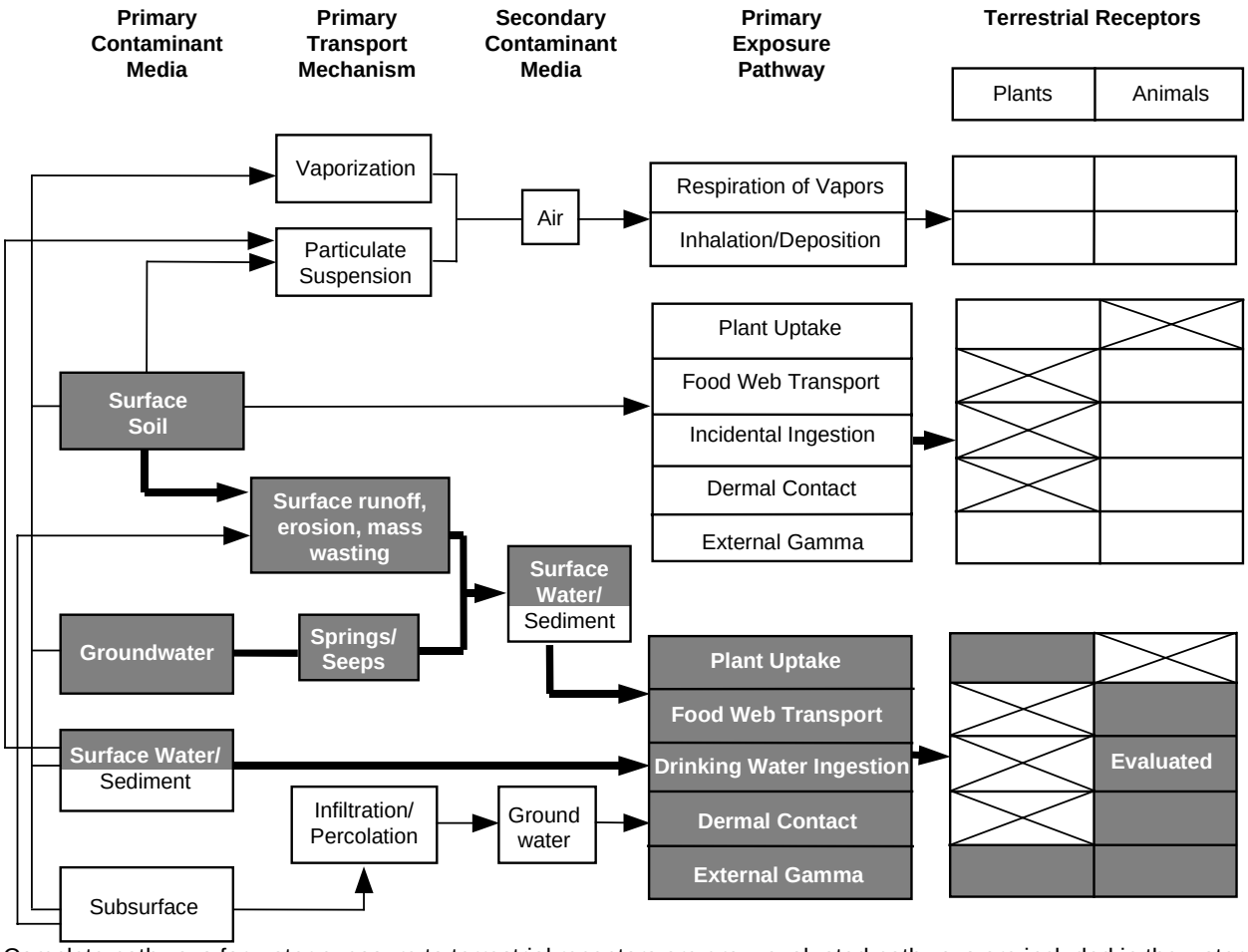
2. Great Lakes methodology Tier I final chronic value (EPA 1995, ER ID 62787);

3. Great Lakes methodology Tier II chronic value (CV) (EPA 1995, ER ID 62787).



Complete pathways for water exposure to aquatic receptors are gray; evaluated pathways are included in the water ESL calculations.

Figure 4.4.3-1. Aquatic conceptual model for water pathways.



Complete pathways for water exposure to terrestrial receptors are gray; evaluated pathways are included in the water terrestrial (bioaccumulation) ESL calculations.

Figure 4.4.3-2. Terrestrial conceptual model for water pathways.

Justification for selecting the above order is provided in greater detail in EPA (1995, ER ID 62787), EPA (1996, ER ID 62792), and Sections 3101.J and 3101.N of the New Mexico water standards (NMED 1994, ER ID 62874). For a complete discussion, the reader is directed to these resources. If more than one WQC exists for a given COPC, then the information implicit in these values will be discussed in the uncertainty analysis (see Section 4.6, Screening Evaluation/Uncertainty Analysis).

Water is screened utilizing WQCs in the rank order presented above. For example, if NM and/or EPA chronic ambient water quality criteria are available for a given constituent, then it will be selected as the most relevant WQC for screening. If there is no NM and/or EPA ambient water quality criterion, but a Tier I WQC is calculable (EPA 1995, ER ID 62787), then this becomes the preferred criterion. Lastly, if no Tier I WQC is calculable, then a Tier II value will be used. One may evaluate all of the acceptable WQC and wildlife ESLs that are available and applicable to the site-specific conditions in the uncertainty analysis. Table 4.4.3-1 shows how WQC values and wildlife ESLs are used to derive a final water ESL. When WQCs and wildlife ESLs are unavailable or not calculable for a given COPC, then the COPC will be retained as a COPEC and discussed in the uncertainty section.

Table 4.4.3-1.
Hierarchy for obtaining final non-radionuclide water ESL.

COPC	WQC[a] (μg/L)	Minimum Wildlife ESL[b] (mg/L)	Final Water ESL (μg/L)
T	Chronic AWQC (WQC1)	Value	Minimum of WQC and wildlife ESL
U	Tier I value (WQC2)	Value	Minimum of WQC and wildlife ESL
V	Tier II value (WQC3)	Value	Minimum of WQC and wildlife ESL
W	No value	Value	Equals wildlife ESL, address uncertainty of no aquatic toxicity information
Y	Value	No value	Equals WQC, address uncertainty of no wildlife ESL
X	No value	No value	No water ESL, retain COPC as COPEC

a. WQC = water quality criterion
b. Based on wildlife exposure calculation

In the paragraphs to follow, we highlight the salient information for the selection of water ESLs from a larger set of possibilities. Methods for the selection of water ESLs are presented in the order of preference, from those most preferable to those least preferable. In addition, we have provided a question and answer logic to assist the user in choosing a method on which to base the calculation of water ESLs. For a complete exposition of calculations used to derive the various criteria/benchmarks, please refer to the documents cited herein. Suter (1996, ER ID 62805) presents a list of many NAWQ criteria, Tier II values, and other TRVs.

WQC1: Chronic Ambient Water Quality Criteria

- Are chronic ambient water quality criteria available, as set forth by EPA (1998, ER ID 62791) and Sections 3101.J and 3101.N of the New Mexico water standards (NMED 1994, ER ID 62874)?

- If **YES**, utilize the criterion as the WQC1.

- If **NO**, go to the WQC2 section.

AWQCs are developed by EPA's Office of Water (OW) under the Clean Water Act, Section 304 (EPA 1998, ER ID 62791). New Mexico has developed similar criteria for "high quality coldwater fisheries" as listed in "Standards for Interstate and Intrastate Streams," Sections 3101.J and 3101.N (NMED 1994, ER ID 62874). The development of AWQCs is outlined in EPA (1995, ER ID 62787). AWQC values are considered applicable or relevant and appropriate requirements (ARARs) and therefore should be considered foremost for water ESL adoption (Sample et al. 1998, ER ID 62807). AWQCs have been developed for chronic exposure of aquatic organisms to waterborne toxicants, but for only a few inorganic and organic chemicals (EPA 1996, ER ID 62792; NMED 1994 ER ID 62874). Metals are often water hardness-dependent and should be adjusted for site-specific conditions (see EPA [1996, ER ID 62792] and Sections 3101.J and 3101.N of the New Mexico water standards [NMED 1994, ER ID 62874] for explanations/delineation of methods, as methods require analyte-specific information). EPA recommends that Tier I WQCs be developed in the absence of AWQCs; this methodology is discussed next.

WQC2: Tier I Water Quality Criteria

- Can Tier 1 WQCs be derived? (See criteria for derivation below.)
- If **YES**, utilize the methods described herein for derivation of WQC2.
- If **NO**, go to the WQC3 section.

Tier I chronic values (CVs) can be determined utilizing methods of the Great Lakes Water Quality Initiative as detailed in EPA (1995, ER ID 62787). The discussion involving the acceptability criteria and details of the determination of Tier I values are too complex to reiterate in this document, and the reader is referred to EPA (EPA 1995, ER ID 62787, pp. 15395-15399). In general, however, the Laboratory will use the methods described therein while utilizing information obtained from the AQUIRE database (AQUIRE 1997, ER ID 62898) Similar information may be available in primary literature. The fundamental requirements and methods for deriving Tier I criteria are outlined below.

To derive Tier I criteria (EPA 1995, ER ID 62787), results of acceptable chronic tests (see criteria for "acceptable", EPA [1995, ER ID 62787, p. 15397]) must be used. In addition, at least one test result must follow from *each* of the following taxonomic categories:

1. The family Salmonidae in the class Osteichthyes;
2. One other family in the class Osteichthyes;
3. A third family in the phylum Chordata (e.g., fish, amphibian);
4. A planktonic crustacean (e.g., a cladoceran, copepod);
5. A benthic crustacean (e.g., ostracod, isopod, amphipod, crayfish);
6. An insect (e.g., mayfly, dragonfly, damselfly, stonefly, caddisfly, mosquito, midge);
7. A family in a phylum other than Chordata or Arthropoda (e.g., Annelida, Molluska, Rotifera);
8. A family in any order of insect or any phylum not already represented.

For each species 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. 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 (1995, ER ID 62787), corresponds to a NOEC and the upper chronic limit corresponds to a lowest observed effect concentration (LOEC). 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 the genus. When these data are compiled:

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, assign them successive ranks;

3. Calculate an empirical cumulative probability (P) for each GMCV as r/(n+1);

4. Select the four GMCVs which have cumulative probabilities closest to 0.05;

5. Using the four selected GMCVs and Ps, calculate a final chronic value (FCV; EPA 1995, ER ID 62787).

FCVs are calculated utilizing the following mathematical relationships:

$$S^2 = \frac{\left[\sum_1^4 (\ln GMCV)^2\right] - \frac{\left[\sum_1^4 (\ln GMCV)\right]^2}{4}}{\left[\sum_1^4 P\right] - \frac{\left[\sum_1^4 \sqrt{P}\right]^2}{4}}$$ **Equation 4.4.3-1**

$$L = \frac{\left[\sum_1^4 \ln GMCV\right] - \left[S \cdot \sum_1^4 \sqrt{P}\right]}{4}$$ **Equation 4.4.3-2**

$$A = L + S \cdot \sqrt{0.05}$$ **Equation 4.4.3-3**

$$FCV = e^A$$ **Equation 4.4.3-4**

The FCV is the final chronic value; *e* is the natural base, while all other parameters are incidental to the calculation. If, however, site-specific considerations identify a species of importance or special concern has a SMCV lower than the FCV, then this SMCV must be used in place of the FCV as the Tier I criterion.

Tier I FCVs are also calculable based on acute values, This method utilizes acute values, calculating the species mean acute value (SMAV) and genus mean acute value (GMAV) in the same manner as the SMCV and GMCV above. To obtain a final acute value (FAV), Equations 4.4.3.1-1 through 4.4.3.1-4 are used substituting GMAV for GMCV. The FCV is derived by dividing the FAV by the final acute-chronic ratio (FACR). The FACR is the geometric mean of at least three acute chronic ratios such as LC_{50}/CV ratios. The LC_{50}/CV ratio is the acute-chronic ratio (ACR) where the LC_{50} is the lethal concentration of 50% of the experimental population, as determined by an acute toxicity test, and the CV is the chronic value determined for the same organism in the same study. Each ratio is derived from a test on one species. However, the FACR must be calculated from ratios derived from at least three different aquatic taxa. The aquatic taxa referred to must conform to the following:

1. At least one is a fish;

2. At least one is an invertebrate; and,

3. At least one species 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:

1. For each species, calculate the species mean acute-chronic ratio (SMACR) as the geometric mean of the ACRs available for all species. (The requirements for meeting the ACR criteria are very specific and demanding [see EPA 1995, ER ID 62787, p. 15398]).

2. Calculate the FACR as the geometric mean of the SMACRs.

3. Calculate an FAV.

4. Calculate the FCV by dividing the FAV by the FACR.

The final Tier I criterion is usually considered the FCV; however, there are exceptions. If one-half of the FAV is lower than the FCV, then that value should be used as the Tier I criterion. Also, if the FCV exceeds the final plant value (FPV), then the FPV should be used as the Tier I criterion. The FPV (EPA 1995, ER ID 62787, 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."

In the end, the Tier I CV for waterborne contamination, may be chosen as the ESL. In the case where a Tier I criterion may not be developed, then a Tier II criterion may be adopted, as outlined next.

WQC3: Tier II Water Quality Criteria

- If no AWQC exists, and a Tier I Water Quality Criterion cannot be derived, the following methods may be used to evaluate a Tier II Water Quality Criterion.

If three or more experimentally determined ACRs (see above) are available for the COPC, determine the FACR as described above. If fewer than three ACRs are calculable, assume that each "missing" ACR value is equal to 18, so that the total number of "ACRs" equals three. Calculate a secondary acute-chronic ratio (SACR) as the geometric mean of the three ACRs. Calculate the secondary chronic value (SCV) using one of the following equations: Equation 4.4.3-5 utilizing the SACR and FAV as calculated under Tier I methods; Equation 4.4.3-6 utilizing methods for calculating a secondary acute value (SAV) as outlined below, and the FACR, as outlined under Tier I methods; and Equation 4.4.3-7 utilizing the SAV and SACR.

$$SCV = \frac{FAV}{SACR}$$ **Equation 4.4.3-5**

$$SCV = \frac{SAV}{FACR}$$ **Equation 4.4.3-6**

$$SCV = \frac{SAV}{SACR}$$ **Equation 4.4.3-7**

SAVs are presented in detail in the Great Lakes document (EPA 1995, ER ID 62787, p. 15400). To calculate a SAV one must utilize, at a minimum, one (1) genus mean acute value (GMAV) for a daphnid (Crustacea: Caldocera), calculated as the geometric mean of the SMAVs available for the genus. The

lowest GMAV calculated is then divided by the secondary acute value factor (SAVF, Table 4.4.3-2). The requirement of at least one daphnid GMAV has been criticized for severely restricting the number of benchmarks that could be calculated (Suter 1996, ER ID 62805). In response to this concern, Suter and Tsao (1996, ER ID 59838) provide SMAVs for calculating SAVs when no daphnid GMAVs are calculable; these are presented in Table 4.4.3-2.

Table 4.4.3-2.
Secondary acute value factors (SAVFs) for estimation of Tier II values.

No. GMAVs[a]	Acute Value from Data Set With Daphnid Values[a]	Acute Value from 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 (1995, ER ID 62787).
b. Factors taken from Suter and Tsao (1996, ER ID 59838).

The lowest of the SCV or the FPV is then considered the Tier II CV. Tier II values are expected to be higher than AWQCs in no more than 20% of all cases. The Tier II CV can then be adopted as WQC3.

When an AWQC, a Tier I, or Tier II value are unavailable or not calculable, then it becomes necessary to seek out other toxicologically-based benchmarks from other sources, particularly primary literature sources. Suter and Tsao (1996, ER ID 59838) and Suter (1996, ER ID 62805) provide excellent reference to a variety of potential benchmarks and resources. When an AWQC, a Tier I, or a Tier II value are unavailable or not calculable for a given COPC, then the COPC will be carried forward to discussion in the uncertainty section.

Water Exposure to Terrestrial Receptors

To address the drinking water exposure pathway to terrestrial receptors, a wildlife ESL model was developed. This model is based on Equation 4.3-3, which is the general wildlife exposure model. To screen the drinking water pathway, it is assumed that all oral exposure is derived from drinking water. Thus, exposure is calculated with the following model:

$$E_{water} = C_{water} \cdot I_{water} \cdot d_{water} \qquad \textbf{Equation 4.4.3-8}$$

Where: E_{water} is the estimated oral daily dose for a COPC (mg/kg/day),
C_{water} is the concentration of chemical constituent *x* in water (mg/L)
I_{water} is the normalized daily water ingestion rate (L of water / [kg of body weight • day])
d_{water} is the density of water (1 kg/L)

The wildlife water ESL is calculated based on the following equation.

$$ESL_{ij} = \frac{0.3 \cdot NOAEL_{ij}}{I_i \cdot d_{water}}$$ **Equation 4.4.3-9**

Where: ESL_{ij} is the water ESL for wildlife species i and COPC j (mg/L),
$NOAEL_{ij}$ is the NOAEL for wildlife species i and COPC j (mg/kg/day),
I_i is the daily water ingestion rate for wildlife species i (L/kg/day)
d_{water} is the density of water (1 kg/L)

The main parameters introduced in Section 4.4.3 are summarized in Table 4.4.3-3. Suter (1996, ER ID 62805) and Suter and Tsao (1996, ER ID 59838) provides an excellent reference for consideration of uncertainties and limitations associated with the basis and generation of water ESLs.

4.4.4 Multimedia Screening Calculations

The purpose of this part of the screening evaluation is to track the potential combined effects of exposure to multiple media. The end result of this evaluation is the calculation of HI values for all wildlife receptors potentially exposed to site-related COPCs. If the HI for a wildlife receptor species exceeds 1.0, then those COPCs that contribute more than 0.3 to the HI for that receptor are identified as COPECs. The list of such receptors will have been identified in the site scoping checklist, which will be a subset of all possible wildlife receptors for all possible contaminated media provided in Table 4.4.4-1.

Calculation of exposure to wildlife should follow the approach outlined in the wildlife exposure model (Section 4.3, Screening Evaluation Overview). In cases where dermal and respiratory pathways require evaluation, the models provided by Hope (1995, ER ID 62787) should supplement the oral pathways discussed in Section 4.3. In either case, Equation 4.4.4-1 should be used:

$$HI_i = \sum_{j=1}^{n} \frac{D_{ij}}{NOAEL_{ij}}$$ **Equation 4.4.4-1**

Where: HI_i is the hazard index for wildlife receptor i
D_{ij} is the exposure of wildlife receptor i to COPC j from all contaminated media for the site (mg/kg/day)
$NOAEL_{ij}$ is the NOAEL for COPC j and wildlife receptor i (mg/kg/day)
n is the number of COPCs for the site with like effects

Table 4.4.3-3.

Summary of variables used in the water ESL models.

Variable	Source
WQC	Water quality criteria. These are generic criteria representing sources of water criteria (e.g., NAWQC) ranked according to above-stated preferences.
NAWQC/AWQC	National or Tier-I generated AWQC. Tier II values were developed so that aquatic life criteria could be established with fewer data than are required for NAWQC. It is expected that Tier II values would be higher than NAWQC in no more than 20% of the cases.
SAV	Secondary acute value. Tier II numerical acute criteria accounting for uncertainty associated with its use by incorporating a penalty factor. The SAV is penalized in proportion to its deviation from Tier I criteria (i.e., Table 4.4.3-2) to generate conservative TRVs.
SCV	Secondary chronic value.
FACR	Final acute chronic ratio. This is the geometric mean of at least three LC_{50}/CV ratios meeting the requirements specified above.
SACR	Secondary acute chronic ratio. The ratio acute/chronic toxicity values (e.g., LC_{50} and CV) calculated in the absence of all the criteria to generate a FACR.
FAV	Final acute value. The acute NAWQC (or their equivalent Tier I AWQC) are based on one-half of the FAV. The acute NAWQC are intended to correspond to concentrations that would cause less than 50% mortality in 5% of exposed populations in a relatively brief exposure.
FCV	Final chronic value. This is a calculated estimate of the concentration of a test material such that 95% of the genera have (on average) higher chronic values.
CCC	Criterion continuous concentration. This is the NAWQC for chronic exposure. The CCC is an estimate of the highest concentration of a material in the water column to which an aquatic community can be exposed to indefinitely without resulting in an unacceptable effect.
E_{water}	Estimated oral daily dose for a COPC
d_{water}	Density of water
C_{water}	Concentration of chemical constituent *x* in water
NOAEL	Receptor and COPC specific values are obtained from reviewing primary ecotoxicity literature. Values for specific receptors and COPCs are provided in the ECORISK Database. To provide bounding information on effects a lowest observed adverse effect level (LOAEL) will also be developed for wildlife receptors. Information on the LOAEL for specific receptors will be provided in the ECORISK Database.
I_i	Body weight normalized water intake for wildlife receptors, see values provided in Table 4.3-5. Thus, body weight is an implicit component of this variable. For this reason Table 4.3-5 provides body weight for each receptor. Note that intake can also be expressed as a gross daily amount (in units of kg of water ingested per day). This alternate formulation of the model requires body weight to be an explicit variable.

Table 4.4.4-1.

List of wildlife receptors and applicable media for multimedia exposure to non-radiological COPCs.

Receptor	Contaminated Media Evaluated for Receptor		
	Soil	Sediment	Water
Robin	Yes	No	Yes
Kestrel	Yes	No	Yes
Deer mouse	Yes	No	Yes
Desert cottontail	Yes	No	Yes
Shrew	Yes	No	Yes
Red fox	Yes	No	Yes
Bat	No	Yes	Yes
Swallow	No	Yes	Yes

4.5 ESL Calculations for Radiological Constituents

Radionuclide ESLs are calculated based on dose rate received by individual plants and animals. Radionuclide dose is related to the energy of the specific radioactive decay emission and the amount or mass of the radionuclide. Thus, the basic radionuclide dose model is:

$$Dose = Effective\ Energy \cdot Amount \qquad \textbf{Equation 4.5-1}$$

Much of the confusion relating to calculating radionuclide dose relates to the units of the terms in Equation 4.5-1. For calculating radionuclide ESLs, "dose" is expressed in units of rad/day, while the "amount" of the radionuclide is expressed in units of pCi/g, which is an activity (decay per unit time) per unit mass of media or organism. Thus, effective energy has units of rad/day per pCi/g, which suggests that the effective energy term can also be viewed as a dose conversion factor (DCF).

Radionuclide ESLs require calculations to account for the dose received from internal (within the organism) and external (from contaminated media) sources. The basic difference between the radionuclide models and non-radionuclide wildlife models is that the radionuclide models require calculation of the internal concentration or body burden and the non-radionuclide models require calculation of the exposure to the contaminant. There are also simple conversion factors required to account for the effective energy for different types of radionuclides in different media. Table 4.5-1 summarizes the receptors and diets used to model exposure to radionuclides in soil, sediment, and water.

Radionuclide Dose Limits

Radionuclide dose limits are the equivalent of the NOAELs used to develop non-radionuclide ESLs. The International Atomic Energy Agency (IAEA) has concluded that doses protective of human health are protective of ecological resources, with certain exceptions (IAEA 1992, ER ID 62802):

- Human access is restricted but access by biota is not restricted,
- Unique exposure pathways exist,
- Rare or endangered species are present, or
- Other stresses are significant.

For these four special situations, IAEA recommends a dose limit of 0.1 rad per day. As this dose limit is considered appropriately conservative, and is consistent with the results of reviews by the National Council on Radiation Protection (NCRP) (1991, ER ID 62803) and Eisler (1994, ER ID 63043), the Laboratory will adopt 0.1 rads per day as the dose limit for ecological receptors for the purposes of screening. Thus, the basic model for calculating acceptable dose for radionuclides is:

$$Total\ Acceptable\ Dose = 0.1\ (\text{rad/day}) = Internal\ Dose + External\ Dose \qquad \textbf{Equation 4.5-2}$$

Table 4.5-1.
ESL media and screening receptors for radionuclides.

ESL Medium	Receptor Group	Receptor Name	Diet Composition
Soil	Bird	American kestrel	50% invertebrate/ 50% flesh
		American kestrel	100% flesh
		American robin	100% invertebrate
		American robin	50% invertebrate/ 50% plant
		American robin	100% plant
	Mammal	Desert cottontail	100% plant
		Deer mouse	50% invertebrate/ 50% plant
		Red fox	100% flesh
		Vagrant shrew	100% invertebrate
	Plant	Generic Plant	Not applicable
	Invertebrate	Earthworm	Not applicable
Water[a]	Bird	American kestrel	No food, water only[b]
		American robin	No food, water only[b]
		Swallow	No food, water only[b]
	Mammal	Desert cottontail	No food, water only[b]
		Deer mouse	No food, water only[b]
		Red fox	No food, water only[b]
		Vagrant shrew	No food, water only[b]
		Bat	No food, water only[b]
	Aquatic	Algae	Not applicable
		Daphnid	Not applicable
		Snail	Not applicable
		Fish	Not applicable
Sediment [a]	Bird	Swallow	100% invertebrate
	Mammal	Bat	100% invertebrate
	Aquatic	Algae	Not applicable
		Daphnid	Not applicable
		Snail	Not applicable
		Fish	Not applicable

a. Water and sediment ESLs are only used to help evaluate significant exposure pathways and COPCs for those media. In all cases where a site has one of the media contaminated, a multimedia assessment is expected.

b. The water ESL for these terrestrial receptors only reflects the exposure from contaminated water from the site. Therefore, a multimedia exposure assessment may be required to address the potential cumulative effects from soil (or sediment) and water for these receptors.

4.5.1 Soil ESLs

The operational definition of soil was provided in Section 4.4.1, Soil ESLs. Radionuclide soil ESLs are based on exposure of terrestrial receptors to contaminated soil. The final radionuclide soil ESL is the lowest receptor-specific ESL (i.e., minimum) among the ten terrestrial receptors (Table 4.5.1-1). ESLs for the other nine terrestrial receptors should be used as appropriate in the site-specific uncertainty analysis discussion (see Section 4.6, Screening Evaluation/Uncertainty Analysis). ESLs are developed to account for dose from a single radionuclide. Because radiological dose is additive, the radionuclide ESLs must be

adjusted to account for doses from multiple radionuclides. Per instructions from NMED, a simple adjustment factor will be used to account for dose from multiple radionuclides. The final soil ESL will be divided by the number of radionuclide COPCs at a site before using the final soil ESLs in a site specific screening. Thus, if there are five radionuclide COPCs at a site, the ESL will be divided by 5. This approach is both simple and conservative, and any bias introduced by this simple process can be evaluated in the site-specific uncertainty analysis.

Table 4.5.1-1.
Method for determining final soil ESL for radionuclides.

COPC	Plant ESL (pCi/g)	Invertebrate ESL (pCi/g)	Wildlife ESLs (pCi/g)	Final Soil ESL (pCi/g)[c]
X	Value[a]	Value	Value	Minimum of plant, invert, and wildlife ESLs
Y	No value[b]	No value	No value	No soil ESL, retain COPC as COPEC

a. Value = value available for that contaminant

b. No value = no value available for that contaminant

c. Note that the final soil radionuclide ESL will be adjusted to account for possible additive effects of radionuclides before applying this value to site-specific screening, the final radionuclide soil ESL will be divided by the number of radionuclide COPCs at a site and this value will be used as the final site-specific soil radionuclide ESL

The radiological dose to terrestrial biota is the sum of the dose from internally deposited radionuclides and the external dose from the same radionuclides in soil. The transport pathways considered for radionuclides in soil are identical to those assumed for non-radionuclides (Figure 4.4.1-1). Conservative assumptions about the size of the organism, its diet, the geometry of the contaminated source, and the location of the receptor relative to the contaminated source are used in the methods presented here for estimating internal and external doses. Thus the calculations are expected to provide high estimates of dose and should be used for screening purposes only. The calculations for estimating internal and external doses from radionuclides in soil are derived form those presented in Higley and Kuperman (1996, ER ID 62804) The basic model for calculating acceptable dose from soil for radionuclides is:

$$Dose_j = 0.1 = C_{organism,j} \cdot DCF_{int,j} + C_{soil,j} \cdot DCF_{ext,j} \qquad \textbf{Equation 4.5.1-1}$$

Where: $Dose_j$ is the total acceptable dose from radionuclide j (rad/day)
$C_{organism,j}$ is the internal concentration of radionuclide j (pCi/g of organism)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (rad/day per pCi/g)
$C_{soil,j}$ is the concentration of radionuclide j in soil (pCi/g)
$DCF_{ext,j}$ is the external dose conversion factor for radionuclide j (rad/day per pCi/g)

Internal dose results from exposure to radionuclides through plant uptake, incidental soil ingestion, and food web uptake (Figure 4.4.1-1). External dose is based on exposure to gamma emitting radionuclides from contaminated soil (Figure 4.4.1-1).

Radionuclide Concentrations in Biota

Plants and Invertebrates

The internal dose to plants is calculated by estimating the internal concentration or "body" burden and the internal DCF (as described below). The internal plant concentration is calculated as:

$$C_{plant,j} = C_{soil,j} \cdot TF_{plant,j} \qquad \textbf{Equation 4.5.1-2}$$

Where: $C_{plant,j}$ is the internal concentration of radionuclide j in plants (pCi/g)
$C_{soil,j}$ is the soil concentration of radionuclide j (pCi/g)
$TF_{plant,j}$ is the soil to plant transfer factor for radionuclide j (unitless)

The same equation is used to calculate dose to soil dwelling invertebrates, with a soil to invertebrate transfer factor (TF_{invert}) substituted in place of the soil to plant transfer factor. Thus, the internal concentration in invertebrates is:

$$C_{invert,j} = C_{soil,j} \cdot TF_{invert,j} \qquad \textbf{Equation 4.5.1-3}$$

Where: $C_{invert,j}$ is the internal concentration of radionuclide j in invertebrates (pCi/g)
$C_{soil,j}$ is the soil concentration of radionuclide j (pCi/g)
$TF_{invert,j}$ is the soil to invertebrate transfer factor for radionuclide j (unitless)

Values and references for transfer factors are presented in the ECORISK Database (LANL 1998, ER ID Package 186). When no literature values are available for soil to invertebrate transfer, a default value of 1 is used.

Wildlife

The internal dose to wildlife is calculated by multiplying the effective energy of a radionuclide by the body burden of that radionuclide in an organism. Body burden is a measure of the accumulation of a radionuclide in an organism through ingestion. The body burden calculation is presented in Equation 4.5.1-4.

$$C_{wildlife,j} = C_{soil,j} \cdot [I_{soil} + TF_{food,j} \cdot I_{food}] \cdot TF_{blood,j} \cdot R_{t,j} \qquad \textbf{Equation 4.5.1-4}$$

Where: C_{soil} is the concentration of radionuclide j in soil (pCi/g)
$C_{wildlife,j}$ is the body burden of radionuclide j in a wildlife species (pCi/g)
I_{soil} is the normalized daily soil ingestion rate (g of soil/g of body weight/day)
I_{food} is the normalized daily dietary ingestion rate (g of food [dry wt]/g of body weight/day)
$TF_{food,j}$ is the soil to food transfer factor for radionuclide j (unitless)
$TF_{blood,j}$ is the food to blood transfer factor (unitless)
$R_{t,j}$ is the retention time of radionuclide j in the organism (days)

Dietary and soil ingestion rates for each receptor are the presented in Table 4.3-5. Values and supporting references for all transfer factors used are provided in the ECORISK Database (LANL 1998, ER ID Package 186). The retention time, R_t, is an equilibrium model, which assumes the activity concentration of a radionuclide reaches steady state in an organism over time, dependent upon the rate of radiological

decay and metabolic elimination of the element from the organisms body. This value is calculated as (modified from Baker and Soldat 1992, ER ID 62801):

$$R_t = (1 - e^{-\lambda Tc}) / \lambda) \qquad \textbf{Equation 4.5.1-5}$$

Where: $\lambda = \lambda r + \lambda b$

$\lambda r = \ln(2) / Tr$, where *Tr* is the radiological half-life of the radionuclide (days)

$\lambda b = \ln(2) / Tb$, where *Tb* is the biological half-life of the radionuclide (days) (based on human biological half-lives)

Tc = exposure duration, or the average life-span of the receptor (days)

Values and references for all of the parameters used in calculating retention times for each radionuclide are presented in the ECORISK Database (LANL 1998, ER ID Package 186)

Internal Dose Conversion Factor

The radionuclides uranium, plutonium, americium, thorium, and radium have radioactive daughters. For screening purposes, the summation of average energies per disintegration for the decay chains of all radioactive daughters for any given isotope will be used. This method provides an overestimate of exposure, as the lifetime of many of the biota of interest is short compared to the time for the build-up of progeny. The energy deposition for radionuclides is usually given in the units MeV/disintegration. In order to calculate internal dose, it is necessary to convert MeV/disintegration to rad/day per pCi/g, as internal radioactivity is measured in pCi/g. A combined conversion factor of 5.11 • 10^{-5} (disintegrations • g • rad) /(MeV • pCi • day) is applied to convert MeV/disintegration to rad/day per pCi/g. This conversion factor is derived in Equation 4.5.1-6.

Equation 4.5.1-6

$$5.11 \cdot 10^{-5} \frac{disintegrations \cdot g \cdot rad}{MeV \cdot pCi \cdot day} = 1.6 \cdot 10^{-6} \frac{ergs}{MeV} \cdot 1 \frac{rad}{100 ergs / g} \cdot 1 \frac{disintegration}{27.03 pCi \cdot s} \bullet 8.64 \cdot 10^{4} \frac{s}{day}$$

The relative biological effectiveness of alpha particle emissions is about 20 times that of beta or gamma emissions, so the fraction of energy deposition due to alpha particles must be taken into account in calculation of internal dose (IAEA 1992, ER ID 62802). Thus, the internal dose conversion factor (DCF) to any organism from radionuclide j can be calculated:

$$DCF_{int,j} = CF_i \cdot (f_a \cdot 20 + [1 - f_a])E_j \qquad \textbf{Equation 4.5.1-7}$$

Where: CF_i is the conversion factor between energy per disintegration and rad/day
[value is 5.11e-5 disintegrations • g • rad) /(MeV • pCi • day)]

f_a is the fraction of disintegrations that are alpha particles

E_j is the sum of deposited energies for radionuclide j and its daughter products (units are MeV/disintegration)

External Dose to Biota

The external dose to biota is the dose an organism receives from being exposed to contaminated soil. This dose varies with several factors, including the size of the organism, the distance of the organism from the contaminated media, the geometry of the contamination within the contaminated media, and the type of radiological decay (Baker and Soldat 1992, ER ID 62801; Eckerman and Ryman 1993, ER ID 62798). Several simplifying assumptions can be made to make this problem more manageable. First, as suggested by the conceptual model diagram (Figure 4.4.1-1), only external exposure from gamma-emitting radionuclides is considered. The basis for eliminating alpha and beta decay from the external pathway is that only a small dose is received from external irradiation compared to internal dose for alpha and beta emitters (Higley and Kuperman 1996, ER ID 62804). To emphasize the protective nature of the screening levels, "worst case" assumptions can be made on the size of the organism, the geometry of the contaminated source, and the location of the receptor relative to the contaminated source. Dose coefficients developed for exposure to soil assume only 180^{o} exposure to the contaminated source, and thus are inappropriate for modeling exposure to organisms dwelling in soil. For soil invertebrates and burrowing mammals, external dose coefficients based upon immersion in water contaminated to an infinite depth will be used (Eckerman and Ryman 1993, ER ID 62798). This will provide a conservative estimate of external dose, as dose resulting from immersion in contaminated soil would be less than dose from water due to the higher density of soil. For terrestrial organisms living on or above the soil surface, dose coefficients for exposure to soil contaminated to an infinite depth will be used (Eckerman and Ryman 1993, ER ID 62798). As larger organisms receive a greater proportion of the external dose, the standard man is suggested as a default organism. This will conservatively represent exposure to all terrestrial receptors living on or above the soil surface. Thus, external DCF is modeled by the following equation:

Invertebrates and Burrowing Mammals:

$$DCF_{ext,j} = DC_{water,skin,j} \cdot CF_{e,w} \qquad \textbf{Equation 4.5.1-8(a)}$$

Terrestrial Receptors On or Above the Soil Surface:

$$DCF_{ext,j} = DC_{soil,skin,j} \cdot CF_{e,s} \qquad \textbf{Equation 4.5.1-8(b)}$$

Where: $DC_{water,skin,j}$ is the dose coefficient for skin exposed to water contaminated to an infinite depth with radionuclide j (from Eckerman and Ryman 1993, ER ID 62798)
$CF_{e,w}$ is the conversion factor between Sv/s per Bq/m^3 and rad/day per pCi/g for an organism immersed in water [value is 3.2 x 10^{11}; see Equation 4.5.1-9],
$DC_{skin,j}$ is the dose coefficient for skin exposed to soil contaminated to an infinite depth with radionuclide j (from Eckerman and Ryman 1993, ER ID 62798),
$CF_{e,w}$ is the conversion factor between Sv/s per Bq/m^3 for an organism on the soil surface [value is 5.11 x 10^{11}; see Equation 4.5.1-10].

$CF_{e,w}$ assumes a water density of 1.0 x 10^3 kg/m^3 and is derived in the following equation:

$$CF_{e,w} = 10^3 \frac{kg}{m^3} \cdot 10^3 \frac{g}{kg} \cdot 100 \frac{rad}{Sv} \cdot 1 \frac{Bq}{27.03 pCi} \cdot 86400 \frac{s}{d} \qquad \textbf{Equation 4.5.1-9}$$

$CF_{e,s}$ assumes a soil density of 1.6 x 10^3 kg/m^3 and is derived in the following equation:

$$CF_{e,s} = 1.6 \cdot 10^3 \frac{kg}{m^3} \cdot 10^3 \frac{g}{kg} \cdot 100 \frac{rad}{Sv} \cdot 1 \frac{Bq}{27.03 pCi} \cdot 86400 \frac{s}{d}$$ **Equation 4.5.1-10**

Calculations of ESLs for Soil

The soil ESL is defined as the soil concentration of a given radionuclide that yields a combined internal and external dose rate of 0.1 rad/day to any organism. For terrestrial plants this calculation is:

$$ESL = \frac{0.1}{TF_{plant,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 4.5.1-11**

Where: 0.1 is the dose limit (rad/day)
$TF_{plant,j}$ is the soil to plant transfer factor for radionuclide j (unitless)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (rad/day per pCi/g) (from Equation 4.5.1-8)
$DCF_{ext,j}$ is the external dose conversion factor for radionuclide j (rad/day per pCi/g) (from Equations 4.5.1-8[a] and 4.5.1-8[b])

For terrestrial invertebrate receptors, the ESL equation can be written:

$$ESL = \frac{0.1}{TF_{invert,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 4.5.1-12**

Where: 0.1 is the dose limit (rad/day)
$TF_{invert,j}$ is the soil to invertebrate transfer factor for radionuclide j (unitless)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (rad/day per pCi/g) (from Equation 4.5.1-7)
$DCF_{ext,j}$ is the external dose conversion factor for radionuclide j (rad/day per pCi/g) (from Equations 4.5.1-8[a] and 4.5.1-8[b])

For terrestrial herbivores, the ESL equation can be written:

$$ESL = \frac{0.1}{[I_{soil,i} + TF_{plant,j} \cdot I_{plant,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 4.5.1-13**

Where: 0.1 is the dose limit (rad/day)
$I_{soil,i}$ is the normalized daily soil ingestion rate for organism i (g of soil/g of body wt/day)
$I_{plant,i}$ is the normalized daily plant ingestion rate for organism i (g of dry plant/g of body wt/day)
$TF_{plant,j}$ is the soil to plant transfer factor for radionuclide j (unitless)
$TF_{blood,j}$ is the food to blood transfer factor for radionuclide j (unitless)
$R_{t,j}$ is the retention time for radionuclide j (days)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (rad/day per pCi/g) (from Equation 4.5.1-7)
$DCF_{ext,j}$ is the external dose conversion factor for radionuclide j (rad/day per pCi/g) (from Equations 4.5.1-8[a] and 4.5.1-8[b])

For terrestrial receptors with a 100% invertebrate diet, the ESL equation is written:

$$ESL = \frac{0.1}{[I_{soil,i} + TF_{invert,j} \cdot I_{invert,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 4.5.1-14**

Where: 0.1 is the dose limit (rad/day)
$I_{soil,i}$ is the normalized daily soil ingestion rate for organism i (g of soil/g of body wt/day)
$I_{invert,i}$ is the normalized daily invertebrate ingestion rate for organism i (g of dry food/g of body wt/day)
$TF_{invert,j}$ is the soil to invertebrate transfer factor for radionuclide j (unitless)
$TF_{blood,j}$ is the food to blood transfer factor for radionuclide j (unitless)
$R_{t,j}$ is the retention time for radionuclide j (days)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (rad/day per pCi/g) (from Equation 4.5.1-7)
$DCF_{ext,j}$ is the external dose conversion factor for radionuclide j (rad/day per pCi/g) (from Equations 4.5.1-8[a] and 4.5.1-8[b])

For terrestrial omnivores feeding upon both plants and invertebrates, the following ESL equation is used:

Equation 4.5.1-15

$$ESL = \frac{0.1}{[I_{soil,i} + TF_{plant,j} \cdot I_{plant,i} + TF_{invert,j} \cdot I_{invert,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$

Where: 0.1 is the dose limit (rad/day)
$I_{soil,i}$ is the normalized daily soil ingestion rate for organism i (g of soil/g of body wt/day)
$I_{plant,j}$ is the normalized daily plant ingestion rate for organism i (g of dry plant/g of body wt/day)
$I_{invert,i}$ is the normalized daily invertebrate ingestion rate for organism i (g of dry food/g of body wt/day)
$TF_{plant,j}$ is the soil to plant transfer factor for radionuclide j (unitless)
$TF_{invert,j}$ is the soil to invertebrate transfer factor for radionuclide j (unitless)
$TF_{blood,j}$ is the food to blood transfer factor for radionuclide j (unitless)
$R_{t,j}$ is the retention time for radionuclide j (days)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (from Equation 4.5.1-7)
$DCF_{ext,j}$ is the external dose conversion factor for radionuclide j (from Equations 4.5.1-8[a] and 4.5.1-8[b])

For terrestrial carnivores, the ESL is calculated as:

$$ESL = \frac{0.1}{[I_{soil,i} + TF_{flesh,j} \cdot I_{flesh,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 4.5.1-16**

Where: 0.1 is the dose limit (rad/day)
$I_{soil,i}$ is the normalized daily soil ingestion rate for organism i (g of soil/g of body wt/day)
$I_{flesh,i}$ is the normalized daily flesh ingestion rate for organism i (g of dry food/g of body wt/day)
$TF_{flesh,j}$ is the food to flesh transfer factor for radionuclide j (unitless)
$TF_{blood,j}$ is the food to blood transfer factor for radionuclide j (unitless)
$R_{t,j}$ is the retention time for radionuclide j (days)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (from Equation 4.5.1-7)
$DCF_{ext,j}$ is the external dose conversion factor for radionuclide j (from Equations 4.5.1-8[a] and 4.5.1-8[b])

Table 4.5.1-2 summarizes the variables used to calculate soil ESLs for radionuclides. There are many common variables between the non-radionuclide and radionuclide ESL calculations (compare Table 4.4.1-1 with Table 4.5.1-2). Thus, much of the discussion concerning uncertainty in the non-radionuclide ESLs is directly relevant to the radionuclide ESLs. Three variables, the retention time, blood transfer factor, and the dose conversion factors are unique to radionuclides. The retention time and blood transfer factors will vary between species and are based on laboratory experimental data. Thus, there will be some uncertainty in these values. However, the retention time typically does not impact the ESL except for radionuclides with short biological clearance times (like tritium). The dose conversion factors are based on hard physical information for the radionuclide and typically have less uncertainty, especially in the screening context where worst-case assumptions are made.

Table 4.5.1-2.

Summary of variables used in soil ESL calculations for radionuclides.

Variable	Source
I_{soil}	Body weight normalized soil ingestion rate for wildlife receptors (food intake x fraction of soil in diet from Table 4.3-5).
I_{plant}	Body weight normalized plant ingestion rate for wildlife receptors (food intake x fraction of plants in diet from Table 4.3-5).
I_{invert}	Body weight normalized invertebrate ingestion rate for wildlife receptors (food intake x fraction of invertebrates in diet from Table 4.3-5).
I_{flesh}	Body weight normalized flesh ingestion rate for wildlife receptors (food intake x fraction of flesh in diet from Table 4.3-5).
R_t	The retention time of a radionuclide in an organism. This is a COPC-specific value based upon both the radiological decay constant and the biological removal rate constant for a given radionuclide. See Equation 4.5.1-5 for calculation of this variable.
TF_{blood}	The transfer factor from food to blood is a COPC-specific value that is derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database (LANL 1998, ER ID Package 186).
TF_{plant}	The transfer from soil to plants is a COPC-specific value that is derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database (LANL 1998, ER ID Package 186).
TF_{invert}	The transfer from soil to invertebrates is a COPC-specific value that is derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database (LANL 1998, ER ID Package 186).
TF_{flesh}	The transfer from soil to flesh is a COPC-specific value that is derived from two other factors. The first factor is a feed to muscle transfer factor (TF_{beef}) derived from studies of beef cattle. The second factor is the maximum of either the TF_{plant} or TF_{invert}, which is used model the prey with the most contaminated diet. It is assumed that the prey being consumed are mammalian. Thus, TF_{flesh} = maximum(TF_{plant}, TF_{invert})*TF_{beef}. Values for specific COPCs are provided in the ECORISK Database (LANL 1998, ER ID Package 186)
f_a	The fraction of energy deposition in an organism due to alpha particle absorption.
DCF_{int}	The internal dose conversion factor for a specific radionuclide. This factor considers the conversion of units of deposited energy from MeV/disintegration to rad/day per pCi/g and accounts for the increased biological effectiveness of alpha particle deposition over beta or gamma deposition (see Equation 4.5.1-7).
DCF_{ext}	The external dose conversion factor for a specific radionuclide. This factor applies only to gamma emitters and is media and COPC specific. It also contains the same unit conversion factor as DCF_{int}.

4.5.2 Sediment ESLs

Discussion on the operational definition of sediment was provided in Section 4.4.2, Sediment ESLs. Radionuclide sediment ESLs are based on exposure of contaminated sediment to aquatic and terrestrial

receptors. The final radionuclide sediment ESL is the lowest receptor-specific ESL among the four aquatic and two terrestrial receptors (Table 4.5.2-1). ESLs for the other five receptors should be used as appropriate in the site-specific uncertainty analysis discussion (see Section 4.6, Screening Evaluation/Uncertainty Analysis). ESLs are developed to account for dose from a single radionuclide. Because radiological dose is additive, the radionuclide ESLs must be adjusted to account for doses from multiple radionuclides. Per instructions from NMED, a simple adjustment factor will be used to account for dose from multiple radionuclides. The final sediment ESL will be divided by the number of radionuclide COPCs at a site before using the final sediment ESLs in a site specific screening. Thus, if there are five radionuclides COPCs at a site the ESL will be divided by 5. This approach is both simple and conservative, and any bias introduced by this simple process can be evaluated in the site-specific uncertainty analysis.

Table 4.5.2-1.
Method for determining final sediment ESL for radionuclides.

COPC	Plant ESL (pCi/g)	Invertebrate ESLs (pCi/g)	Fish ESL (pCi/g)	Wildlife ESLs (pCi/g)	Final Sediment ESL (pCi/g)[c]
X	Value[a]	Value	Value	Value	Minimum of plant, invert, fish and wildlife ESLs
Y	No value[b]	No value	No value	No value	No sediment ESL, retain COPC as COPEC

a. Value = value available for that contaminant

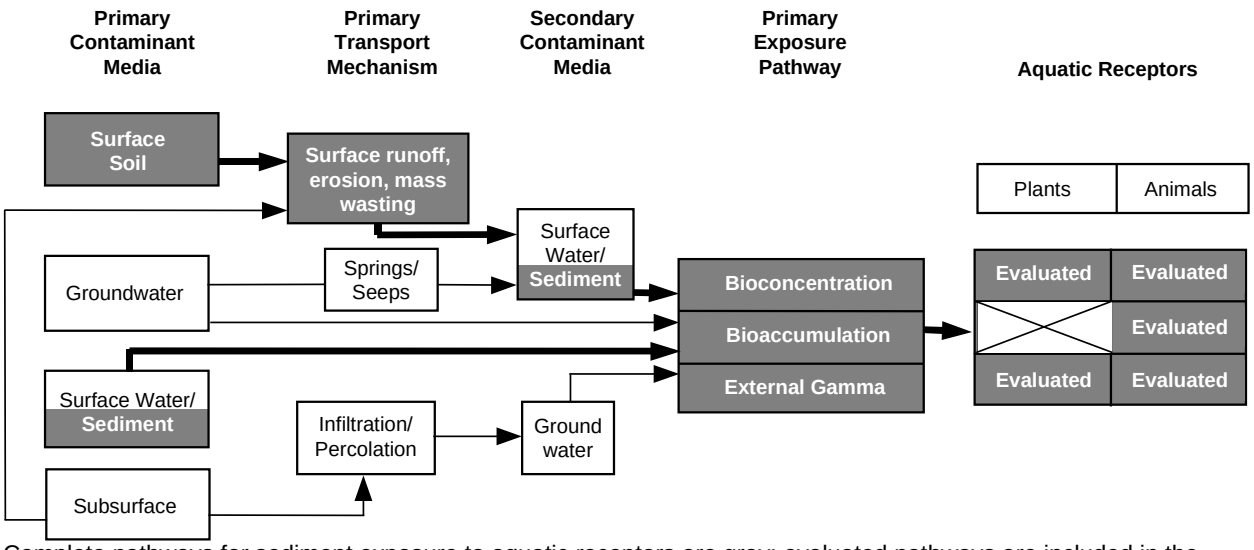
b. No value = no value available for that contaminant

c. Note that the final sediment radionuclide ESL will be adjusted to account for possible additive effects of radionuclides before applying this value to site-specific screening, the final radionuclide soil ESL will be divided by the number of radionuclide COPCs at a site and this value will be used as the final site-specific sediment radionuclide ESL

Calculation of ESLs for aquatic organisms exposed to sediment is based on the models presented by Baker and Soldat (1992, ER ID 62801). The radiological dose to aquatic organisms is the sum of the dose from internally deposited radionuclides and the external dose from the same radionuclides in sediment. Sediment based thresholds that can be used as screening values do not exist for radionuclides, so algae, daphnids, and snails and fish have been chosen as assessment endpoint surrogates for organisms living in aquatic environments at the Laboratory. Transport pathways from sediment to aquatic organisms are presented in Figure 4.5.2-1. In addition, to address bioaccumulation and some biomagnification concerns, bats and swallows have been chosen as potential higher trophic level terrestrial receptors that feed primarily upon insects emerging from sediment in aquatic environments. ESLs are calculated for these receptors assuming they are feeding 100% upon aquatic invertebrates. The pathways for terrestrial receptor exposure to sediment are the same as presented in Figure 4.4.2-2. The basic model for calculating acceptable dose from sediment for radionuclides is:

$$Dose_j = 0.1 = C_{organism,j} \cdot DCF_{int,j} + C_{sediment,j} \cdot DCF_{ext,j} \qquad \textbf{Equation 4.5.2-1}$$

Where: $Dose_j$ is the total acceptable dose from radionuclide j (rad/day)
$C_{organism,j}$ is the internal concentration of radionuclide j (pCi/g of organism)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (rad/day per pCi/g)
$C_{sediment,j}$ is the concentration of radionuclide j in sediment (pCi/g)
$DCF_{ext,j}$ is the external dose conversion factor for radionuclide j (rad/day per pCi/g)



Complete pathways for sediment exposure to aquatic receptors are gray; evaluated pathways are included in the sediment ESL calculations for aquatic receptors.

Figure 4.5.2-1. Aquatic conceptual model for sediment pathways.

Radionuclide Concentrations in Biota

For organisms living in or on sediment (algae, daphnid, snail, and bottom-feeding fish), internal concentration of any radionuclide is modeled by applying a bioconcentration factor (Baker and Soldat 1992, ER ID 62801):

$$C_{organism,\,j} = C_{sediment\,,j} \cdot BCF_{organism\,,j} \qquad \textbf{Equation 4.5.2-2}$$

Where: $C_{organism,j}$ is the internal concentration of radionuclide j in organism i (pCi/g of organism)
$C_{sediment,j}$ is the concentration of radionuclide j in sediment (pCi/g)
$BCF_{organism,j}$ is the bioconcentration factor for radionuclide j in the organism (g/g)

Assuming the bat and swallow are eating only flying insects that have emerged from aquatic systems (an extremely conservative assumption), the body burden for these organisms is calculated:

$$C_{organism,\,j} = C_{water\,,j} \cdot BCF_{invert\,,j} \cdot I_{food,i} \cdot TF_{blood\,,j} \cdot R_{t,\,j} \qquad \textbf{Equation 4.5.2-3}$$

Where: $C_{sediment,j}$ is the concentration of radionuclide j in sediment (pCi/g)
$BCF_{invert,j}$ is the sediment to invertebrate bioconcentration factor for radionuclide j (g/g)
$I_{food,i}$ is the normalized daily dietary ingestion rate of organism i (g of food [dry wt]/g of body weight /day)
$TF_{blood,j}$ is the food to blood transfer factor for radionuclide j (unitless)
$R_{t,j}$ is the retention time of radionuclide j in the organism (days) (see Equation 4.5.1-5)

Values and references for the transfer factors and bioconcentration factors are provided in the ECORISK Database (LANL 1998, ER ID Package 186).

Dose Conversion Factors

For aquatic organisms, internal dose conversion factors are identical to those used for terrestrial receptors (see Equation 4.5.1-7). For organisms that reside in or on the sediment (algae, snail, fish), external dose is estimated as for terrestrial receptors living in or on soil (see Equations 4.5.1-8[a] and 4.5.1-8[b]). As with terrestrial receptors, external dose is deemed significant only for gamma emitters.

Internal dose to terrestrial receptors from sediment is assumed to come entirely from uptake from the food chain. Because these receptors would have limited contact with sediments, it is assumed that external dose is insignificant and all dose received is internal.

ESL Calculations

The sediment ESL is defined as the sediment concentration of a given radionuclide that yields a combined internal and external dose rate of 0.1 rad/day to a particular organism. For organisms that spend at least part of their lives in close association with sediment, the sediment ESL calculation is:

$$ESL = \frac{0.1}{BCF_{i,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 4.5.2-4**

Where: 0.1 is the dose limit (rad/day)
$BCF_{i,j}$ is the bioconcentration factor for organism i and radionuclide j (g/g)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (rad/day per pCi/g)
$DCF_{ext,j}$ is the external dose conversion factor for radionuclide j (rad/day per pCi/g)

For the terrestrial receptors feeding primarily on emergent aquatic invertebrates, with little contact with the sediment itself, the ESL calculation is written:

$$ESL = \frac{0.1}{I_{food,i} \cdot BCF_{invert,j} \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j}}$$ **Equation 4.5.2-5**

Where: 0.1 is the dose limit (rad/day)
$I_{food,i}$ is the normalized daily dietary ingestion rate for organism i (g of food [dry wt]/g of body weight /day)
$BCF_{invert,j}$ is the invertebrate bioconcentration factor for radionuclide j (g/g)
$TF_{blood,j}$ is the food to blood transfer factor for radionuclide j (unitless)
$R_{t,j}$ is the retention time for radionuclide j (days)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (rad/day per pCi/g)

Table 4.5.2-2 summarizes the input variables for the radionuclide sediment ESL models. Uncertainties associated with radionuclide sediment ESLs fall into two main categories. The first group of uncertainties is associated with contaminants, including bioconcentration (ratio between the concentration in an organism and sediment) and dose conversion factors. The second group of uncertainties relates to receptors, including feeding rates and diets. The uncertainties associated with the variables in Table 4.5.2-2 are similar to those identified for radionuclides in soil.

Table 4.5.2-2.

Summary of variables used in the radionuclide sediment ESL models.

Variable	Source
I_{food}	Body weight normalized food ingestion rate for aerial insectivores receptors (food intake from Table 4.3-5).
R_t	The retention time of a radionuclide in an organism. This is a COPC-specific value based upon both the radiological decay constant and the biological removal rate constant for a given radionuclide. See Equation 4.5.1-5 for calculation of this variable
TF_{blood}	The transfer factor from food to blood is a COPC-specific value that is derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database (LANL 1998, ER ID Package 186)
BCF_{invert}	The transfer from sediment to invertebrates is a COPC-specific value that is derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database (LANL 1998, ER ID Package 186)
$BCF_{organism}$	The transfer from sediment to organisms is a COPC-specific value that is derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database (LANL 1998, ER ID Package 186)
DCF_{int}	The internal dose conversion factor for a specific radionuclide. This factor considers the conversion of units of deposited energy from MeV/disintegration to rad/day per pCi/g and accounts for the increased biological effectiveness of alpha particle deposition over beta or gamma deposition (see Equation 4.5.1-7).
DCF_{ext}	The external dose conversion factor for a specific radionuclide. This factor applies only to gamma emitters and is media and COPC specific. It also contains the same unit conversion factor as DCF_{int}.

4.5.3 Water ESLs

Discussion on the operational definition of water was provided in Section 4.4.3, Water ESLs. Radionuclide water ESLs are based on exposure of contaminated surface water to aquatic and terrestrial receptors. The final radionuclide water ESL is the lowest receptor-specific ESL among the four aquatic and ten terrestrial receptors (Table 4.5.3-1). ESLs for the other thirteen receptors should be used as appropriate in the site-specific uncertainty analysis discussion (see Section 4.6, Screening Evaluation/Uncertainty Analysis). ESLs are developed to account for dose from a single radionuclide. Because radiological dose is additive, the radionuclide ESLs must be adjusted to account for doses from multiple radionuclides. Per instructions from NMED, a simple adjustment factor will be used to account for dose from multiple radionuclides. The final water ESL will be divided by the number of radionuclide COPCs at a site before using the final water ESLs in a site specific screening. Thus, if there are five radionuclides COPCs at a site the ESL will be divided by 5. This approach is both simple and conservative, and any bias introduced by this simple process can be evaluated in the site-specific uncertainty analysis.

Table 4.5.3-1.

Method for determining final water ESL for radionuclides.

COPC	Plant ESL (pCi/L)	Invertebrate ESLs (pCi/L)	Fish ESL (pCi/L)	Wildlife ESLs (pCi/L)	Final Water[c] ESL (pCi/L)
X	Value[a]	Value	Value	Value	Minimum of plant, invert, and wildlife ESLs
Y	No value[b]	No value	No value	No value	No water ESL, retain COPC as COPEC

a. Value = value available for that contaminant

b. No value = no value available for that contaminant

c. Note that the final sediment radionuclide ESL will be adjusted to account for possible additive effects of radionuclides before applying this value to site-specific screening, the final radionuclide soil ESL will be divided by the number of radionuclide COPCs at a site and this value will be used as the final site-specific sediment radionuclide ESL

Calculation of ESLs for aquatic organisms is based on the models presented by Baker and Soldat (1992, ER ID 62801). The radiological dose to aquatic organisms is the sum of the dose from internally deposited radionuclides and the external dose from the same radionuclides in water. Media based screening values for radionuclides do not exist, so algae, daphnids, and snails and fish have been chosen as assessment endpoint surrogates for organisms living in aquatic environments at the Laboratory. Transport pathways to aquatic organisms are presented in Figure 4.5.3-1. The only water exposure pathway considered for terrestrial receptors is ingestion of drinking water (Figure 4.4.3-2). The basic model for calculating acceptable dose from water for radionuclides is:

$$Dose_j = 0.1 = C_{organism,j} \cdot DCF_{int,j} + C_{water,j} \cdot DCF_{ext,j} \qquad \textbf{Equation 4.5.3-1}$$

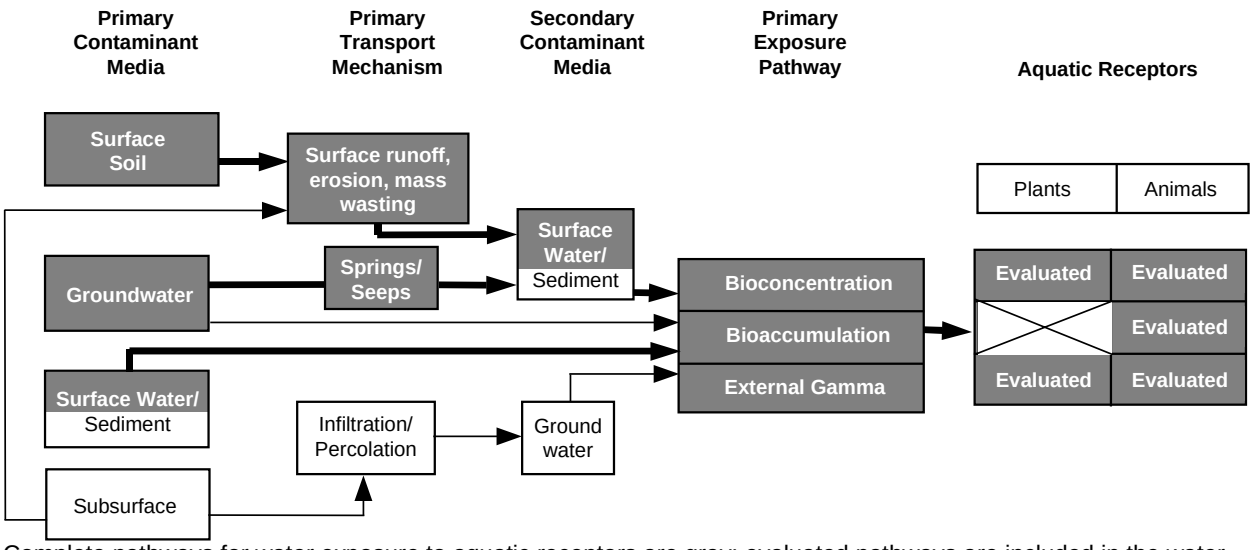
Where: $Dose_j$ is the total acceptable dose from radionuclide j (rad/day)
$C_{organism,j}$ is the internal concentration of radionuclide j (pCi/g of organism)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (rad/day per pCi/g)
$C_{water,j}$ is the concentration of radionuclide j in water (pCi/ml)
$DCF_{ext,j}$ is the external dose conversion factor for radionuclide j (rad/day per pCi/ml)

Radionuclide Concentrations in Biota

For organisms immersed in water (algae, daphnid, snail, and fish), internal concentration of any radionuclide is modeled by applying a simple bioconcentration factor (Baker and Soldat 1992, ER ID 62801) :

$$C_{organism,j} = C_{water,j} \cdot BCF_{organism,j} \qquad \textbf{Equation 4.5.3-2}$$

Where: $C_{water,j}$ is the concentration of radionuclide j in water (pCi/ml)
$BCF_{organism,j}$ is the bioconcentration factor for radionuclide j in the organism (ml/g)



Complete pathways for water exposure to aquatic receptors are gray; evaluated pathways are included in the water ESL calculations.

Figure 4.5.3-1. Aquatic conceptual model for water pathways.

For wildlife, it is assumed that the major exposure pathway to radionuclides in water is through ingestion of contaminated water. The body burden from drinking water containing radionuclides is calculated:

$$C_{organism,\,j} = C_{water,\,j} \cdot I_{water} \cdot TF_{blood,\,j} \cdot R_{t,\,j} \qquad \textbf{Equation 4.5.3-3}$$

Where: $C_{water,j}$ is the concentration of radionuclide j in water (pCi/ml)
I_{water} is the normalized daily water ingestion rate (ml of water/g of body weight /day)
$TF_{blood,j}$ is the water to blood transfer factor for radionuclide j (unitless) (assumed to be equal to the food to blood transfer factor)
$R_{t,j}$ is the retention time of radionuclide j in the organism (days)(see Equation 4.5.1-5)

Values and references for the transfer factors and bioconcentration factors are provided in the ECORISK Database (LANL 1998, ER ID Package 186).

Dose Conversion Factors

For aquatic organisms, internal dose conversion factors are identical to those used for terrestrial receptors (see Equation 4.5.1-7). For organisms which are immersed in water (algae, daphnid, snail, fish), the external dose coefficients of Eckerman and Ryman (1993, ER ID 62798) are used to estimate external dose. Coefficients used are for skin immersed in water contaminated to an infinite depth. A conversion factor of 3.2×10^{-11} is used to convert the dose coefficients from Sv/s per Bq/m^3 to rad/day per pCi/g (see Equation 4.5.1-9).

Internal dose to terrestrial receptors from water is assumed to come entirely from water ingestion. Because of the limited amount of perennial surface water at the Laboratory, and the conservative model used to calculate internal dose to terrestrial receptors, external dose is assumed to be insignificant, and all dose received is assumed to be internal.

Water ESL Calculations

The water ESL is defined as the water concentration (pCi/L) of a given radionuclide that yields a combined internal and external dose rate of 0.1 rad/day to a particular organism. For aquatic organisms that spend at least part of their lives immersed in water, the ESL calculation is:

$$ESL = \frac{0.1}{(BCF_{i,j} \cdot DCF_{int,j} + DCF_{ext,j})/1000}$$ **Equation 4.5.3-4**

Where: 0.1 is the acceptable dose limit (rad/day)
$BCF_{i,j}$ is the bioconcentration factor for organism i and radionuclide j (ml/g)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (rad/day per pCi/g)
$DCF_{ext,j}$ is the external dose conversion factor for radionuclide j (rad/day per pCi/ml)
1000 is the number of ml/L

For the terrestrial receptors drinking contaminated water, the ESL calculation is written:

$$ESL = \frac{0.1}{(I_{water,i} \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j})/1000}$$ **Equation 4.5.3-5**

Where: 0.1 is the dose limit (rad/day)
I_{water} is the normalized daily water ingestion rate (ml of water/g of body weight per day)
$TF_{blood,j}$ is the water to blood transfer factor for radionuclide j (unitless)(assumed to be equal to the food to blood transfer factor)
$R_{t,j}$ is the retention time for radionuclide j (days)
$DCF_{int,j}$ is the internal dose conversion factor for radionuclide j (rad/day per pCi/g)
1000 is the number of ml/L

Table 4.5.3-2 summarizes the input variables for the radionuclide water ESL models and indicates the general sources used for these variables. Uncertainties associated with radionuclide water ESLs fall into two main categories. The first group of uncertainties is associated with contaminants, including bioconcentration (ratio between the concentration in an organism and sediment) and dose conversion factors. The second group of uncertainties relates to receptors, which includes the water ingestion rate. The uncertainties associated with the variables in Table 4.5.3-2 are similar to those identified for radionuclides in soil.

4.5.4 Multimedia Screening Calculations

Multimedia calculations for radionuclides use the same principles and equations as for non-radionuclides. Because all radionuclides are evaluated in terms of radiological dose, it is appropriate to combine exposure across all radionuclides and contaminated media. Table 4.5.4-1 provides the list of ecological receptors considered in the multiple radionuclide and multimedia assessment.

Table 4.5.3-2.
Summary of variables used in the radionuclide water ESL models.

Variable	Source
I_{water}	Body weight normalized water ingestion rate for terrestrial ecological receptors (water intake from Table 4.3-5).
R_t	The retention time of a radionuclide in an organism. This is a COPC-specific value based upon both the radiological decay constant and the biological removal rate constant for a given radionuclide. See Equation 4.5.1-5 for calculation of this variable
TF_{blood}	The transfer factor from food to blood is a COPC-specific value that is derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database (LANL 1998, ER ID Package 186)
$BCF_{organism}$	The transfer from water to organisms is a COPC-specific value that is derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database (LANL 1998, ER ID Package 186)
DCF_{int}	The internal dose conversion factor for a specific radionuclide. This factor considers the conversion of units of deposited energy from MeV/disintegration to rad/day per pCi/g and accounts for the increased biological effectiveness of alpha particle deposition over beta or gamma deposition (see Equation 4.5.1-7).
DCF_{ext}	The external dose conversion factor for a specific radionuclide. This factor applies only to gamma emitters and is media and COPC specific. It also contains the same unit conversion factor as DCF_{int}.

Table 4.5.4-1.
List of wildlife receptors and applicable media for multiple contaminants and multimedia exposure to radionuclides.

Receptor	Contaminated Media Evaluated for Receptor		
	Soil	Sediment	Water
Robin	Yes	No	Yes
Kestrel	Yes	No	Yes
Deer mouse	Yes	No	Yes
Desert cottontail	Yes	No	Yes
Shrew	Yes	No	Yes
Red fox	Yes	No	Yes
Bat	No	Yes	Yes
Swallow	No	Yes	Yes
Algae	No	Yes	Yes
Daphnids	No	Yes	Yes
Snails	No	Yes	Yes
Fish	No	Yes	Yes

4.6 Screening Evaluation/Uncertainty Analysis

Much of the uncertainty in the screening assessment is addressed by the application of simple and protective exposure and toxicity assessments. However, the net result is likely to greatly overestimate exposure to ecological receptors from contaminated media. Thus, more accurate estimates of exposure

should be evaluated by considering factors like area use factors and bioavailability of COPECs (Pastorok et al. 1996, ER ID 62784). Bioavailability is often a key parameter in the evaluation of exposure to wildlife, and mechanistic bioconcentration or bioaccumulation models should be evaluated for their applicability (Jager 1998, ER ID 62736). Another key uncertainty is the availability of toxicity information for receptor groups (e.g., birds, mammals, plants, and invertebrates). Absence of toxicity information greatly reduces the meaning of a screening assessment, and the uncertainty analysis should determine the impact of missing or incomplete toxicity information on the identification of COPECs.

Therefore, the uncertainty analysis should focus, at a minimum, on the following key sources of uncertainty:

- Exposure-related parameters
  - Likelihood that the screening pathways are complete
  - Extent to which pathways identified in site-specific scoping are addressed by the screening calculations (e.g., foliar uptake by plants, inhalation, shallow groundwater, etc.)
  - Use of conservative values to estimate the receptors exposure to contaminated media (e.g., use of the maximum food intake for a species)
  - Availability of information for exposure media
  - Findings of data review (precision and bias of sample results for environmental media samples)
  - Possible bias introduced by use of the maximum site concentration to assess exposure
  - Application of area use factors and representative statistics (e.g., 95% UCL for the mean) to assess exposure
  - Environmental fate and transport of contaminants (including uncertainties associated with the assessment of persistent bioaccumulation and/or magnification)
- Toxicity-related parameters
  - Availability and quality of toxicity effects information
  - Possibility of cumulative effects and the method used to account for combined effects of multiple COPECs
  - Possibility of biomagnification concerns at higher tropic levels (e.g., top level carnivores)
  - Additivity of effects assumed by the HI calculation
  - Chemical form likely to be present in the environment as compared to form used in toxicity studies, which is important factor affecting the bioavailability of the COPEC
  - Possibility of contaminant interactions
  - Metabolic fate of COPEC

It is important to identify the implications for decision making that the uncertainty introduces into risk characterization. Do the uncertainties lead to a significant bias in risk estimates, or do uncertainties lead to a less precise estimate of risk? What data could be collected to cost-effectively reduce uncertainty?

What part of the uncertainty is linked to variation in the dynamical nature of contaminant releases and natural variation in biological populations?

4.7 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 the communication is to provide the risk manager with sufficient information to support a risk management decision with respect to ecological concerns. It may also be appropriate to make a recommendation. It is the responsibility of the risk manager to determine if sufficient information is provided to identify a risk management strategy (in terms of ecological concerns) or if more information is needed to better inform the risk management decision.

Some of the recommendations and risk management strategies that could emanate from the screening assessment include the following.

1. There is adequate information to conclude that the ecological risks are negligible and NFA for ecological risk is recommended.

2. There is adequate information to recommend immediate action, such as stabilization, run-on and run-off controls, and other interim actions (this strategy should be identified during the scoping phase of the screening assessment).

3. There are sufficient lines of evidence to document potential or actual adverse ecological effects. Thus, remediation to approved risk-based levels or background may be recommended (e.g., strategies might include voluntary corrective action (VCA), voluntary corrective measure (VCM), corrective measures study/corrective measure implementation CMS/CMI).

4. Ecological risks are not negligible, but there is not sufficient information to suggest that adverse ecological effects are occurring. A recommendation and risk management strategy is to move to the next level of ecological risk assessment to properly evaluate the potential for adverse ecological impacts. This next level of investigation will use the results of the screening assessment to focus the investigation. The approach used in the next level of investigation can vary from simply collecting more and/or better site-specific information to reduce uncertainties in the screening assessment, to conducting a baseline risk assessment.

5. There is not adequate information to make a risk management decision. A recommendation is to identify data needs, based on the results of the screening, and to develop a plan to collect additional data.

Note that Item 2 can occur concomitantly with Items 3, 4, and 5.

5.0 REFERENCES

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Appendix A

Ecological Scoping Checklist

A-1.0 PART A—SCOPING MEETING DOCUMENTATION

Site ID	
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.	
List of Primary Impacted Media **(Indicate all that apply.)**	**Surface soil –** **Surface water/sediment –** **Subsurface –** **Groundwater –** **Other, explain –**
FIMAD vegetation class based on Arcview vegetation coverage **(Indicate all that apply.)**	**Water** – **Bare Ground/Unvegetated** – **Spruce/fir/aspen/mixed conifer** – **Ponderosa pine –** **Piñon juniper/juniper savannah** – **Grassland/shrubland** – **Developed** –
Is T&E Habitat Present? **If applicable, list species known or suspected to use the site for breeding or foraging.**	
Provide list, of Neighboring/ Contiguous/ Upgradient sites, 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.)**	
Surface Water Erosion Potential Information **Summarize information from SOP 2.01, including the run-off subscore (maximum of 46); terminal point of surface water transport; slope; and surface water runon sources.**	
Other Scoping Meeting Notes	

A-2.0 PART B—SITE VISIT DOCUMENTATION

Site ID	
Date of Site Visit	
Site Visit Conducted by	

Receptor Information:

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)** =
Field notes on the FIMAD vegetation class to assist in ground-truthing the Arcview information	
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.	
Are ecological receptors present at the site? **(yes/no/uncertain)** **Describe the general types of receptors present at the site (terrestrial and aquatic), and make notes on the quality of habitat present at the site.**	

Contaminant Transport Information:

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

Ecological Effects Information:

Physical Disturbance **(Provide list of major types of disturbances, including erosion and construction activities, review historical aerial photos where appropriate.)**	
Are there obvious ecological effects? **(yes/no/uncertain)** **Provide explanation and apparent cause (e.g., contamination, physical disturbance, other).**	
Interim action needed to limit apparent ecological effects? **(yes/no/uncertain)** **Provide explanation and recommendations to mitigate apparent exposure pathways to project lead for IA SMDP.**	

No Exposure/Transport Pathways:

If there are no complete exposure pathways to ecological receptors onsite and no transport pathways to offsite 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:

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.)**	
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 aggregated to characterize potential ecological risk.)**	

Additional Field Notes:

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

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 (use SOP 2.01 run-off score and terminal point of surface water runoff to help answer this question)?

- **If the SOP 2.01 run-off score* for each PRS included in the site is equal to zero, this suggests that erosion at the site is not a transport pathway. (* note that the runoff score is not the entire erosion potential score, rather it is a subtotal of this score with a maximum value of 46 points).**

- **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 erosion or mass wasting events be a potential release mechanism for contaminants from subsurface materials or perched aquifers to the surface?

- **This question is only applicable to release sites located on or near the mesa edge.**
- **Consider the erodability of surficial material and the geologic processes of canyon/mesa edges.**

Answer (likely/unlikely/uncertain):

Provide explanation:

Question G:

Could airborne contaminants interact with receptors through respiration of vapors?

- **Contaminants must be present as volatiles in the air.**
- **Consider the importance of inhalation of vapors for burrowing animals.**
- **Foliar uptake of organic vapors is typically not a significant exposure pathway**.

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.**

- **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 surficial soils?

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

- **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 surficial 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 surficial 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.**

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 surficial 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:

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 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 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 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:

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 organisms?

- **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 organisms?

- **Lipophillic 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?

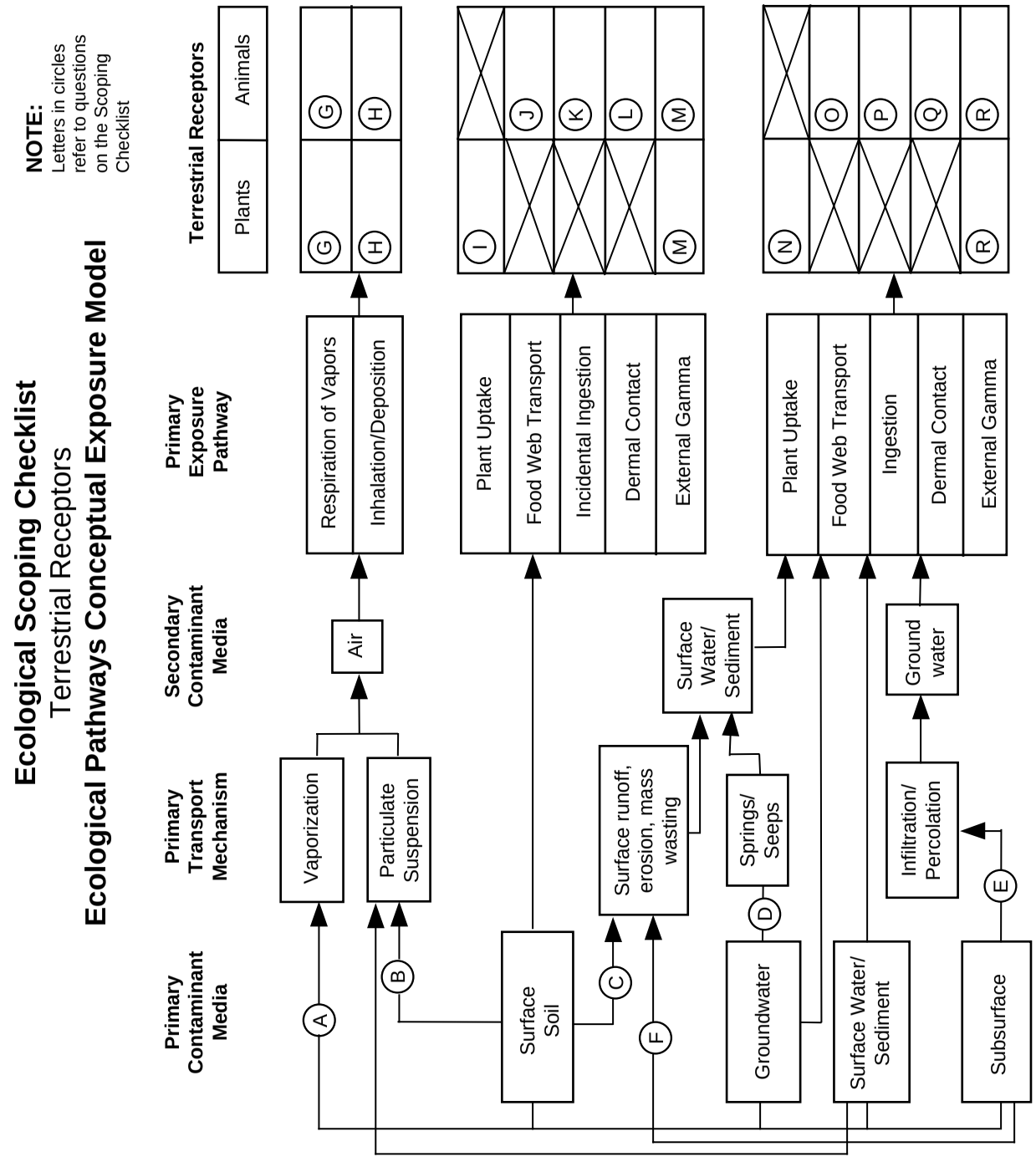
- **External irradiation effects are most relevant for gamma emitting radionuclides.**
- **The water column acts to absorb radiation, thus external irradiation is typically more important for sediment dwelling organisms.**

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

Aquatic Plants:

Aquatic Animals:

Provide explanation:


Ecological Scoping Checklist
Terrestrial Receptors
Ecological Pathways Conceptual Exposure Model
NOTE:
Letters in circles refer to questions on the Scoping Checklist
Primary Contaminant Media
Primary Transport Mechanism
Secondary Contaminant Media
Primary Exposure Pathway
Terrestrial Receptors
Plants
Animals
Vaporization
Particulate Suspension
Air
Respiration of Vapors
Inhalation/Deposition
Surface Soil
Plant Uptake
Food Web Transport
Incidental Ingestion
Dermal Contact
External Gamma
Surface runoff, erosion, mass wasting
Surface Water/ Sediment
Springs/ Seeps
Groundwater
Plant Uptake
Food Web Transport
Ingestion
Dermal Contact
External Gamma
Infiltration/ Percolation
Ground water
Surface Water/ Sediment
Subsurface
A
B
C
D
E
F
G
H
I
J
K
L
M
N
O
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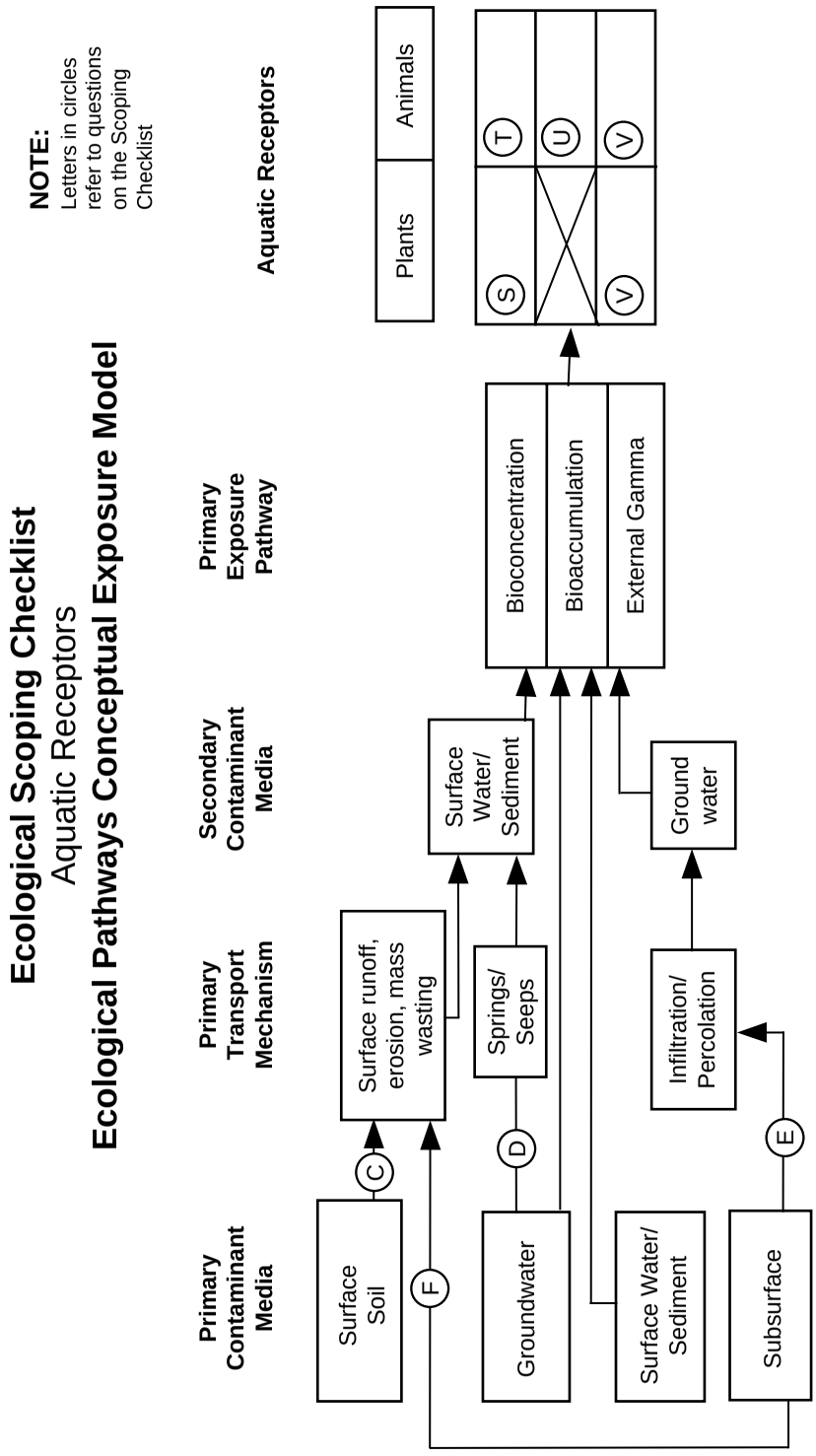
Ecological Scoping Checklist
Aquatic Receptors
Ecological Pathways Conceptual Exposure Model
NOTE:
Letters in circles refer to questions on the Scoping Checklist
Primary Contaminant Media
Primary Transport Mechanism
Secondary Contaminant Media
Primary Exposure Pathway
Aquatic Receptors
Surface Soil
Groundwater
Surface Water/ Sediment
Subsurface
C
F
D
E
Surface runoff, erosion, mass wasting
Springs/ Seeps
Infiltration/ Percolation
Surface Water/ Sediment
Ground water
Bioconcentration
Bioaccumulation
External Gamma
Plants
Animals
S
T
U
V
V

Signatures and certifications:

Checklist completed by (provide name, organization and phone number):

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

Date completed: _____

Verification by a member of ER Project Ecological Risk Task Team (provide name, organization and phone number):

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