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Date: January 28, 2005
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NMED Hazardous Waste Bureau

Mr. James Bearzi
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SUBJECT: REVISION OF SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT METHODS DOCUMENT

Dear Mr. Bearzi:

Enclosed please find two copies of *Screening-Level Ecological Risk Assessment Methods, Revision 2*. This document has been revised to reflect changes in approaches and conditions since the previous 1999 version. The revisions document the methods that have been used in the ecological risk screening process and presented in reports. The revised document is not intended to introduce new, unapproved methods.

The Environmental Stewardship-Remediation Services project requests that the New Mexico Environment Department's Hazardous Waste Bureau review the document and let the project know if there are any questions or concerns. If you or your staff have any questions please contact Richard Mirenda at (505)665-6953 or rmirenda@lanl.gov.

Sincerely,

David McInroy, Deputy Project Director
Environmental Stewardship
Los Alamos National Laboratory

Sincerely,

David Gregory, Federal Project Director
Department of Energy
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13894

Attachment: Screening-Level Ecological Risk Assessment Methods, Revision 2

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Screening-Level Ecological Risk Assessment Methods, Revision 2

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Prepared by
Environmental Stewardship–Remediation Services

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Screening-Level Ecological Risk Assessment Methods, Revision 2

December 2004

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EXECUTIVE SUMMARY

This document provides guidance for screening-level assessments of potential adverse impacts to ecological resources from releases of contaminants to the environment at the Los Alamos National Laboratory (LANL or the Laboratory). It replaces the 1999 screening-level method document developed by the Laboratory. This version of the Screening-Level Ecological Risk Assessment (SLERA) methods document is updated to be consistent with Environmental Protection Agency and State of New Mexico guidance and regulations. This version also incorporates new ecological screening levels (ESLs) developed for air exposures of burrowing mammals, an area use factor approach for screening-level assessments of impacts to populations, and an expanded discussion of the uncertainty analysis portion of the SLERA.

This methods document has two purposes. First, it provides 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. Second, it provides guidance to the Environmental Stewardship–Remediation Services (ENV-RS) project ecological risk assessors. This guidance promotes consistency, rigor, and defensibility in ecological screening investigations and in reporting those investigation results.

Section 1, Introduction, provides a brief introduction to the document. Section 2, Ecological Screening Process Overview, provides an overview of the ENV–RS project screening assessment process (including a process flow diagram). Section 3, Generic Problem Formulation for Ecological Screening Assessments, describes the Laboratory-wide information 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 assessment endpoints (Section 3.5) and of screening-level ecological receptors (Section 3.6) to represent those assessment endpoints.

Section 4, Site-Specific Screening-Level Ecological Risk Assessment, describes in detail the two phases of the screening assessment: the scoping evaluation (Section 4.1) and the screening evaluation (Section 4.2). 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 solid waste management unit (SWMU)[1] or area of concern (AOC); the area may also be a collection of SWMUs and/or AOCs in a watershed or some other aggregate. In this document, the term *site* is used broadly to include these different possibilities.

The basis for the problem formulation is found in Appendix A, Ecological Scoping Checklist. The 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.

The purpose of the screening evaluation is to identify those COPCs that should be retained as chemicals 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.

---

[1] The term SWMU replaces the term potential release site used in previous SLERA versions. A SWMU is defined as any discernable unit at which solid wastes have been placed at any time, regardless of whether the unit was intended for the management of solid or hazardous waste. Such units include any area at which solid wastes have been routinely and systematically released.

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, protective assumptions are made throughout the screening evaluation to ensure that contaminants, exposure pathways, and sensitive species are not missed.

Section 4.2, Screening Evaluation Overview, provides an overview of the methods used for screening. The key component of the screening evaluation is the ESLs that are developed for each chemical and receptor and are media specific. The ESLs 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. To evaluate the potential risk for each COPC, the ESL and the representative site concentration are used to calculate the hazard quotient (HQ). If the HQ for a COPC at a site with only a single COPC is greater than 1.0 or the HQ for a COPC is greater than 0.3 for a site with multiple COPCs, then that chemical is identified as a COPEC. 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 HQ values. HQs are calculated for each screening receptor and each chemical and are considered 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 4.2, 4.3, and 4.4 describe the HQ and HI calculations and present in detail the ESL calculations for various media, receptors, and chemicals.

The ESL, HQ, and HI calculations require toxicity information, including toxicity reference values (TRVs) and knowledge of bioconcentration and bioaccumulation factors for all chemicals for all receptors and media. This information is not provided in this document. The Laboratory documents this information in detail to ensure that the best available studies are used to develop TRVs, ESLs, HQs, and HIs. The Laboratory's ECORISK Database provides the necessary information and supporting detailed documentation (including TRVs and ESLs). The database includes values for the TRVs used to develop ESLs, information on other studies considered for TRVs, transfer and bioaccumulation factors, and exposure parameters for the representative receptor species. The ECORISK Database is updated annually with new ecological toxicity data as appropriate.

Section 4.5, Screening Evaluation/Uncertainty Analysis, describes the uncertainty analysis that follows the COPEC identification and 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 4.6, 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
BCF	bioconcentration factor
BMP	best management practice
Bq	Becquerel
CCC	criterion continuous concentration
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CMS	corrective measure study
CMI	corrective measure implementation
COPC	chemical of potential concern
COPEC	chemical of potential ecological concern
CV	chronic value
DCF	dose conversion factor
DOE	Department of Energy
EqP	equilibrium partitioning method
EPA	Environmental Protection Agency

ER	Environmental Restoration
ESL	ecological screening level
ENV-RS	Environmental Stewardship-Remediation Services
ERL	effects range low
ERM	effects range median
FACR	final acute-chronic ratio
FAV	final acute value
FCV	final chronic value
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	geographic information system
GPS	global positioning system
HI	hazard index
HQ	hazard quotient
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) that is lethal to 50% of test population
LOAEL	lowest observed adverse effect level
LOEC	lowest observed effect concentration
MC	moisture content
MeV	million electron volts
NAWQC	National Ambient Water-Quality Criteria
NCRP	National Council on Radiation Protection
NMED	New Mexico Environment Department
NOAEL	no observed adverse effect level
NOEC	no observed effect concentration
OU	operable unit
ORNL	Oak Ridge National Laboratory
OW	Office of Water (EPA)
P	probability
PAUF	population area use factor
PEL	probable effects level

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 value
SAVF	secondary acute value factor
SCM	site conceptual model
SCV	secondary chronic values
SEC	sediment effects concentration
SI	Standards International
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
Sv	Sievert
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
VOC	volatile organic compound
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 Stewardship–Remediation Services (ENV–RS) project for screening-level assessments of potential impacts to ecological resources resulting from exposure to contaminants. This approach is consistent with the Environmental Protection Agency's (EPA's) "Ecological Risk Assessment Guidance for Superfund" (EPA 1997, 59370), the Guidelines for Ecological Risk Assessment" (EPA 1998, 62809), "Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites (EPA 1999, 70086), and the "Guidance for Developing Ecological Soil Screening Levels (EPA 2003, 85643). This guidance incorporates the assessment endpoints developed in "Generic Assessment Endpoints for Ecological Risk Assessment at the Los Alamos National Laboratory" (LANL 1999, 64137). The guidance in this document is consistent with the New Mexico Environment Department's (NMED's) "Guidance for Assessing Ecological Risks Posed by Chemicals: Screening-Level Ecological Risk Assessment" (NMED 2000, 70107). The approach to ecological risk screening for radionuclides provided in this document is also consistent with the Department of Energy's (DOE's) "Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota" (DOE 2002, 85637) and DOE's "RESRAD-BIOTA: A Tool for Implementing a Graded Approach to Biota Dose Evaluation, User's Guide, Version 1" (DOE 2004, 85639). This version of the document incorporates additional state and federal guidance on ecological risk assessment that has been issued since 1999 as well as updated methods for ecological risk screening developed by the Laboratory and by EPA.

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 exposure to contaminants 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 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 characterize risks accurately 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, receptors, 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 solid waste management units (SWMUs) or areas of concern (AOCs) as well as clusters of SWMUs and/or AOCs. 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 from individual sites to larger areas such as canyon systems. 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 and the application of area use factors for screening potential risks to populations of receptors. The methods described in this document are intended for assessing present-day risk at the site where contamination has been investigated. However, these methods, coupled with the appropriate transport models, may 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

A diagram of the ecological screening process flow is shown in Figure 2.0-1. The large solid boxes designate those steps that are part of the scoping evaluation and those steps that are part of the screening evaluation. During the initial step, the ecological risk assessor should determine if chemicals of potential concern (COPCs) are known or expected to occur at the site. If not, the site should be recommended as not requiring further ecological evaluation and justified in the risk assessment report. Although these recommendations are made for an individual SWMU or AOC, in the remainder of this document, the term *site* is used broadly to represent a SWMU or AOC or an aggregate of SWMUs and/or AOCs.

Scoping Evaluation

Sites being investigated by the ENV-RS project to determine the nature and extent of contamination as well as the potential need for corrective actions must undergo ecological scoping, including a site visit by a member of the ecological risk assessment team and completion of the ecological scoping checklist (described in detail in Section 4.1, 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. The scoping evaluation should address whether a SWMU or AOC should be combined (aggregated) on an "appropriate scale to support risk-based corrective action decision-making" (LANL 2000, 66802) with neighboring SWMUs or AOCs for the purposes of the screening assessment. Sites may be combined based on size, geography, common contaminants, common transport pathways, common land use, common receptors and/or habitat, or on programmatic considerations. For ecological risk, sites may be aggregated on a larger scale than might be used for consideration of human health risk (LANL 2000, 66802). Any aggregation of the SWMUs and/or AOCs under consideration should be established prior to starting the screening-level ecological risk.

After the scoping evaluation, if the ecological risk assessor determines that the site poses no threat to the environment because there are no ecological receptors and/or no pathways to receptors exist, a recommendation is made that no further assessment of ecological risk is necessary. The justification for this recommendation is documented in the risk assessment report.

During scoping, a decision is made about the adequacy of the data and the site conceptual model (SCM) 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 and extent of contamination. Data adequacy in scoping involves determining whether the geographic and biotic limits of sampling, as well as depths and media sampled, match the potential extent of contamination 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[2] and appropriately distributed, the 95% upper confidence limit (UCL) of the mean concentration may be used instead of the maximum value in calculations and comparisons (EPA 2002, 85640).

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

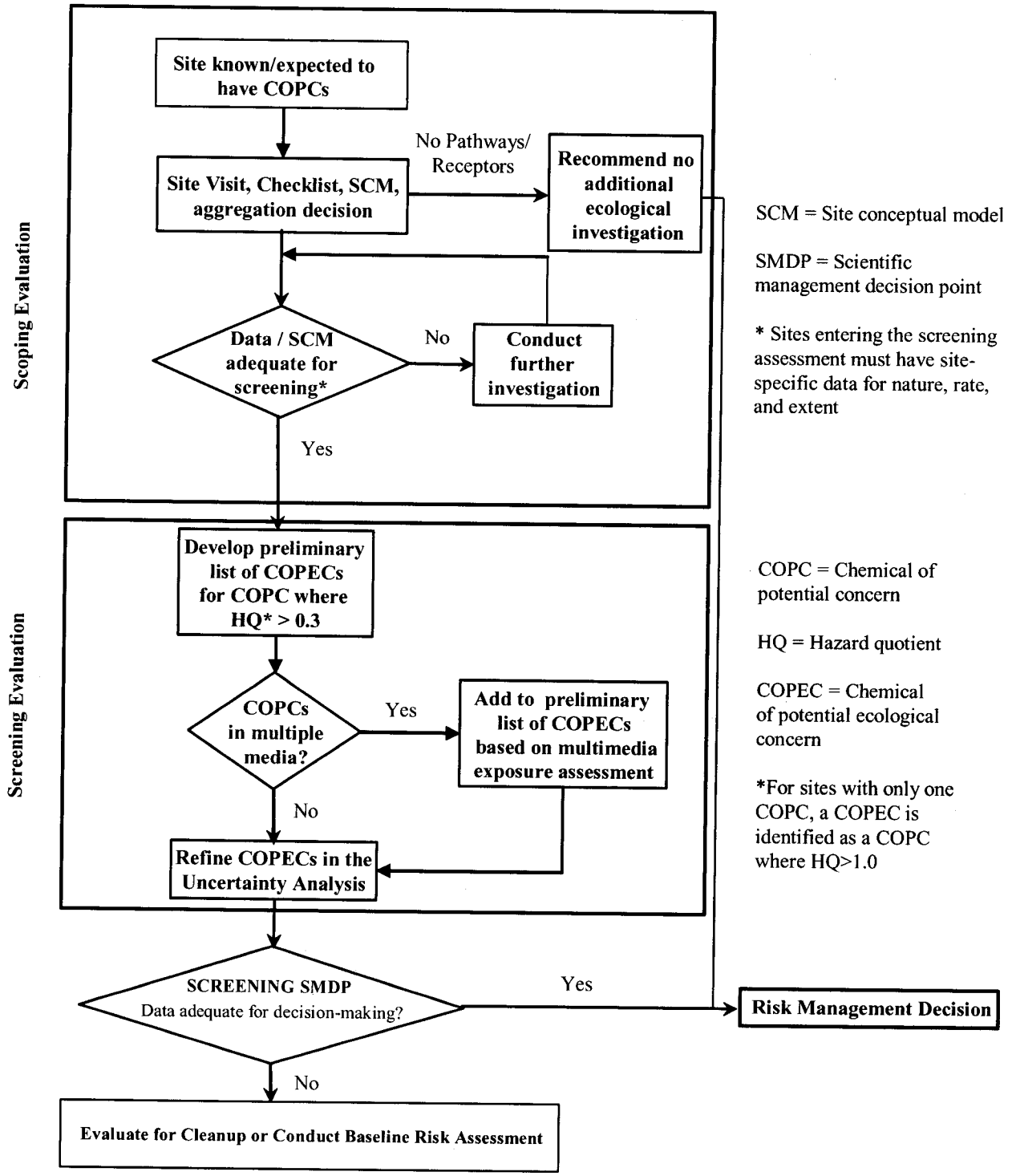


Figure 2.0-1. Process flow for ecological screening assessment

Screening Evaluation

Once the scoping process is complete, the screening evaluation is conducted. The goal of the screening evaluation is to identify the chemicals of potential ecological concern (COPECs), which 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 is addressed in the uncertainty analysis. Figure 2.0-1 shows two steps that may be necessary to generate a preliminary list of COPECs. Identification of COPECs first requires assembling site concentrations and ecological screening levels (ESLs) for all media, receptors, and COPCs. All the ESLs for all receptors for a chemical-media combination are obtained; the lowest ESL for that chemical in that media becomes the "final ESL" used for the ecological screening. If a site with multiple COPCs has concentrations of a chemical in any medium that yield a hazard quotient (HQ) >0.3, then this COPC is considered a COPEC for the site. If a site with a single COPC has concentrations in any medium that yield an HQ>1.0, then this COPC is considered a COPEC for the site. If the site has multimedia contamination, the hazard index (HI) is also calculated for each screening receptor. The HQ is the ratio of each COPC site exposure concentration to the corresponding ESL, and the HI is the sum of HQs for all of the COPCs. If the HI is greater than 1.0, COPCs contributing 0.3 or more to the HI are identified as COPECs.

The ESLs and the toxicity and other parameter information required for their calculation are maintained in a separate document, the Laboratory's ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version). The ECORISK Database is available to anyone performing or reviewing ecological screening assessments for the Laboratory, and notices of updates to this database are issued as new information becomes available.

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 may result in adding or removing COPECs. The list of COPECs is not considered final until the uncertainty analysis is completed. The main components of the uncertainty analysis are described in Section 4.5, 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 scientific management decision point (SMDP) is required. As part of this SMDP, a risk management strategy may be recommended by the risk assessors. Possible recommendations and risk management strategies are discussed in Section 4.6, Risk Interpretation.

3.0 GENERIC PROBLEM FORMULATION FOR ECOLOGICAL SCREENING ASSESSMENTS

As noted in the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA or Superfund) specific ecological risk guidance (EPA 1997, 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

Therefore, problem formulation requires understanding 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). Information on the feeding relationships among animals and plants may 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 relevant biota. In the following sections, the general physical setting of the Laboratory and the surrounding area is summarized, 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 2377 m (7800 ft) on the flanks of the Jemez Mountains to about 1890 m (6200 ft) at their eastern termination above the Rio Grande. 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 mi north-northeast of Albuquerque and 25 mi northwest of Santa Fe. 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 (Figure 3.1-1). 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 organisms.

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 Work Plan for Environmental Restoration Project" or IWP (LANL 2000, 66802, p. 2-41).

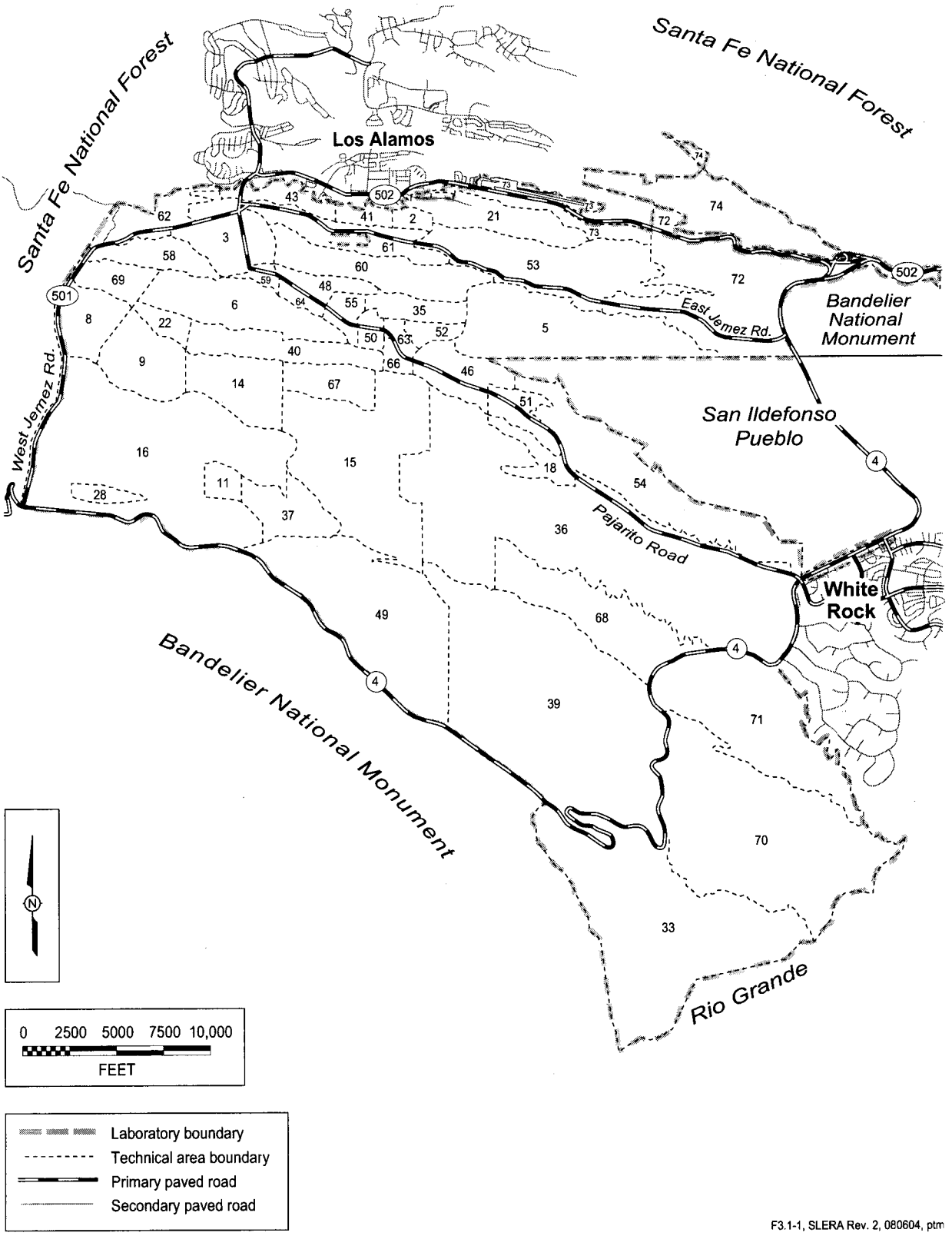


Figure 3.1-1. Technical areas of Los Alamos National Laboratory in relation to surrounding landholdings

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 persistence of wind influence the airborne transport of contaminants. High winds, common in the spring, can result in atmospheric transport of contaminants. Additional discussion of atmospheric pathways may be found in the IWP (LANL 2000, 66802, pp. 2-41 to 2-44). The role of climate in the atmospheric contaminant pathway is considered part of the site-specific scoping evaluation.

Intense thunderstorms in the summer can cause erosion of unstable sediment or soil. The form, frequency, intensity, and evaporation potential of precipitation strongly influences surface water runoff and infiltration of contaminants. As discussed below, fires also change hydrological regimes, and small precipitation events may lead to large amounts of runoff.

3.1.3 Geology and Soils

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 4.1, Scoping Evaluation). The geologic and hydrologic characteristics in and around the Laboratory as they relate to the potential for contaminant transport are complex. A detailed discussion is provided in Sections 2.2.1 and 2.2.2 of the IWP (LANL 2000, 66802, pp. 2-7 to 2-37). 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 SWMUs and AOCs at the Laboratory (LANL 1990, 47588).

Geology

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 3.1-2) are formed in Bandelier Tuff, which includes ash fall, ash fall pumice, and rhyolite tuff. The tuff is more than 305 m (1000 ft) thick in the western part of the plateau and thins to about 79 m (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 mil yr ago. Deep canyons are incised into the Bandelier Tuff and exposed 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 groundwater zones and form a source for springs and seeps in the area. Faults, cooling joints, and fractures potentially occur throughout the Pajarito Plateau (LANL 2000, 66802, pp. 2-23 and 2-24).

On the western part of the Pajarito Plateau, the Bandelier Tuff overlaps onto the Tschicoma Formation, which consists of older volcanic rock that comprises 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 1006 m (3300 ft) thick.

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

Figure 3.1-2. Topography of the Los Alamos area

Soils

Soil erodability is important to understanding the potential for contaminant transport and accurately completion the "contaminant transport information" in site-specific scoping evaluations (Section 4.2, Scoping Evaluation). Soils on the Pajarito Plateau were initially mapped and described by Nyhan et al. (1978, 05702). A large variety of soils and sediment have developed on the Pajarito Plateau as the result of interactions of the underlying bedrock, slope, biota, 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, 05702). General patterns of soil erosion rates are summarized by the following text from Section 2.2.1.6 (Geomorphic Processes) of the IWP (LANL 2000, 66802, p. 2-25):

> 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. The rates and processes of erosion may differ significantly between the north and south slopes of the canyons. Given current vegetation and climate, the more extensive exposure of bedrock on south-facing sides and greater soil cover on north-facing sides suggest that erosion rates of fine-grained material that can be transported by runoff are higher on the drier, less-vegetated, south-facing sides of canyons, although this material is largely retained on the north-facing slopes.

The mesa tops generally consist of finer-textured soils and the canyon bottoms consist of relatively coarse sediment. Finer-textured soils of mesa tops are prone to overland runoff whereas soil fines may accumulate in canyon bottoms. The latter are subject to mobilization during flood events.

3.1.4 Hydrology

Surface water on 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 the processes of evaporation, transpiration, and infiltration described in the "Core Document for Canyons Investigations" (LANL 1997, 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 arrive at the Rio Grande through drainages that extend from the Laboratory in a southwest direction but not as continuous flow. Only five of the canyons within Laboratory boundaries contain reaches with perennial water flow. These canyons are Los Alamos Canyon, Pajarito Canyon, Water Canyon, Ancho Canyon, and Chaquehui Canyon. In addition to these limited natural perennial reaches, several effluent-supported reaches also exist within the watershed (LANL 2000, 66802).

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 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 provides a detailed discussion of this subject (LANL 2000, 66802, pp. 2-28 to 2-37).

3.1.5 Biology

The biota within the Laboratory includes approximately 500 plant species, 29 mammal species, 200 bird species, 19 reptile species, 8 amphibian species, and thousands of insect species (LANL 2000, 66802). 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, 62887) and five state-protected T&E species (Loftin and Haarmann 1998, 62881) have been identified at the Laboratory (LANL 1999, 62887). The federally listed species include the southwestern willow flycatcher (*Empidonax traillii extimus*), arctic peregrine falcon (*Falco peregrinus tundrius*), 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, 62887). Results of preliminary risk assessments for the Mexican spotted owl, bald eagle, and southwestern willow flycatcher are available in Gallegos et al. (1996, 57915), Gonzales et al. (1997, 62879), Gonzales (1998, 62349), Gonzales (1998, 62350), and Gonzales et al. (2004, 85207). 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 may be found in LANL (1999, 62887) and in Loftin and Haarmann (1998, 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 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 developed a post-Cerro Grande Fire vegetation land cover map (Plate 1) to support endangered species modeling and other region-wide environmental studies (McKown et al. 2003, 87150). The land cover map identifies areas by the dominant overstory vegetation. The map was developed based on a Landsat Enhanced Thematic Mapper Plus satellite scene acquired on June 4, 2001. The version of the vegetation land cover map in Plate 1 is based on the eight taxonomic vegetation classes (Table 3.1-1) and on resolution smoothed to a quarter-hectare minimum mapping unit. Estimates of the accuracy of the mapping technique compared to field data are provided in Land Cover for the Eastern Jemez Region (McKown et al. 2003, 87150). The resulting cover types include major vegetation zones and physiognomic types important to the distribution and abundance of several T&E species (McKown et al. 2003, 87150). The approximate areal extent of each cover type on Laboratory property is provided in Table 3.1-1. The ecologist or risk assessor who conducts scoping verifies the vegetation cover type during the site visit that supports the site-specific problem scoping (Section 4.1, Scoping Evaluation).

Table 3.1-1
Approximate Areal Extent of Land Cover Types at Los Alamos National Laboratory

Cover Type	Area (mi^2)	Area (ha^2)	Proportion of Total Area (%)
Open water	0.05	12.6	0.11
Aspen-riparian-wetland	0.78	201.6	1.85
Mixed conifer-spruce-fir	0.96	248.9	2.28
Grass species	4.33	1121.5	10.30
Shrub species	4.86	1258.4	11.56
Urban-sparse-bare rock	5.67	1468.8	13.50
Ponderosa pine	8.20	2123.1	19.51
Piñon-juniper	17.16	4443.8	40.84
Total	42	10,879	100

Note: Table from McKown et al. 2003, 87150, Appendix E based on taxonomic vegetation classes and areal extent within the Laboratory boundary calculated from 15-m map.

The land cover types can be subdivided to correspond with the National Vegetation Classification System (McKown et al. 2003, 87150). The elevation and climatic gradients in the region of the Laboratory most strongly influence distribution of three vegetative cover types defined by their dominant tree species and by their structural characteristics; these include piñon-juniper woodlands, ponderosa pine forests, and mixed conifer-spruce-fir forests. In contrast, aspen-riparian-wetland areas, grass species areas, shrub species areas, open water, and urban-sparse-bare rock 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.

Mixed conifer-spruce-fir forests. Mixed conifer forests may be found above 2070 m (6900 ft) above sea level (asl), blended with ponderosa pine communities, but they also extend to lower elevations on north-facing slopes of canyons. These communities continue to the highest elevations of the Sierra de los Valles, 3150 m (10,500 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-riparian-wetland. 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 found mostly 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. Riparian areas and wetlands (Section 3.1.6) are also included in this vegetation land cover type.

Grass species. Grass species areas are dominated by grasses, narrow-leaf plants (e.g., yucca), and species that invade disturbed areas (colonizing species). Forbs and other nonshrubby species may be dominant components of these communities. Shrubs and trees are absent or rare. The grass species cover type may include areas undergoing post-fire succession, abandoned homestead areas, montane meadows, and subalpine grasslands.

Shrub species. These areas include both evergreen, microphyllus shrubs and temperate, cold-deciduous shrub species. Post-fire shrub-sized sprouts of aspen, Gambel oak, and New Mexico locust are also included in this vegetation type.

Ponderosa pine. This vegetation consists of open-canopied woodlands with needle-leaved evergreen trees, primarily ponderosa pine (*Pinus ponderosa*). There may be an understory of Gambel oak or grasses and bare ground between the trees.

Piñon-juniper. This vegetation cover also consists of open-canopied woodlands with needle-leaved evergreen trees, primarily piñon pines (*Pinus edulis*) and junipers (*Juniperus monosperma*); there may be bare soil under the trees or an understory of Basin big sage (*Artemisia tridentate)* and blue grama grass (*Bouteloua gracilis).*

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 3.1-1.

Urban-sparse-bare rock. This land type includes all undeveloped land covered by less than 7% vegetation. These land surfaces are dominated by cobbles, boulders, bedrock, or bare ground, including tuffaceous cliffs, basalt cliffs, felsenmeers, and basalt talus. Areas of sparse vegetation resulting from development, such as the Los Alamos townsite, White Rock, and some technical areas, are also part of the vegetation land cover class.

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, 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 county facilities. Wetlands locations and areal coverage for 90% of the Laboratory have been determined using the global positioning system (GPS) integrated with the geographic information system (GIS) (Bennett 1999, 62891). One hundred fifty-six wetland areas have been identified at the Laboratory, comprising a total of approximately 49 acres (Sigler 1999, 62878). The approximate locations of many of the larger wetlands are shown in Plate 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).

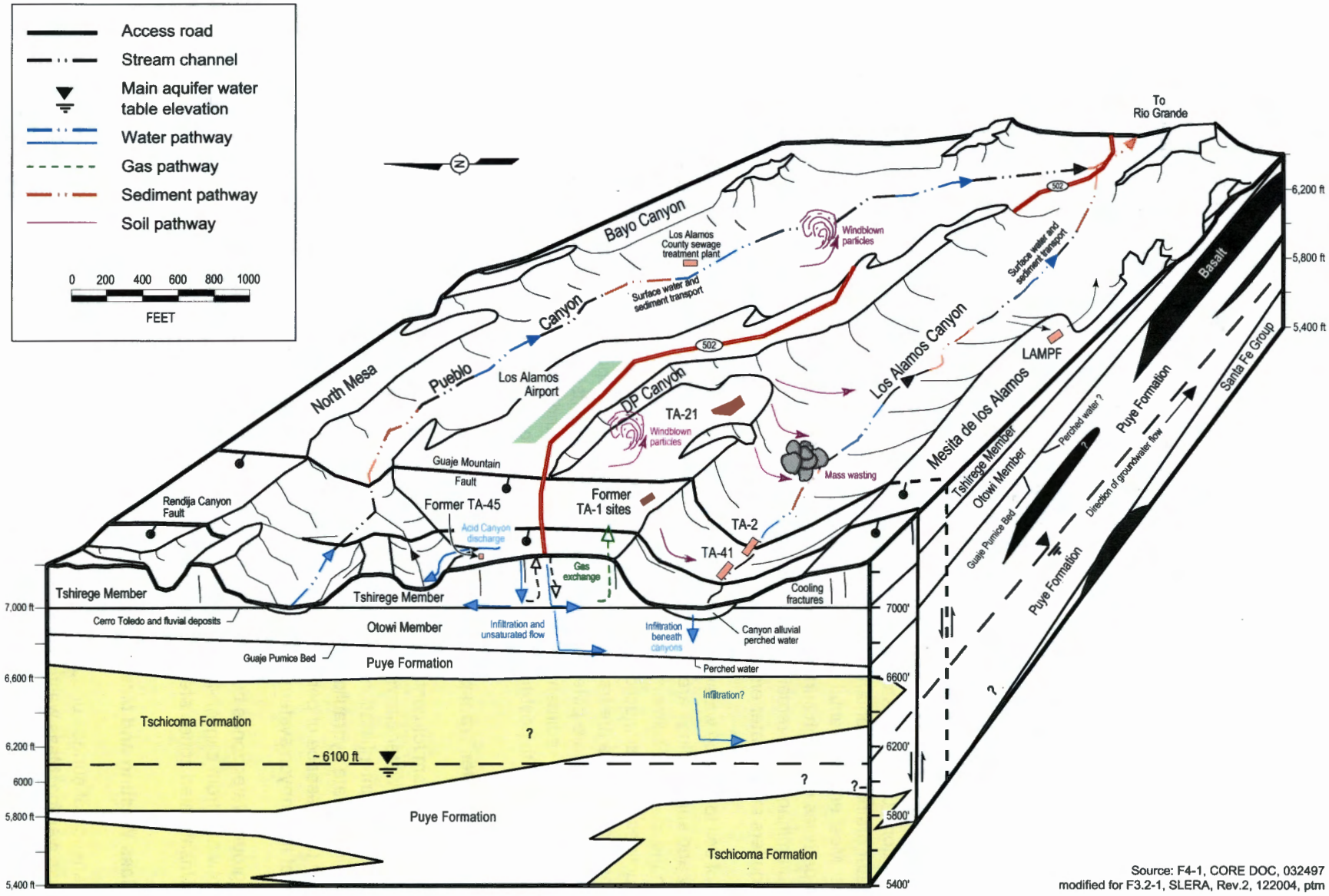
The protection of wetland ecosystems at the Laboratory from the impacts of contaminants is especially important because of the diversity of associated fauna and because wetlands provide significant potential contaminant uptake pathways. These pathways include food web, direct media contact, and gamma 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, groundwater 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. Figure 2.2-15 in the IWP provides a map of the Laboratory showing wetlands by location and type (LANL 2000, 66802, p. 2-40).

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 environmental contamination. Figure 3.2-1 shows the key transport pathways:

- hydrologic transport (e.g., surface water and groundwater),
- physical transport (e.g., mass wasting of cliffs), and
- atmospheric transport (e.g., dust resuspension).

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



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modified for F3.2-1, SLERA, Rev.2, 122004, ptm

Figure 3.2-1. Key transport pathways

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 mil tons of suspended sediment to Cochiti Reservoir (LANL 1997, 55622). A more thorough description of canyon streams is found in "Core Document for Canyons Investigations" (LANL 1997, 55622).

Sediment transport by surface water may 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.

Transport of soil to surface water by runoff has been significantly increased in those areas of the Laboratory and surroundings areas that burned during the 2000 Cerro Grande Fire. In addition to an increase in the mass of sediments transported in the years following the fire, the concentrations of both nonradionuclides and radionuclides in those sediments also increased significantly (Kraig et al. 2000, 85536). These sediments are transported downstream and are deposited at some locations where these elevated concentrations are potentially available to both terrestrial and aquatic receptors. Increased flow also leads to erosion of sediment deposits in other settings and contaminants in these mobilized sediments would mix with post-fire material and other upstream sediment sources.

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, 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, which may be partly directed and accelerated by faults and fractures. Unsaturated zones are considered only an occasional transport pathway.

3.2.2 Mass Wasting and Mass Deposition

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.

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 is a means of dispersing contaminants. Wind resuspension and transport of surficial contaminant-laden soil or sediment is not a significant transport pathway because the volume of contaminated media mobilized by this pathway is small compared to the total amount of soil to which the receptor is exposed. Exposure of surface-dwelling animals to vapors does not represent a significant pathway because vapors disperse in the open atmosphere. Within burrows, vapors from subsurface contamination may accumulate and result in potentially significant exposures of animals occupying burrows.

3.3 Exposure Pathways

Contaminants associated with surface soil may be available to biological receptors through the following exposure pathways:

- Rain splash or saltation-creep of contaminated soil onto plants
- Root uptake of water-soluble contaminants
- Incidental ingestion of soil
- Dermal contact with soil
- Inhalation of particulates by animals during aboveground activity or while in burrows
- Deposition of particulates on foliage
- Deposition of particulates on animals, and subsequent ingestion during grooming
- Food web transport (consumption of contaminated plants and animals)
- Direct exposure to soil containing gamma-emitting radioactive contaminants

Contaminants associated with sediments or surface water may be taken up by biota primarily through the following exposure pathways:

- Ingestion of surface water
- Root uptake of surface water
- Root uptake of water-soluble contaminants from sediment
- Incidental ingestion of sediments
- Rain splash or saltation-creep of contaminated sediment onto plants
- 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
- Direct exposure to surface water containing gamma-emitting radioactive contaminants (immersion)

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 as vapors 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
- Uptake by plants of contaminants present as vapors

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 important 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. Figures 3.4-1 and 3.4-2 represent the terrestrial and aquatic functional food webs for the Laboratory, respectively. The food webs are 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) are considered one feeding guild, browsers/grazers another, and top carnivores yet another. Feeding guilds, may be organized in many ways, from general to specific, but they are too numerous to describe here.

A food web organized by feeding guilds forms a basis for selecting individual species from each guild that represent the guild as a whole. This approach forms the basis of receptor selection for the ecological screening assessments at the Laboratory. The food webs for the Laboratory include three fundamental trophic positions: producers (vascular and nonvascular plants); consumers (herbivores, omnivores, carnivores, and parasites); and decomposers. 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. Figure 3.4-1 shows a terrestrial food web for the Laboratory and Figure 3.4-2 is a food web specific to Laboratory aquatic habitats.

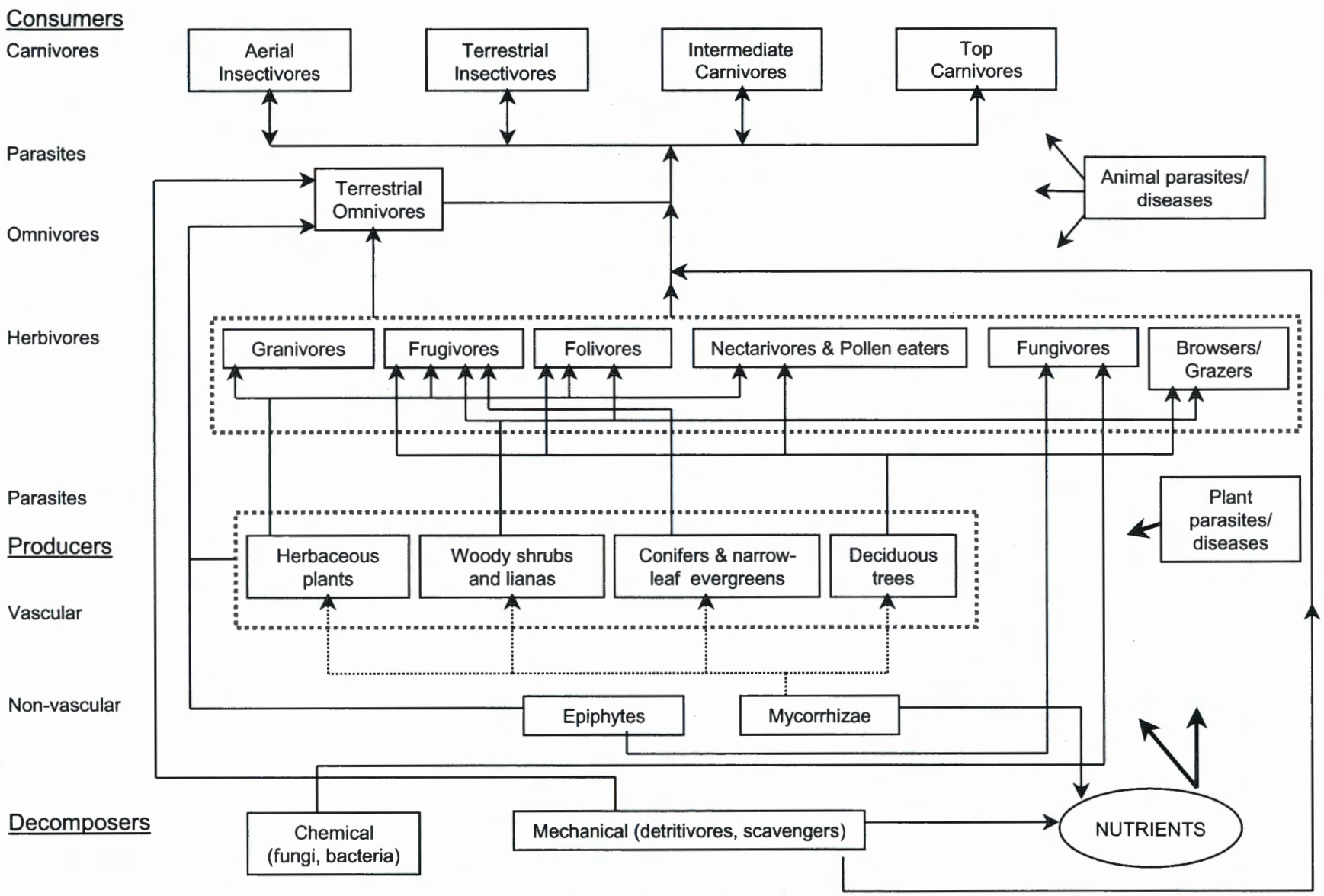


Figure 3.4-1. Terrestrial food web based on feeding relationships of the biota at the Laboratory

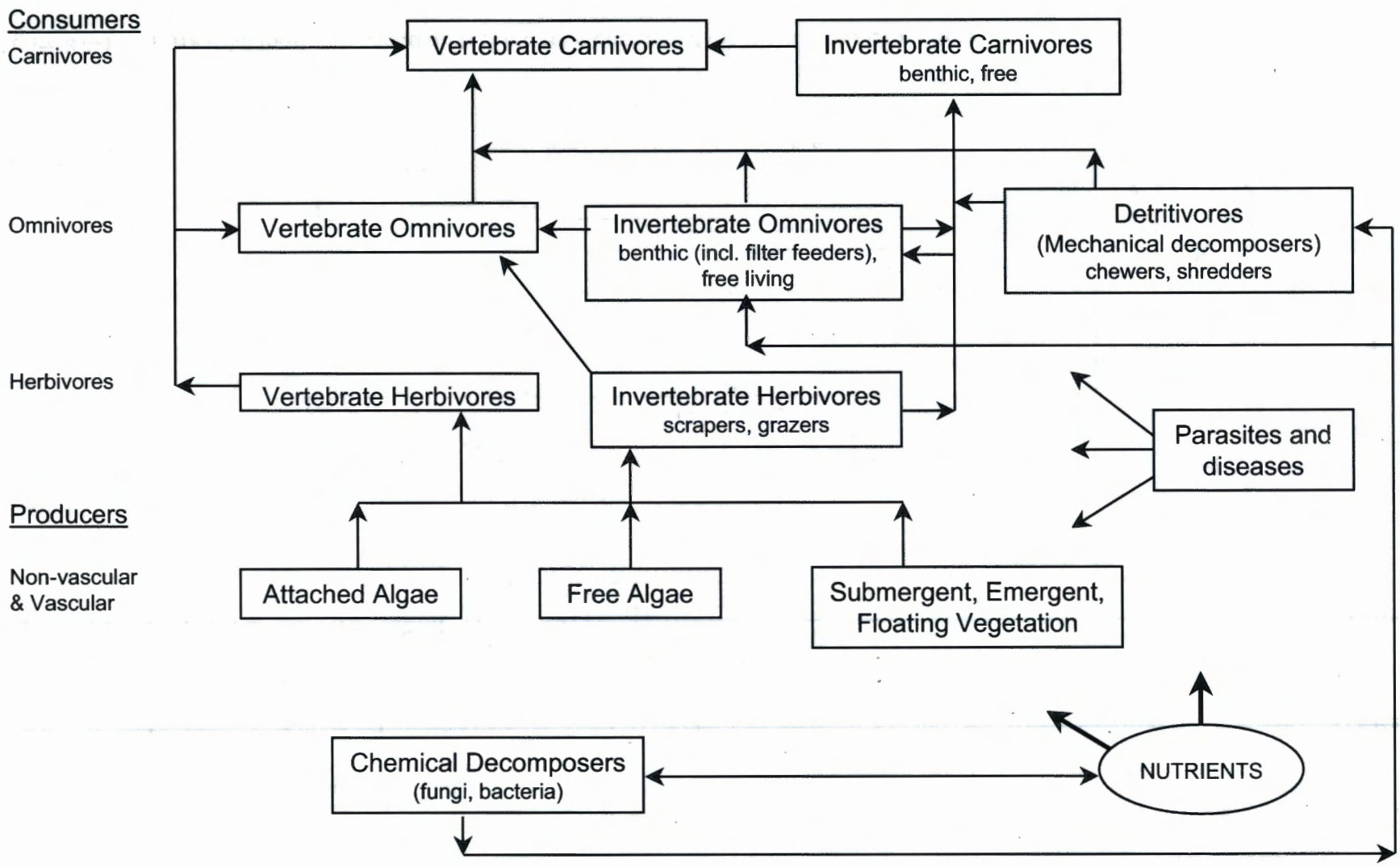


Figure 3.4-2. Aquatic food web based on feeding relationships of the biota at the Laboratory

As shown in Plate 1, terrestrial communities dominate most Laboratory areas. Aquatic environments on the Laboratory are of limited spatial extent and typically occur in canyon settings. Therefore the primary connection between the terrestrial and aquatic food webs is not riparian species but rather aerial insectivores, for which receptors are designated as part of the terrestrial food web in Section 3.6. Separate screening receptors are developed for the terrestrial and aquatic food webs described in Section 3.6 because of the limited connectivity between the aquatic and terrestrial systems at the Laboratory. Vertebrate herbivores, omnivores, and carnivores are listed on the aquatic food web to represent the trophic positions of fish species. However, fish species do not occur in the ephemeral or permanent reaches of water within the Laboratory; therefore, these feeding guilds do not have screening receptors but are included to acknowledge that this portion of the food web exists downstream in the Rio Grande. The SLERA methodology explicitly addresses only those receptors found on the Laboratory, not the additional species found in the Rio Grande itself. The dashed lines in Figure 3.4-1, enclosing a number of guilds in a single rectangle, represent broad categories for which a single member may suffice as a screening receptor.

3.5 Assessment Endpoints

To represent the feeding guilds in the food webs as described in Section 3.4, some attribute of that receptor must be selected as an assessment endpoint, an explicit expression of the environmental value to be protected. These endpoints should be ecologically relevant and should help sustain the natural structure, function, and biodiversity of an ecosystem or its components (EPA 1998, 62809). In a screening-level assessment, assessment endpoints are any adverse effects on ecological receptors, where receptors are populations and communities (EPA 1997, 59370).

Superfund guidance also indicates that an ecological risk assessment should be designed to protect local populations and communities of biota rather than individual organisms, except for listed or candidate T&E species or treaty-protected species (EPA 1999, 70086). The protection of individuals within these designated protected species could also be protected at the population level; the populations of these species tend to be small, and the loss of an individual adversely affects the species.

In accordance with this guidance, the Laboratory developed generic assessment endpoints (LANL 1999, 64137) to ensure that values at all levels of ecological organization are considered in the ecological screening process. These general assessment endpoints can be measured using impacts on reproduction, growth, and survival to represent categories of effects that may adversely impact populations. In addition, specific receptor species, described in Section 3.6, are chosen to represent each functional group. The receptor species were chosen based on their presence at the site, their sensitivity to the COPCs, and their potential for exposure to those COPCs. These categories of effects and the chosen receptor species were used to select the types of effects seen in toxicity studies considered in the development of the toxicity reference values (TRVs). Toxicity studies used in the development of TRVs included only studies in which the adverse effect evaluated affected reproduction, survival, and/or growth. The toxicity information sources and uses are discussed further in Sections 4.3 and 4.4.

The selection of receptors and assessment endpoints are designed to be protective of both the representative species used as screening receptors and the other species within their feeding guilds and the overall food web for the terrestrial and aquatic ecosystems. Focusing assessment endpoints on these general characteristics of species that affect populations (versus biochemical and behavior changes that may affect only the studied species) also ensures applicability of the estimates of affect to the ecosystems of concern.

3.6 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 ecological screening. Therefore, the rationale behind receptor selection is to choose an appropriate set of receptors that address the primary feeding relationships outlined in Section 3.4, Functional Food Web. Receptor selection facilitates the determination of potential adverse ecological impacts across the Laboratory and satisfies the following criteria (based on Fordham and Reagan 1991, 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 T&E and other species of special interest or concern.
- Toxicity information is available that indicates the receptor is sensitive to contaminants occurring at the Laboratory.
- Exposure information for the species is available, and these data show that the species has greater exposure per unit body mass than other candidate species (small species typically have greater intake rates per unit body mass based on allometric relationships [e.g., EPA 1993, 59384]).
- The home range of the receptor is of an appropriate spatial scale for ecological evaluations at the SWMU or AOC or site aggregate scale, leading to selecting species of small body weight and therefore small home range to maximize exposure at most SWMUs or AOCs (<0.1 ha to several ha in area).

Given these criteria, the selection of receptors for the Laboratory is outlined below. The selection of terrestrial receptors, including those considered to be 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 nonradionuclide contaminants in aquatic environs, however, the Laboratory has chosen methods, recommended by NMED and 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 protecting roughly 95% of all aquatic organisms, and thus is inclusive of all trophic guilds illustrated in Figure 3.4-2. The use of benchmarks for screening aquatic environments is recommended in EPA guidance (EPA 1996, 62792). The details for the selection and application of water and sediment benchmarks for nonradionuclide chemicals are described in Section 4.3.3, Sediment ESLs, and Section 4.3.4, Water ESLs. Specific aquatic receptors to evaluate effects from radionuclides are described in Section 4.4.3.

Terrestrial Receptors

Table 3.6-1 summarizes the factors that led to the selection of the eight terrestrial, four aquatic, two aerial, and one burrowing receptor species used for screening. 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 using a generic plant, a broadly protective view of the methods for development of ecological screening levels was chosen, as discussed in Section 4, Site-Specific Screening-Level Ecological Risk Assessment.

Table 3.6-1
List of Receptor Species Selected for Screening at the Laboratory

Receptor Category	Receptor Species	Selection Factors
Terrestrial autotroph (producer)	Generic plant	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
Soil-dwelling invertebrate	Earthworm	Represents decomposer group 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
Mammalian herbivore	Desert cottontail	Food source for carnivores Ubiquitous and abundant Exposure data and toxicity data available Surrogate for economically important browsers (deer and elk)
Mammalian omnivore	Deer mouse	Food source for carnivores Ubiquitous and abundant Exposure data and toxicity data available Surrogate for T&E (New Mexico meadow jumping mouse)
Mammalian insectivore	Montane shrew	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 all terrestrial insectivores, including T&E (Jemez Mountain salamander)
Three diets modeled: Avian omnivore Avian herbivore Avian insectivore	American robin	Food source for some carnivores Exposure data available Large fraction of soil in diet
Two diets modeled: Intermediate carnivore Top carnivore	American kestrel	Surrogate for 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 4.2)

Table 3.6-1 (continued)

Receptor Category	Receptor Species	Selection Factors
Top carnivore	Red fox	Exposure data available Addresses potential biomagnification from soils Conservative choice for this category, given the food intake to body weight ratio (see Section 4.2)
Aquatic autotroph (producer)	Algae (for radionuclides only)	Food source for aquatic animals Provides structure (substrate) for animals Ubiquitous and abundant Exposure and toxicity data available
Aquatic omnivore/herbivore	Daphnids (for radionuclides only)	Food source for carnivores High exposure to contaminated water and sediment Ubiquitous and abundant Exposure and toxicity data available *Daphnia* and *Cerodaphnia* typically the most sensitive aquatic organisms for a variety of contaminants
Aquatic herbivore (grazer)	Aquatic snails (for radionuclides only)	Food source for some carnivores (e.g., fish) High exposure to contaminated sediment Ubiquitous and abundant Exposure and toxicity data available
Intermediate carnivore	Fish (for radionuclides only)	Representative of potential waterborne contaminant effects in the Rio Grande High potential exposure to contaminants; potentially sensitive to persistent bioaccumulators and biomagnifiers
Mammalian aerial insectivore	Occult little brown myotis bat	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
Avian aerial insectivore	Violet-green swallow	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
Burrowing mammal	Pocket gopher (for air pathway only)	Representative for potential inhalation exposure inside a burrow for fossorial or semifossorial mammals (mouse, gopher, rabbit, fox) Exposure through air pathway only and evaluated only for vapor-phase COPCs

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 of 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

montane shrew *(Sorex monticolus)* 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 montane 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 because of 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 because of its high food consumption rate per unit body mass. The American robin is 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 Mexican spotted owl *(Strix occidentalis lucida)*. Additionally, abundant information has been gathered for the kestrel's biology, and 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. A large fraction of the brown myotis bat's diet consists of emergent aquatic insects, as the habitats surrounding water are favorite hunting haunts. The violet-green swallow is common on Laboratory grounds, and some portion of its diet consists 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 information is available on their general biology.

The pocket gopher (*Thomomysus bottae*) was chosen as receptor for air inhalation within a burrow because it represents several fossorial and semifossorial species (gophers, mice, rabbits, foxes) that may occupy burrows at sites with subsurface vapor-phase COPCs present. Gophers spend most of their time underground. Although the deer mouse and shrew have smaller body weights and higher weight-normalized air inhalation rates, these species are likely to spend much less time underground over the course of a year than a gopher. Thus, pocket gophers are a protective representative for all the burrowing mammal species.

Figure 3.6-1 shows the terrestrial food web with a box representing each screening receptor species superimposed over the feeding guilds represented by that receptor. 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, 59384).

Aquatic Receptors

No specific aquatic receptors were chosen for the screening assessment of nonradiological contaminants. Methods adopted for screening are considered by the EPA (e.g., EPA 1995, 62787; EPA 1996, 62792) and others (e.g., Jones et al. 1997, 62789) to be protective of a large fraction (roughly 95%) of aquatic organisms at large (plants, invertebrates, and vertebrates). Although few vertebrates reside in the aquatic realms of the Laboratory (primarily herpetiles), it was considered prudent to adopt methods that are otherwise considered pervasively protective and that include organisms that may be found in the Rio Grande (e.g., fish). The aquatic food web, as shown in Figure 3.4-2, is useful for organizing the scoping portion of screening, but for contaminant-based ecological screening comparisons for nonradionuclides, the methods employed broadly cover all species represented in all trophic guilds.

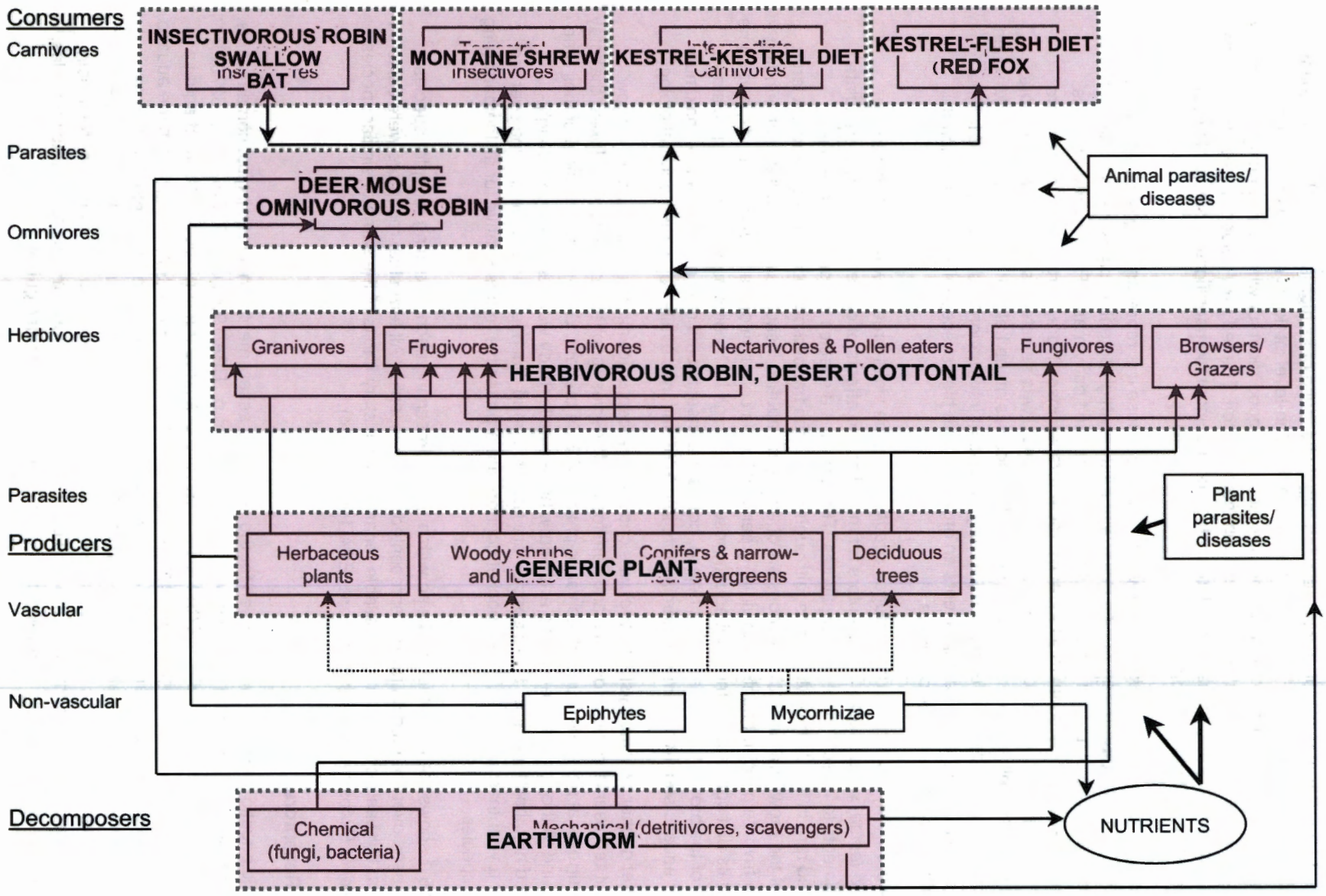


Figure 3.6-1. Screening receptors for terrestrial food web for nonradionuclides and radionuclides

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.

Figure 3.6-2 shows the aquatic food web with a box representing each screening receptor species superimposed over the feeding guilds represented by that receptor. 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 may be considered functionally similar to shrews; therefore, they are covered by terrestrial screening procedures. It is assumed that juvenile salamanders or other amphibians are represented by the aquatic herbivore and omnivore receptors described above.

4.0 SITE-SPECIFIC SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT

The screening-level ecological risk assessment is conducted only for sites known or suspected to have COPCs present in soil, sediment, or water. Sites without COPCs do not require an ecological screening; a recommendation not to conduct any further ecological investigation at these sites is included in the risk assessment report.

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

1. The scoping evaluation (or problem-formulation phase) described in Section 4.1;
2. The screening evaluation (or the screening-level risk and uncertainty-analysis phase) described in Sections 4.2 to 4.5; and
3. Risk interpretation (or screening-level risk characterization) described in Section 4.6.

4.1 Scoping Evaluation

The goals of the ecological scoping evaluation are to identify 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.

4.1.1 Scoping Checklist

The purpose of the ecological scoping checklist is 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;
- determine if adequate data exist for the screening evaluation, primarily as related to nature 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 SCM (e.g., what are the dominant/important transport pathways, exposure routes, and receptors).

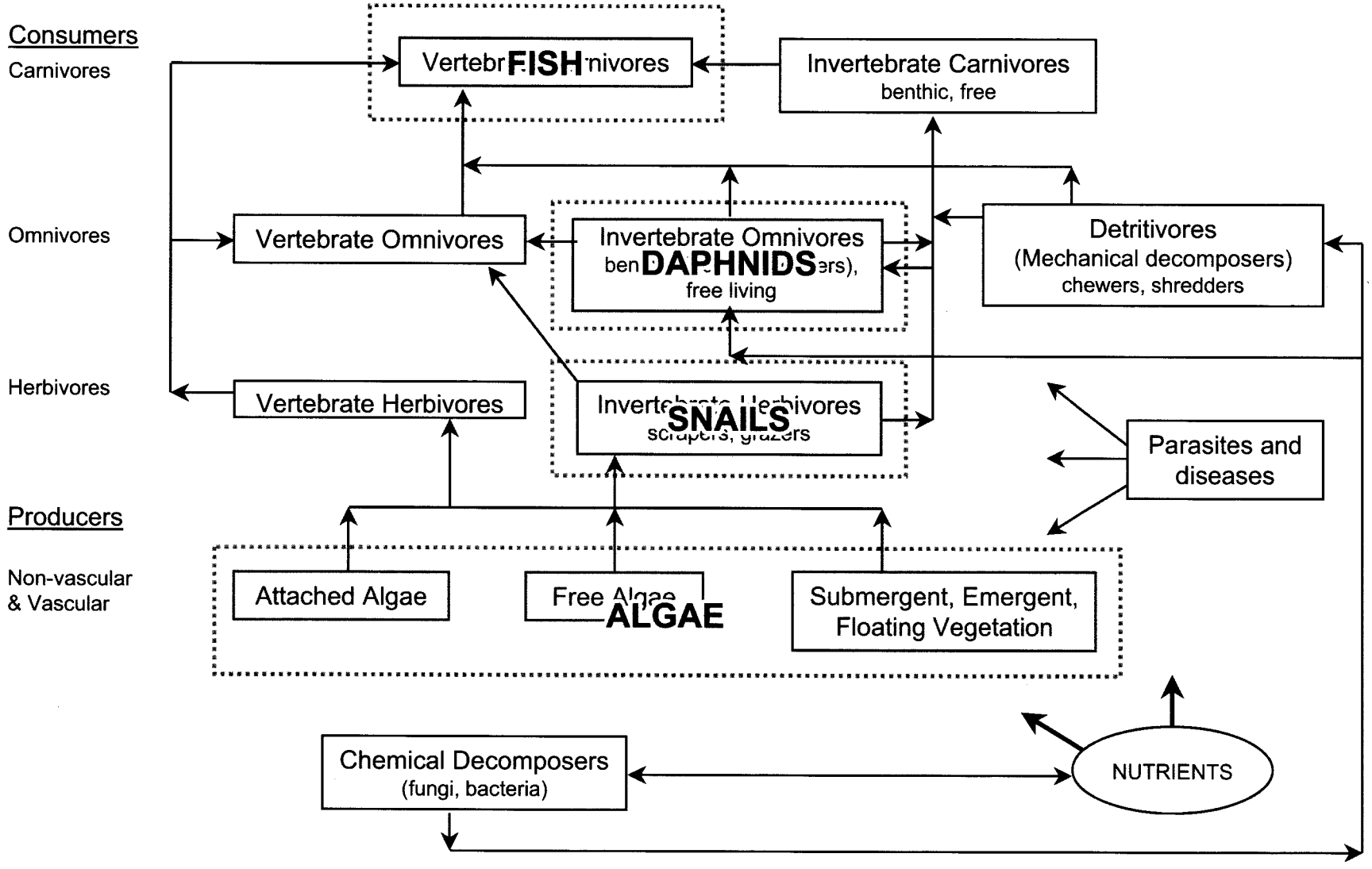


Figure 3.6-2. Screening receptors for aquatic food web for radionuclides

Completion of the ecological scoping checklist consists of three steps, detailed in Sections 4.1.1.1 through 4.1.1.3:

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 Step 1 and collecting field notes for completing the SCM. 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 and extent of contamination.
3. Completing the SCM diagrams identifies the complete and incomplete exposure pathways as well as the major and minor pathways.

4.1.1.1 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 document for applicable operable unit [OU] and/or TA); (2) information on site erosion potential; (3) the Resource Conservation and Recovery Act (RCRA) facility investigation (RFI) work plan or report, as applicable, that provides information on contamination source, sample locations, analytical suites, and sample results; (4) GIS maps that show (if applicable) neighboring SWMUs and AOCs, 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 SWMUs and AOCs, is often necessary. The results of the meeting (or equivalent) are documented in Part A of the ecological scoping checklist (Appendix A). The information required for Part A of the checklist includes (1) site identification; (2) nature of releases (solid, liquid, gaseous, or other); (3) a list of the primary impacted media (soil, water/sediment, subsurface [greater than 1 m (3 ft) in depth], or other); (4) specification of the applicable vegetation classes (open water, aspen-riparian-wetland, mixed conifer-spruce-fir, grassland, shrubland, urban-sparse-bare rock, ponderosa pine, and piñon-juniper; (5) identification of T&E habitat, if present (list species if applicable); (6) a list and description of neighboring/contiguous/upgradient SWMUs and AOCs (discuss whether it is necessary to aggregate the site with additional SWMUs and/or AOCs for screening); (7) standard operating procedure (SOP-) 2.01, Revision 1, information (runoff score and the terminal point of surface water transport[3]); and (8) documentation of other scoping meeting notes (as appropriate).

The project leader for the site is responsible for arranging the scoping meeting before the site visit, if needed. Scoping meeting participants should include the project leader, ecological risk assessor, regulatory compliance expert, and other site subject-matter experts, as necessary (i.e., a soil scientist, data steward, chemist, biological resources expert, geohydrologist, and field sampling personnel).

[3] SOP-2.01 Revision 1 (previously known as Administrative Procedure-4.5) provides site-specific information on the potential for erosion from surface water. This document considers factors relating to the cover (vegetative and nonvegetative), slope, surface water run-on sources, and other 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 SWMU or AOC. The terminal point of surface water discharge helps to determine if erosion from the SWMU or AOC reaches aquatic settings.

4.1.1.2 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 a qualitative evaluation of whether site data provide adequate information to determine the nature, 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 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 approximately locate relevant biological features (measuring tape and/or rangefinder and pin flags or other markers to specify locations for surveying).

Part B of the checklist is 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 whether ecological receptors are present at the site. Contaminant transport information, emphasizing surface water and 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).

If no complete pathways to receptors and no transport pathways to off-site receptors are present, the remainder of the checklist (last part of Part B and Part C) is not completed, 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 off-site 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 receptors and pathways are present, then subsequent questions in Part B involving data adequacy are addressed. Specifically, do existing data provide adequate information on the nature 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 assessor's 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 is not adequately defined to permit an ecological assessment, a recommendation for additional sampling should be provided. Once data issues are resolved, the process of scoping and screening the site for potential ecological impacts should proceed.

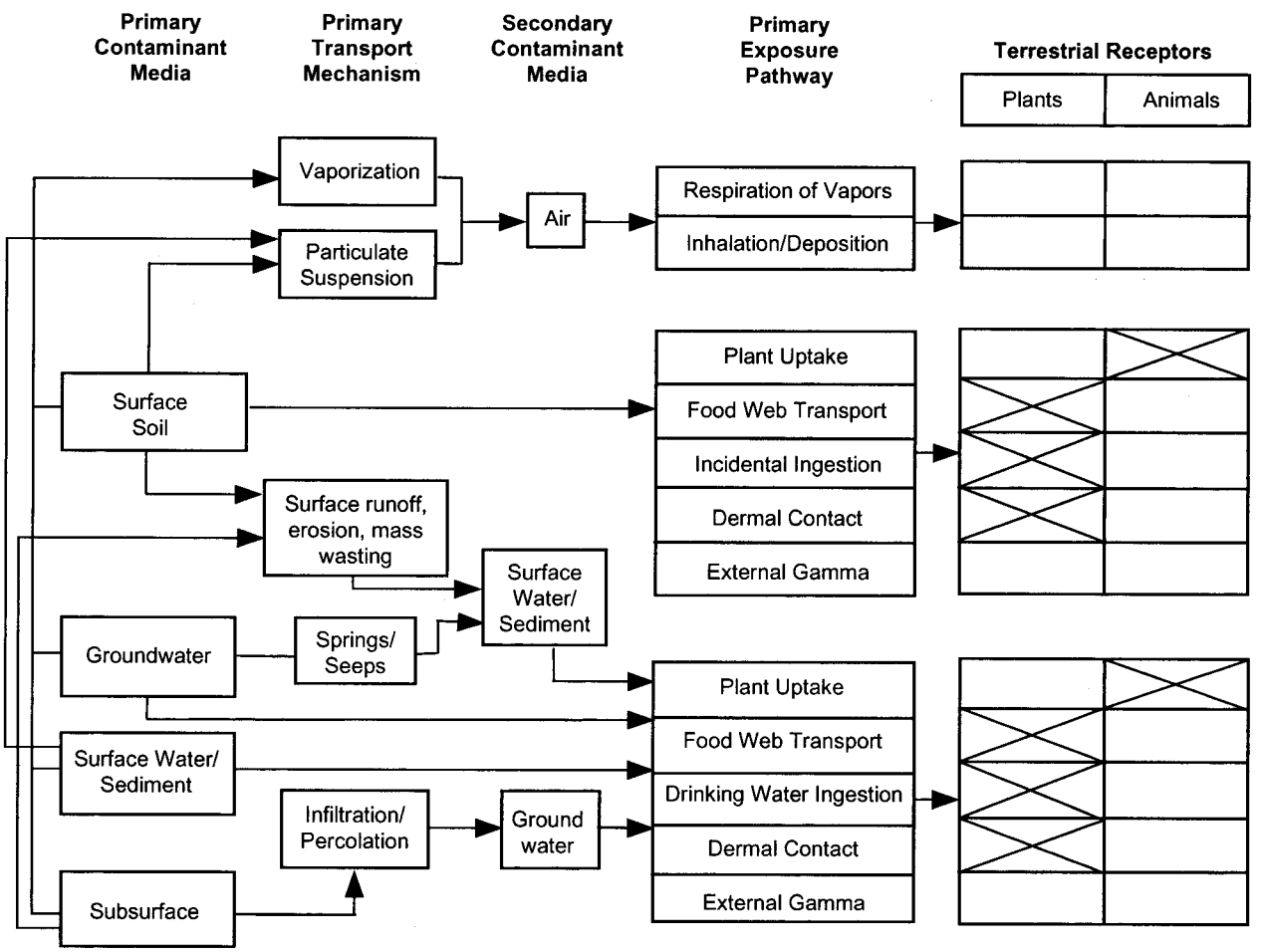
Completion of Part B also includes additional field notes on the site setting and potential ecological receptors to document other site observations relevant to the ecological screening evaluation of the site. Such information may include observations on the variability in the type and density of ecological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

4.1.1.3 Checklist Step 3: Ecological Site Conceptual Model

Part C of the checklist relates to the SCM 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 team. Part C consists of up to 22 questions related to contaminant transport and the potential for exposure of

biota (see Appendix A). Answers to questions in Part C 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.

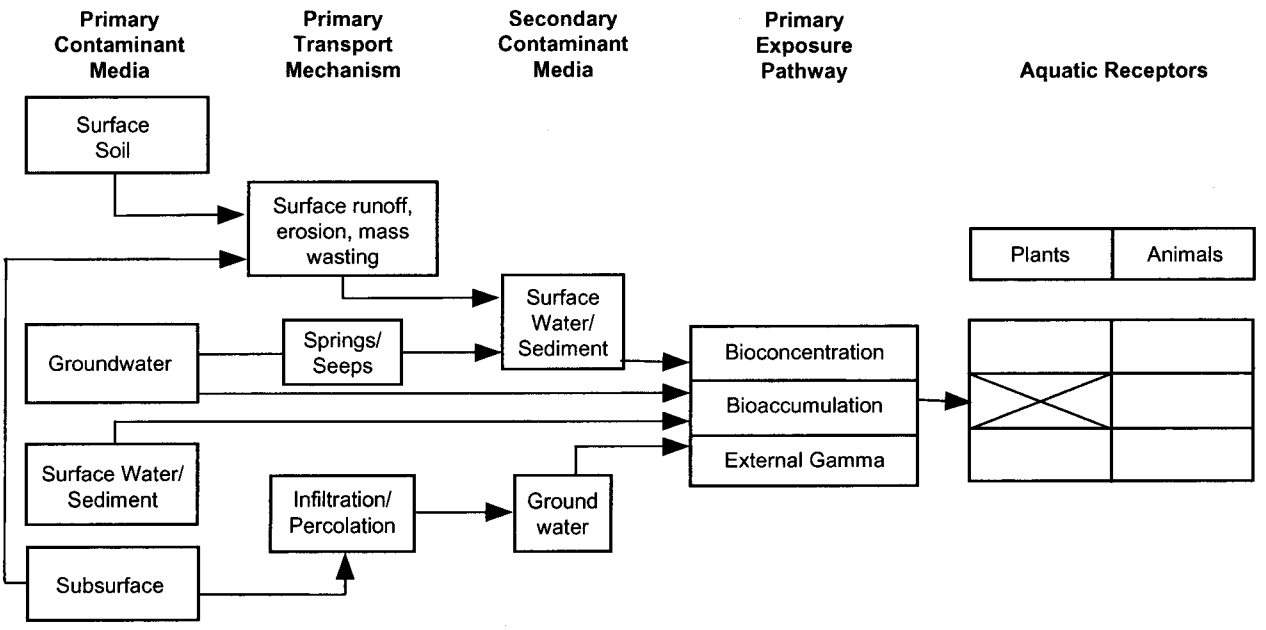
The generic terrestrial receptor site conceptual model is depicted in Figure 4.1-1. The questions provided in the scoping checklist help evaluate the transport and exposure routes to terrestrial receptors. The model evaluates surface soil, groundwater, surface water/sediment, and the subsurface as potentially contaminated media. Surface soil is generally assumed to represent the 0–1.5 m (0–5 ft) interval, but the site-specific scoping should present a rationale and justification for the depth interval assumed to represent surface soils. Figure 4.1-1 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 typically does not have complete exposure pathways to animals.



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

Figure 4.1-1. Terrestrial receptor conceptual exposure and transport model

The generic aquatic receptor site conceptual model is shown in Figure 4.1-2. 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 4.1-2 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, because volatile contaminants are rapidly lost from surface water and sediment, and the 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 may need to be evaluated in both terrestrial and aquatic site conceptual models to ensure that all contaminant exposure pathways are evaluated. There are three possible exposure routes to aquatic receptors from contaminated surface water/sediment: bioconcentration, bioaccumulation, and external gamma. Bioconcentration covers all nontrophic exposure routes, which include respiration and dermal absorption. Bioaccumulation covers only trophic exposure routes, i.e., food web transport.



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

Figure 4.1-2. Aquatic receptor conceptual exposure and transport model

4.2 Screening Evaluation Overview

The purpose of the screening evaluation is to identify COPECs by exposure media, and the outcome of the evaluation is to determine whether contaminants pose a potential unacceptable risk to ecological receptors. The 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 compares site concentration data to final ESLs through the HQ. The HQ is a ratio between a representative exposure concentration and a concentration in a medium corresponding to a potential indicator of effects the ESL. The HI is a sum of HQ values for COPCs with

similar toxicological modes of action. The following equations show how the HQ and HI are calculated, and are based on EPA (1997, 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 the representative exposure concentration for COPC *j* for receptor *i* (units are mg of COPC per kg medium)

$effect_{ij}$ is medium concentration corresponding to an effect level for exposure to COPC *j* for receptor *i* (mg/kg)

HI_i is hazard index for receptor *i* to n COPCs (unitless)

The final ESLs are specific to the medium and include values for soil, sediment, and water, as appropriate. For each medium and each COPC, there is a final ESL. The basis for the final ESL is discussed in Sections 4.3 and 4.4. Simply stated, the final ESL is the minimum applicable ESL value for a COPC in soil, sediment, and water. The final ESL is intended to be protective for all ecological receptors in a given functional group for exposure to that single media. The site exposure concentration and the final ESL are used to calculate the COPC and medium-specific HQ. If the HQ based on the exposure[4] concentration of a COPC exceeds 0.3 for sites with multiple COPCs (or HQ>1 for sites with a single COPC), it is a COPEC. An example result of the single medium HQ calculation table for a site with contaminated soil is provided in Table 4.2-1.

Table 4.2-1
Example Comparison of Site Media Concentrations to ESLs

COPC	Representative Concentration[a] (mg/kg)	Final Soil ESL (mg/kg)	Receptor for Final ESL	Hazard Quotient (unitless result of representative concentration/ESL)
X	100	1000	Shrew	0.1
Y	10	0.1	Deer mouse	100[b]
Z	50	10	Earthworm	5[b]

[a] The representative concentration may be the maximum detected concentration at the site or the 95% UCL of the mean detected concentration at the site.

[b] Shaded values indicate HQs greater than 1.0.

When the HQ generated by comparison of the site exposure concentration is >0.3 for a COPC in a medium, the HQ and the HI based on the ESL for each Laboratory receptor for that COPC should be presented in a table as shown in Table 4.2-2.

[4] Following EPA guidance, 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, the Laboratory may substitute the 95% UCL value for the maximum media concentration (EPA 2002, 85640).

Table 4.2-2
Example Hazard Quotients and Hazard Indices for All Receptors

COPC	Representative Concentration	Omnivorous Kestrel	Carnivorous Kestrel	Herbivorous Robin	Invertivorous Robin	Omnivorous Robin	Deer Mouse	Cottontail	Red fox	Shrew	Earthworm	Plant
Y	100	0.11	0.01	0.07	0.80	0.44	100	3.8	0.14	120	0.06	0.17
Z	5	0.03	0.01	0.12	0.4	0.18	0.9	0.07	0.01	0.16	5	0.07
HI for Receptor		0.14	0.02	0.19	1.2	0.62	101	3.1	0.15	120	5.1	0.24

Note: Shaded cells indicate HQ values greater than 0.3 and HI values greater than 1.0.

In cases where COPCs are detected in multiple media, a second step is required where conservative estimates of exposure to wildlife receptors from multiple exposure media are calculated. All COPCs with potential additive effects are addressed through the calculation of the HQ/HI for individual wildlife receptors. If the HI for a wildlife receptor exceeds 1.0, then those COPCs contributing more than 0.3 to the HI for that receptor are identified as COPECs. Table 4.2-3 provides an example summary of HI /HQ calculations for COPCs detected in sediment and water. Thus, to determine which are COPECs, the COPCs in both media that contribute to the HI for a particular receptor are evaluated. This evaluation shows that COPC "Z" is a COPEC in sediment but not in water (Table 4.2-3).

Table 4.2-3
Example HQ Calculation for Wildlife Receptor "X"

COPC	Medium	Exposure Concentration (mg/kg)	ESL for Wildlife Receptor X (mg/kg)	HQ
X	Sediment	1	100	0.01
Y		10	100	0.1
Z		100	1	100
X	Water	0.1	1	0.1
Y		3	3000	0.005
Z		0.01	1	0.01
				HI =100

Note: Shaded values indicate HQs and HI greater than 1.0.

Derivation of ESLs

Sections 4.3 and 4.4 describe the methods used to derive ESLs for nonradiological and radiological COPCs for soil, sediment, and water, respectively. These methods are based on wildlife exposure models and, for nonradionuclides in aquatic environments, on water and sediment benchmark values from a number of data sources. Calculation of ESLs requires information derived from the primary toxicological literature, toxicologically based numerical standards, exposure parameters for wildlife species, and compilations of ecological risk-based screening values. Although methods for ESL derivation are presented, the ESLs and the supporting information are not included. The Laboratory's ECORISK

Database (LANL 2004, 87386; LANL 2004, 87437, or latest version) provides the necessary information and documentation as well as the ESLs.

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

Ecological Effects of Concern for Screening

In this methodology, effects of ecological concern are considered those that adversely affect reproduction, development, growth, and survival of organisms. Table 4.2-4 shows receptors and some of the ecological effects relevant for screening-level ecological risk assessments. This table is not intended to document all relevant effects, but rather it only shows those effects primarily considered in evaluating the toxicological literature. Other effects may be evaluated and used on a chemical by chemical basis, and the rationale for selecting the relevant effect for each chemical is documented in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).

Table 4.2-4
Relevant Ecological Effects for Ecological Receptors

Receptor	Effect Category		
	Reproduction/Development	Survival	Weight Change/ Size Change
Generic plant	Percent germination, seedling emergence, shoot biomass	Seedling survival	Biomass (root/shoot weight) of mature plant
Earthworm	Number of hatched cocoons, production of juveniles	Adult survival	Body weight
Robin	Eggs produced, hatching success, fledging survival	Adult survival	Body weight
Kestrel	Eggs produced, hatching success, fledging survival	Adult survival	Body weight
Deer mouse	Young produced, juvenile survival	Adult survival	Body weight
Desert cottontail	Young produced, juvenile survival	Adult survival	Body weight
Shrew	Young produced, juvenile survival	Adult survival	Body weight
Red fox	Young produced, juvenile survival	Adult survival	Body weight
Bat	Young produced, juvenile survival	Adult survival	Body weight
Swallow	Eggs produced, hatching success, fledging survival	Adult survival	Body weight

Effects on reproduction include measurable impacts to sexually mature adults from 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 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). Although the life stages of vertebrates may differ for invertebrates and plants, the developmental 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 expected allometric development (e.g., body weight, length, diameter, or other related measures) resulting from chemical exposure. Survival effects include mortality from 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 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 nonradionuclides, 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 effect means that there is a maximum dose or environmental concentration that has no effect. This dose or concentration is of interest for screening-level assessments for nonradionuclides. For radionuclides, no assumed no-effect or threshold dose is assumed, but it is assumed that a dose exists below which the risk is acceptable. This dose is of interest for screening-level assessments for radionuclides.

Most ecological screening assessments for nonradiological 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 TRV. The dose limit for radionuclides is 0.1 rad per day as discussed in Section 4.4 (IAEA 1992, 62802). The 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, 59370).

To determine if wildlife receptors receive COPC doses exceeding the NOAEL (or 0.1 rad/day), a wildlife exposure model is developed and used. This wildlife exposure model considers various dietary and nondietary exposure pathways for wildlife. Modeling is not needed to evaluate exposure to nonwildlife 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 plants, soil invertebrates, and aquatic organisms are exposed by direct contact to, and uptake from, a contaminated medium. For example, root uptake for plants is the primary exposure pathway. If site-specific scoping indicates that foliar uptake may be a primary exposure route for a contaminant, the lack of foliar uptake in the plant toxicity testing is 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 4.2-1 and is based on EPA's general wildlife exposure models (EPA 1993, 59384).

$$E_{total} = E_{oral} + E_{dermal} + E_{respiration}$$ **Equation 4.2-1**

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 inhabiting the soil surface, it is assumed that most contaminant exposure to nonradiological chemicals is through the oral exposure pathway (Sample et al. 1997, 62807). The dermal contact pathway is not typically assessed quantitatively in ecological risk assessments, based on guidance indicating that the ingestion route is most important to terrestrial animals (EPA 1997, 59370). Dermal exposure to wildlife is mitigated by the fur or feathers covering the bodies of most vertebrates. In addition, the incidental consumption of soil during grooming is included in the direct soil ingestion estimates. Soil exposure pathway analysis has shown that dermal pathways contribute a small fraction of the dose obtained orally (EPA 2003, 85643). Therefore, the exposure pathways considered in the development of the ESLs used in screening assessment for this site capture the primary exposures for wildlife receptors. Inhalation exposures may contribute a significant component of exposure to VOCs for species occupying burrows for a significant fraction of the time. Therefore, ESLs have been developed for inhalation exposure for VOCs only for burrowing mammals. For other receptor species and for burrowing mammals, for COPCs other than VOCs, the terrestrial wildlife exposure model for nonradionuclides simplifies to Equation 4.2-2.

$$E_{total} = E_{oral}$$ **Equation 4.2-2**

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, 62783) should be used to evaluate these pathways. The oral exposure model used for terrestrial wildlife is from the *Wildlife Exposure Factors Handbook* (EPA 1993, 59384, Chapter 4) and is provided in Equation 4.2-3:

$$E_{oral} = C_{soil} \cdot I_{soil} \cdot AUF_{soil} + C_{water} \cdot I_{water} \cdot AUF_{water} \cdot (1/d_{water}) + C_{food} \cdot I_{food} \cdot AUF_{food}$$ **Equation 4.2-3**

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)

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 (this fraction is set to one for the initial screening)

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 (this fraction is set to one for the initial screening)

d_{water} is the density of water (1 kg/L)

C_{food} is the concentration of COPC in food (mg/kg dry weight)

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 fraction is set to one for the initial screening)

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, and it considers incidental ingestion of soil and contaminated water. Soil ingestion is calculated from a fraction of the dietary intake of soil (EPA 1993, 59384, Chapter 4). As a protective assumption

appropriate for ecological risk screening, the area use factor (AUF) is set to 1 to indicate the animal receives all its exposure from the contaminated site. An additional conservative assessment is made if 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 little information currently exists 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 information should be included in the site-specific uncertainty analysis.

The above model requires all measures of ingestion (except water) to be on a dry-weight basis. Because the EPA presents most normalized food ingestion rates on a wet weight basis, these dietary constituents must undergo wet-to-dry weight conversions (EPA 1993, 59384). Food intakes rates are provided in units of dry weight, and any conversion factors used in this calculation are also provided. Parameters required for calculations of the general wildlife exposure model, conversions and other elements of the model are provided for terrestrial vertebrate receptors in Table 4.2-5. The information provided in Table 4.2-5 is for the screening receptors adopted by the Laboratory. It is also important to note that exposure parameters provided in Table 4.2-5 represent conservative upper estimates of potential exposure. For example, the value provided for water intake represents the total daily water intake requirement, and the receptors obtain much of that water in their diet, not from surface water sources. More realistic exposure information may be considered in the uncertainty analysis. Information about body weight and inhalation rates, which are not required by Equation 4.2-3, is provided to assist with alternate forms of the wildlife exposure model. For example, the exposure models discussed by Hope (1995, 62783) require these additional parameters.

Table 4.2-5
Measures Required for the Wildlife Exposure Model

Species	Parameter	Value	Units	Reference	Notes
American kestrel	Body weight	0.103	kg	EPA 1993, 59384, p. 2-112	Lowest male average weight was 103 g used to provide more conservative ESL value
	Food intake[a]	0.099	kg/kg/day	EPA 1993, 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, 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, 59384, p. 2-112	Higher of 2 estimated values
	Inhalation rate	0.089	m^3/day	EPA 1993, 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, 59384, p. 2-113	Rounded EPA value to 50% to equally expose receptor to potentially contaminated invertebrates and flesh

Table 4.2-5 (continued)

Species	Parameter	Value	Units	Reference	Notes
American kestrel (continued)	Flesh diet[b]	0.5 (1)	Unitless	EPA 1993, 59384, p. 2-113	Rounded EPA value to 50% to equally expose receptor to potentially contaminated invertebrates and flesh
	Home range	106	ha	EPA 1993, 59384	Average of all home range data for woods, forests, and agricultural areas
American robin	Body weight	0.077	kg	EPA 1993, 59384, p. 2-197	Lowest weight was 77 g used to provide more conservative ESL value
	Food intake[a]	0.35	kg/kg/day	EPA 1993, 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
	Food moisture content	0.77	Proportional	EPA 1993, 59384, pp. 4-13,14	Diet includes invertebrates, plants (fruits), assumed fruit
	Water intake	0.14	L/kg/day	EPA 1993, 59384, p. 2-197	Estimated from allometric equations
	Inhalation rate	0.057	m^3/day	EPA 1993, 59384, p. 3-11	Based on allometric scaling formula for nonpasserine birds
	Fraction soil in diet	0.1	Unitless	Beyer et al. 1994, 62785, Table 1	Used woodcock value, most similar of birds available in source
	Plant diet[c]	0, 0.5, or 1	Unitless	None	Modeled with three diets, herbivore, omnivore, insectivore
	Soil invertebrate diet[c]	1, 0.5, or 0	Unitless	None	Modeled with three diets, herbivore, omnivore, insectivore
	Home range	0.42	ha	EPA 1993, 59384, p. 2-199	Home range data represent average territory size in an open, semi-urban environment
Deer mouse	Body weight	0.020	kg	EPA 1993, 59384, p. 2-295	For females, which have lower body weights and therefore used to provide more conservative ESL value
	Food intake[a]	0.20	kg/kg/day	EPA 1993, 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	EPA 1993, 59384, p. 2-296	moisture content of lab chow used to determine food intake
	Water intake	0.19	L/kg/day	EPA 1993, 59384, p. 2-296	Adult male or female
	Inhalation rate	0.025	m^3/day	EPA 1993, 59384, p. 2-296	Higher of 2 values, estimated
	Fraction soil in diet	0.02	Unitless	Beyer et al. 1994, 62785, Table 1	For white-footed mouse, most closely related of species available in table
	Plant diet	0.5	Unitless	EPA 1993, 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, 59384, p. 2-297	Rounded EPA value to 50% to equally expose receptor to potentially contaminated plants and invertebrates
	Home range	0.077	ha	EPA 1993, 59384, p. 2-298	Average of data from representative environments

Table 4.2-5 (continued)

Species	Parameter	Value	Units	Reference	Notes
Eastern cottontail as a surrogate for desert cottontail	Body weight	0.900	kg	EPA 1993, 59384, p. 2-355	average of range of reported values for desert cottontail
	Food intake[a]	0.093	kg/kg/day	Nagy 1987, 62782	Estimated as 95% UCL using Nagy (1987) allometric scaling formula for herbivores
	Water intake	0.097	L/kg/day	EPA 1993, 59384, p. 2-356	Estimated by EPA from allometric equations
	Inhalation rate	0.63	m^3/day	EPA 1993, 59384, p. 2-356	Estimated by EPA from allometric equations
	Fraction soil in diet	0.024	Unitless	Beyer et al. 1994, 62785, Table 1	For meadow vole, most ecologically similar species of those available in table
	Plant diet	1	Unitless	EPA 1993, 59384, p. 2-356	Assume strict herbivore diet
	Home range	3.1	ha	EPA 1993, 59384, p. 2-357	Average of all home range data for a woodlot and for mixed habitats
Short-tailed shrew as a surrogate for montane shrew	Body weight	0.015	kg	EPA 1993, 59384, p. 2-213	Lowest weight of 15 g used to provide more conservative ESL value
	Food intake[a]	0.198	kg/kg/day	EPA 1993, 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, 59384, p. 4-13	Laboratory feeding study used beef liver
	Water intake	0.223	L/kg/day	EPA 1993, 59384, p. 2-213	One value reported
	Inhalation rate	0.026	m^3/day	EPA 1993, 59384, p. 2-213	One value reported
	Fraction soil in diet	0.1	Unitless	Beyer et al. 1994, 62785, Table 1	Used woodcock
	Soil invertebrate diet	1	Unitless	EPA 1993, 59384, p. 2-214	Assume strict insectivore diet
	Home range	0.39	ha	EPA 1993, 59384, p. 2-214	Reported average of home range for one environment.
Pocket Gopher	Body weight	0.104	kg	Gonzales et al. 2000, 85653	Laboratory-specific minimum measured field value used to provide more conservative ESL value
	Inhalation rate	0.089	m^3/day	EPA 1993, 59384, p. 3-12	Calculated from body weight by equation 3-20 in EPA (1993, 59384)
Red fox	Body weight	3.94	kg	EPA 1993, 59384, p. 2-224	Lowest of 4 mean values used to provide more conservative ESL value
	Food intake[a]	0.045	kg/kg/day	EPA 1993, 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, 59384, p. 4-13	Mean value for mammals and passerine birds
	Water intake	0.086	L/kg/day	EPA 1993, 59384, p. 2-224	Higher of 2 values, estimated

Table 4.2-5 (continued)

Species	Parameter	Value	Units	Reference	Notes
Red fox (continued)	Inhalation rate	2	m^3/day	EPA 1993, 59384, p. 2-224	Higher of 2 values, estimated
	Fraction soil in diet	0.03	Unitless	Beyer et al. 1994, 62785, Table 1	For red fox
	Flesh diet	1	Unitless	EPA 1993, 59384, p. 2-224	Rounded diet to 100% flesh
	Home range	1038	ha	EPA 1993, 59384, p. 2-226	Average of all home range data over a variety of unspecified environments
Violet-green swallow	Body weight	0.0139	kg	Dunning 1993, 62886	Average body weight of females for *Tachycineta thalassina*
	Food intake[a]	0.268	kg/kg/day	Nagy 1987, 62782	Estimated as 95% UCL using Nagy (1987, 62782) allometric scaling formula for passerines
	Water intake	0.242	L/kg/day	EPA 1993, 59384, p. 3-8	Based on allometric scaling formula for birds
	Inhalation rate	0.015	m^3/day	EPA 1993, 59384, p. 3-11	Based on allometric scaling formula for nonpasserine birds
	Invertebrate diet	1	Unitless	None	Assume 100% invertebrate diet
	Home range	0.68	ha	Bowman 2003, 87148	Using general allometric equation of 10^(1.8+log(BW) × 1.06)
Occult little brown myotis bat	Body weight	0.0088	kg	Whitaker 1980, 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, 62782	Estimated as 95% UCL using Nagy (1987, 62782) allometric scaling formula for all mammals
	Food moisture content	0.69	Proportional	EPA 1993, 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, 59384, p. 3-10	Based on allometric scaling formula for mammals
	Inhalation rate	0.012	m^3/day	EPA 1993, 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
	Home range	100	ha	Menzel et al. 2003, 87151	Minimum of 100 ha to 500 ha home range given for southeastern myotis bat

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

[b] Two variants on the American kestrel are used: 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.

[c] Three variants on the American robin are used: one modeled as a strict herbivore, one an omnivore eating 50% plants and 50% invertebrates, and lastly as a strict insectivore.

4.3 ESLs for Nonradiological Chemicals

This section provides an overview of the approach used to develop ESLs for nonradionuclides for soil, burrow air, sediment, and water. Table 4.3-1 summarizes the receptors and diet compositions used in equations for ESL development for each exposure medium.

Table 4.3-1
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
		Montane 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]
		Montane 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
Burrow Air[a]	Mammal	Pocket gopher	Not applicable[c]

[a] Water, sediment, and burrow air ESLs are only used to evaluate whether those media may have significant exposure pathways and COPCs because ESLs for one media do not account for exposure to the same COPC in another media. In all cases where a site has one of these media contaminated, a multimedia assessment is expected (Section 4.3.5).

[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 (Section 4.3.5).

[c] The burrow air ESL applies only to burrowing mammals and only for COPCs that are considered VOCs. The air ESL only reflects the exposure from vapors in the air within the burrow. The mammalian herbivore feeding guild has been modeled with the desert cottontail, so a multimedia exposure assessment to address the potential cumulative effects from soil, water, and air is not possible for this representative species.

4.3.1 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 (LANL 1998, 59730). For the

purposes of ecological risk screening, imported fill or disturbed soils are evaluated similarly to well-developed soils, because they have common exposure and transport pathways. Tuff and bedrock are not evaluated for risk to ecological receptors because tuff and bedrock are not generally considered accessible media to these receptors (LANL 2002, 73791).

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, ESLs 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 exposure calculation (EPA 1993, 59384). The transport and exposure pathways likely to be complete for sites with soil contamination are depicted in Figure 4.3-1. Pathways included in all the ESL calculations are designated as "evaluated" in this figure. The respiration of air vapors pathway is evaluated only for burrow air of terrestrial mammals. For wildlife receptors, ESLs are based on the dietary regimen of the receptor, including consumption of plants, invertebrates, and vertebrate flesh, with some incidental soil ingestion.

Contaminant transport from soil or transport from subsurface media (soil or bedrock) is not evaluated under the soil conceptual model for ESL derivation. However, ESLs combined with transport models may be used to evaluate these pathways. For purposes of wildlife exposure, soil is generally assumed to represent the 0-1.5 m (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 is the lowest receptor-specific soil ESL value available among plants, invertebrates, robin, kestrel, shrew, mouse, cottontail, and fox. The strategy for choosing the final soil ESL from the available ESLs for a COPC is presented in Table 4.3-2.

For plants and invertebrates the soil ESL is the NOEC. Information supporting the selected NOECs is provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version). For wildlife, the soil ESL is the soil concentration of the COPC that results in an exposure dose equal to the NOAEL.

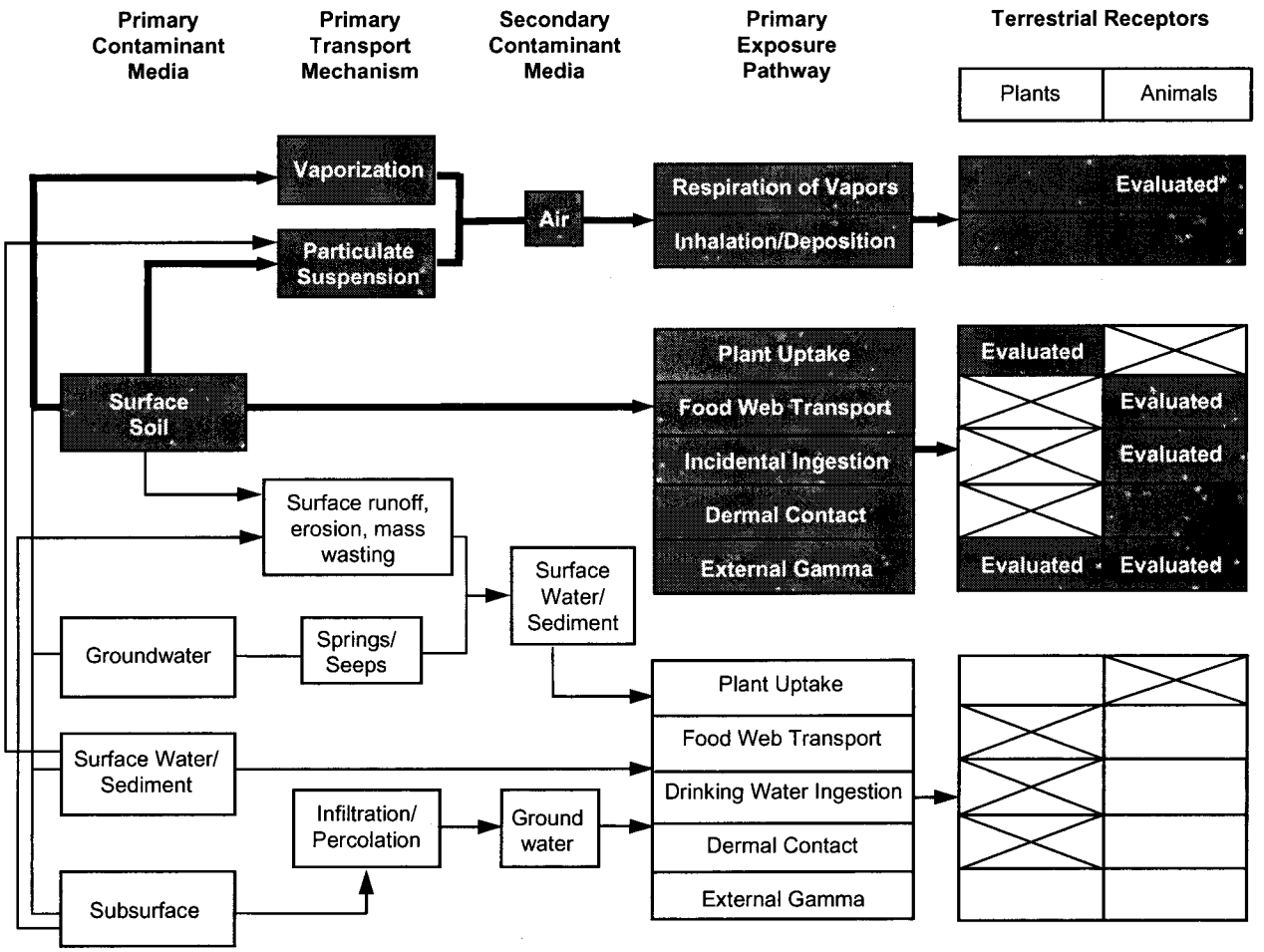
The conversion of soil concentration to dose ingested requires a simple inversion of the wildlife exposure model (with the intake of contaminated water assumed to be zero) discussed below. This inversion is possible because the food intake value may be related to concentration in soil. The general basis for this relationship is shown in Equation 4.3-1.

$$C_{food} = C_{soil} \cdot TF_{food}$$ **Equation 4.3-1**

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 transfer factor from soil to food (mg/kg food dry weight food per mg/kg soil dry weight)



Note: Boxes with Xs indicate incomplete pathways. Open boxes indicate complete pathways. Complete pathways for soil exposure are gray; evaluated pathways are included in the soil ESL calculations.

*For burrowing animals only.

Figure 4.3-1. Ecological conceptual model for soil pathways

Table 4.3-2
Method for Obtaining the Final Nonradionuclide 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). (Note wildlife ESLs are calculated from an exposure model.)
[d] Value = Value available for that COPC.
[e] No value = No value available for that COPC.

Thus, the general wildlife exposure model can be re-written in the following form (Equation 4.3-2), after setting the AUF to 1 and using the relationship between C_{soil} and C_{food} shown in Equation 4.3-1.

$$E_{oral} = C_{soil} \cdot I_{soil} + C_{soil} \cdot TF_{food} \cdot I_{food}$$ **Equation 4.3-2**

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 transfer factor from soil to food (mg/kg food dry weight per mg/kg soil dry weight)

Because the intake of soil can be related to the intake of food, Equation 4.3-2 can be further simplified to Equation 4.3-3. This manner of modeling soil intake rate is conservative because it assumes incidental soil intake in addition to food intake. An alternate model would be based on 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 4.3-3**

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 transfer factor from soil to food (mg/kg food dry weight per mg/kg soil dry weight)

Solving Equation 4.3-3 for the COPC and wildlife receptor-specific ESL yields Equation 4.3-4:

$$ESL_{ij} = \frac{NOAEL_{ij}}{I_i \cdot [fs_i + TF_j]}$$ **Equation 4.3-4**

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)

I_i is the normalized daily dietary ingestion rate for wildlife receptor *i* (kg/kg/day)

fs_i is the fraction of soil ingested by wildlife receptor *i*, expressed as a fraction of the dietary intake

TF_j is a transfer factor from soil to food for COPC *i* (mg/kg dry weight food per mg/kg dry weight soil)

Equation 4.3-4 assumes a single food type is ingested and must be specific for herbivores, omnivores, invertevores, and carnivores. Equations for these functional groups of wildlife receptors are shown in Equations 4.3-5 through 4.3-8.

$$ESL_{ij} = \frac{NOAEL_{ij}}{I_i \cdot [fs_i + TF_{plant,j}]}$$ **Equation 4.3-5**

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)

I_i is the normalized daily dietary ingestion rate for herbivore *i* (kg/kg/day])

fs_i is the fraction of soil ingested by herbivore *i*, expressed as a fraction of the dietary intake

TF_{plantj} is a transfer factor from soil to plants for COPC *i* (mg/kg dry plant weight per mg/kg dry weight soil)

$$ESL_{ij} = \frac{NOAEL_{ij}}{I_i \cdot [fs_i + fp_i \cdot TF_{plant,j} + fi_i \cdot TF_{invert,j}]}$$ **Equation 4.3-6**

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)

I_i is the normalized daily dietary ingestion rate for omnivore *i* (kg/kg/day)

fs_i is the fraction of soil ingested by omnivore *i*, expressed as a fraction of the dietary intake

fp_i is the fraction of plants in diet for omnivore *i*

$TF_{plant,i}$ is a transfer factor from soil to plants for COPC *j* (mg/kg dry plant weight per mg/kg dry weight soil)

fi_i is the fraction of invertebrates or flesh in diet for omnivore *i*

$TF_{invert,j}$ is a transfer factor from soil to invertebrates (mg/kg dry insect weight per mg/kg soil dry weight) or soil to flesh (mg/kg dry flesh weight per mg/kg soil dry weight for COPC *j*

$$ESL_{ij} = \frac{NOAEL_{ij}}{I_i \cdot [fs_i + TF_{invert,j}]}$$ **Equation 4.3-7**

Where ESL_{ij} is the soil ESL for invertevore *i* and COPC *j* (mg/kg)

$NOAEL_{ij}$ is the NOAEL for invertevore *i* and COPC *j* (mg/kg/day)

I_i is the normalized daily dietary ingestion rate for invertevore *i* (kg/kg/day)

fs_i is the fraction of soil ingested by invertevore *i*, expressed as a fraction of the dietary intake

$TF_{invert,j}$ is a transfer factor from soil to invertebrates for COPC *j* (mg/kg dry invertebrate weight per mg/kg soil dry weight)

$$ESL_{ij} = \frac{NOAEL_{ij}}{I_i \cdot [fs_i + TF_{flesh,j}]}$$ **Equation 4.3-8**

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)

I_i is the normalized daily dietary ingestion rate for carnivore *i* (kg/kg/day)

fs_i is the fraction of soil ingested by carnivore *i*, expressed as a fraction of the dietary intake

$TF_{flesh,j}$ is a transfer factor from soil to flesh for COPC *j* (mg/kg dry flesh weight per mg/kg soil dry weight)

The wildlife ESL model (Equation 4.3-4 and the functional group-specific Equations 4.3-5 through 4.3-8) shows the ESL as proportional to the NOAEL. Thus, larger values of the NOAEL lead to larger ESL values, which indicates 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. A receptor with higher exposure will have lower ESLs for the same NOAEL value as a receptor with lower exposure. Table 4.3-3 summarizes the input variables for the wildlife exposure models and indicates the general sources used for these variables.

Uncertainties associated with soil ESLs fall into two main categories. The first group is associated with COPCs, including toxicity and bioavailability (or transfer factors between soil and food). The second group 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, which 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 for each COPC to ensure the screening evaluation is protective and inclusive of all COPCs. Soil ESLs only screen individual COPCs, and Section 4.3.5, Multimedia Screening Calculations, describes how multiple exposure media are evaluated.

[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.

Table 4.3-3
Summary of Variables Used in the Nonradionuclide Wildlife ESL Models

Variable	Source
NOAEL	Receptor and COPC specific NOAEL values are obtained from reviewing primary literature on toxicity to ecological receptors. Values for specific receptors and COPCs are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).
fs	Receptor-specific values are provided in Table 4.2-5.
I	Body weight normalized food intake for wildlife receptors (see values provided in Table 4.2-5). Body weight is an implicit component of this variable. For this reason Table 4.2-5 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 4.2-5.
fi	Fraction of invertebrates in diet is provided in Table 4.2-5.
TF_{plant}	The transfer from soil to plants is a COPC-specific value derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version). 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 derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).
TF_{flesh}	The transfer from soil to flesh is a COPC-specific value that is derived from three other factors (LANL 2002, 72641). The first factor is a fresh weight feed to muscle transfer factor (TF_{beef}) derived from studies of beef cattle. The second factor is the maximum of either the moisture content (MC) adjusted dry weight TF_{plant} or the moisture content adjusted dry weight TF_{invert}. This transfer factor term represents the prey with the most contaminated diet. The two transfer factors are multiplied by a food ingestion rate. This rate is based on a composite prey species value developed from the four potential mammalian prey species (robin, deer mouse, cottontail, and shrew). The highest food and soil intake rates among these four potential prey species were used to represent the composite prey species in the equation below: Thus, $TF_{flesh} = TF_{beef} \cdot (I_{food} \cdot \text{maximum of } (TF_{plant} \cdot (1\text{-}MC_{plant}), TF_{invert} \cdot (1\text{-}MC_{invert})) + I_{soil})/(1\text{-}MC_{flesh})$ Values for specific COPCs are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).

One important factor not considered in developing wildlife ESLs is the potential for biomagnification of COPCs in higher trophic levels. The carnivore is modeled as eating herbivore or 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 complex and beyond the realm of screening. The potential for biomagnification for top carnivores depends on factors relating to the spatial distribution of the COPC relative to the distribution of prey and the biological retention time within the prey. This uncertainty should be 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, 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 (e.g., Newman et al. 1994, 62788). The energy value of the food consumed by the animal also shows a relationship to food intake (Nagy 1987, 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, 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 nonlinear TF relationships, is viewed as unnecessary for the purposes of screening.

4.3.2 Burrow Air ESLs (Vapor-Phase Contaminants Only)

Quantitative evaluations of ecological risk do not typically include the inhalation pathway because ingestion-related exposure is relatively more important for most chemicals. However, burrow air exposure is potentially a significant exposure pathway for burrowing mammals at some Laboratory SWMUs and AOCs. These SWMUs and AOCs are typically colonized by pocket gophers (*Thomomys bottae*) and other ecological receptors exposed to vapor-phase contaminants in burrows. Simple fate and transport models indicate that vapor-phase contaminants are at much lower concentrations in surface air (Markwiese et al. 2003, 87149), and, therefore, quantitative evaluation of surface air inhalation as a pathway to ecological receptors is not warranted. Vapor-phase contaminants are not prone to bioaccumulation, so the pathways considered for burrow air ESLs are limited to inhalation or respiration of vapors. The pocket gopher is designated as the representative receptor for burrowing mammals. The best estimate of burrow air concentrations is obtained by using soil pore-gas data collected from depths corresponding to those occupied by pocket gophers. Exposure parameters for the pocket gopher are provided in Table 4.3-4.

Table 4.3-4
Measures Required for the Pocket Gopher Air Exposure Model

Species	Parameter	Value	Units	Reference	Notes
Pocket Gopher	Body weight	0.104	kg	Gonzales et al. 2000, 85653	Laboratory-specific minimum measured field value
	Inhalation rate	0.089	m^3/day	EPA 1993, 59384, p. 3-12	Calculated from BW by Equation 3-20 in EPA (1993, 59384)

The gopher's inhalation rate (IR) is based on body weight (BW) according to the allometric equation from Stahl (1967, 63119) shown in Equation 4.3-9:

$$IR = 0.5458 \cdot BW^{0.80}$$ **Equation 4.3-9**

It is assumed that the gopher stays in its burrow 100% of the time; therefore, the exposure through air is described by Equation 4.3-10:

$$E_{air} = C_{air} \cdot I_{air}$$ **Equation 4.3-10**

Where E_{air} is the estimated inhalation daily dose for a COPC (mg/kg/day)

C_{air} is the concentration of chemical constituent x in air inside the burrow (mg/m^3)

I_{air} is the normalized daily air inhalation rate (m^3/kg/day)

Therefore, the ESL can be expressed as shown in Equation 4.3-11:

$$ESL_j = \frac{NOAEL_j \cdot BW}{IR}$$ **Equation 4.3-11**

Where ESL_j is the soil ESL for burrow animal and COPC *j* (mg/m^3)

$NOAEL_j$ is the NOAEL for burrow animal inhalation and COPC *j* (mg/kg/day)

BW is the body weight for burrow animal (kg)

IR is the normalized daily inhalation rate for the pocket gopher (m^3/day)

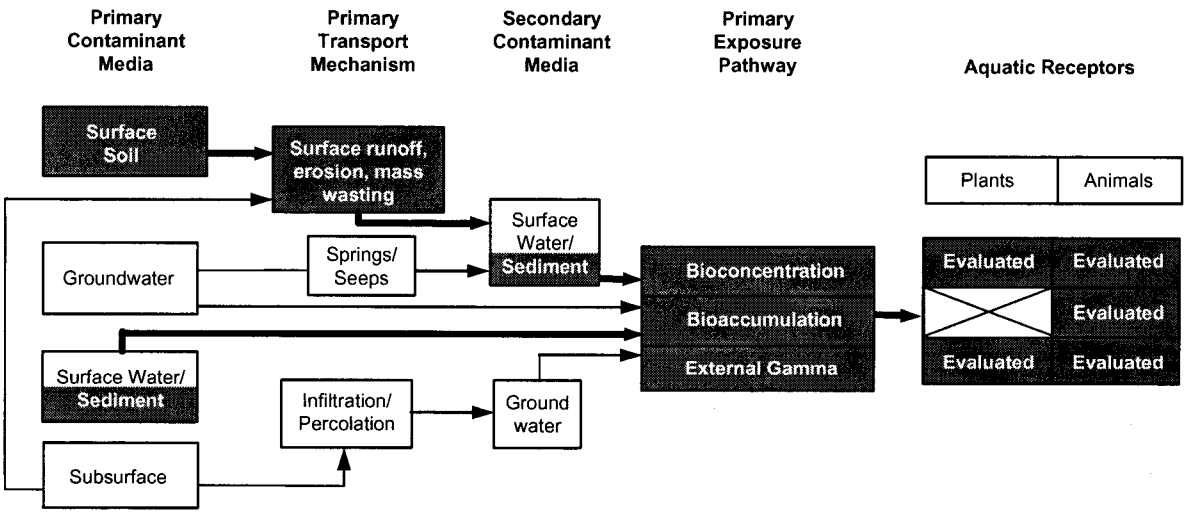
4.3.3 Sediment ESLs

Geomorphologists define sediments as young alluvium occurring within or near stream channels, which would be generally classified as A or C generic horizons in soil nomenclature (LANL 1998, 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 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 applicable 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 is 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.

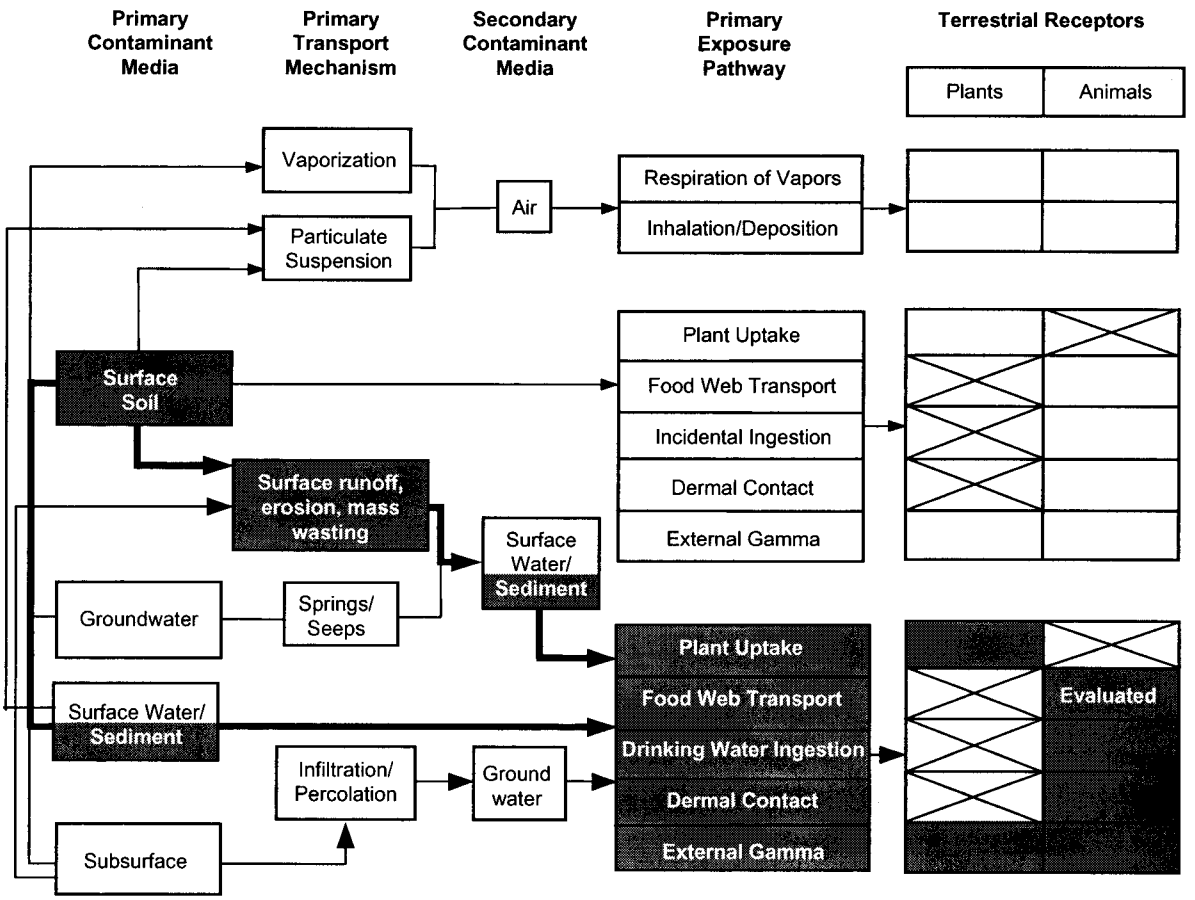
Sediment ESLs for the protection of aquatic life are derived from information on direct effects of contaminated sediments on aquatic organisms. Only limited modeling is needed to develop sediment ESLs. Modeling is used to evaluate potential effects of contaminated sediments on terrestrial receptors through accumulation of COPCs in emergent insects. Thus, sediment ESLs incorporate bioaccumulation issues and trophic transfer concerns.

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 4.3-2 and 4.3-3). Of primary concern are the first three transport mechanisms, which are included in Figures 4.3-2 and 4.3-3. Rare instances where mass wasting or wind-blown soils may significantly influence the sediment load of a water body are identified during site-specific problem scoping. With the limited water resources in the region, the primary focus should be on pathways of sediment transport from areas adjacent to or contiguous with permanent or seasonally intermittent surface water resources.



Note: Boxes with Xs indicate incomplete pathways. Open boxes indicate complete pathways. Complete pathways for sediment exposure to aquatic receptors are gray; evaluated pathways are included in the sediment ESL calculations.

Figure 4.3-2. Aquatic conceptual model for sediment pathways



Note: Boxes with Xs indicate incomplete pathways. Open boxes indicate complete pathways. Complete pathways for sediment exposure to terrestrial receptors are gray; evaluated pathways are included in the sediment ESL calculations.

Figure 4.3-3. Terrestrial conceptual model for sediment pathways (to account for bioaccumulation concerns)

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, 62902) cites several reasons for the requirement of sediment-quality criteria (SQC) and sediment quality benchmarks (SQBs), including

- various toxic contaminants found only in trace amounts in the water column that accumulate in sediments to elevated levels;
- sediments that serve as both a reservoir and a source of contaminants to the water column;
- sediments that integrate contaminant concentrations over time, whereas water column contaminant concentrations are much more variable and dynamic;
- sediment contaminants, in addition to water column contaminants, that affect benthic and other sediment-associated organisms; and
- sediments that 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 conform with those proposed by the EPA for developing ecotoxicity thresholds (EPA 1996, 62792). Methods for screening sediments are based on the assumption that aquatic organisms are generally exposed to the greatest fraction of contamination by means of direct media contact, i.e., continuous bodily contact with sediments. Thus, the exposure pathways for aquatic receptors (using EPA methods) include bioconcentration and, for radionuclides only, external gamma exposure (Figure 4.3-2). Aquatic ecological screening pertains to receptors that are generally associated with benthic surfaces. Generally, to be protective of aquatic plant and animal species, 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.

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 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 4.3-3. This conceptual model indicates that several exposure pathways are complete, but only the food web transport pathway is evaluated, as other pathways make only minor contributions. Additionally, the uptake of COPCs from sediments is much more significant for aquatic plants and animals in direct contact with the sediment medium, which is covered by the sediment pathways model (Figure 4.3-2) and screening methods.

Sediment-screening benchmarks (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 different measurement endpoints. Values from studies using freshwater sediments have been assigned highest priority, and in general only values endorsed by various EPA entities have been considered. Based upon EPA and other methods (EPA 1995, 62787; EPA 1996, 62792; EPA 1996, 62877; EPA 1996, 62794; Jones et al. 1997, 62789), the Laboratory ranks SSBs in the following order:

1. SQCs calculated from national ambient water-quality criteria (NAWQC) or from Great Lakes Tier I water-quality criteria (WQC) (EPA 1995, 62787) according to EPA guidance (EPA 1996, 62792);
2. SQBs calculated from Great Lakes Tier II WQC (EPA 1995, 62787) according to EPA guidance (EPA 1996, 62792);

3. Sediment Effects Concentrations (SECs) derived from EPA methods described below (EPA 1996, 62877);
4. EPA Region IV screening values (EPA 1996, 62794); and
5. Jones et al. (1997, 62789).

Sediment is screened using SSBs in the above rank order. For example, if an SQC can be calculated from NAWQC or Tier I WQC (EPA 1995, 62787), then this SQC becomes the preferred criterion; if an SQC can be calculated, then it is used, and so on. Table 4.3-5 shows how SSB values and wildlife ESLs are used to derive a final sediment ESL. The approach outlined in Table 4.3-5 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 not available or cannot be calculated for a given COPC, then the COPC is retained as a COPEC and discussed in the uncertainty analysis. 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.

Table 4.3-5
Method for Obtaining the Final Nonradionuclide 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.

The following paragraphs summarize the salient information about selecting SSBs and describe the methods used to calculate the wildlife ESL. Jones et al. (1997, 62789) summarize the SSB methods and expand upon some of the definitions used here. In addition, the discussion uses a question-and-answer approach to aid in selecting SSBs for COPCs. The sources cited in this discussion provide more comprehensive information on the calculations used to determine the various criteria and benchmarks.

SSB1: Sediment-Quality Criteria

- Is the COPC a nonionic organic compound?
- Do NAWQC 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, 62792). SQCs have been proposed by EPA's Office of Water (OW) for acenaphthene, dieldrin, endrin, fluoranthene, and phenanthrene (EPA 1996, 62792). A number of alternatives/benchmarks have been proposed by EPA (1996, 62792) and Jones et al. (1997, 62789). All these values were derived using the EqP method, which quantifies the hydrophobicity of the chemical by using the octanol/water partition coefficient (K_{ow}) and determines the sorption capacity of the sediment by the mass fraction of organic carbon (f_{oc}) of the sediment. It is important to note that the EqP method is appropriate for *nonionic organic compounds only* (EPA 1996, 62794). The relationship between K_{ow} and the sediment organic carbon partitioning coefficient, K_{oc}, is described by Equation 4.3-12 (Di Toro 1985, 62876):

$$\log_{10} K_{oc} = 0.00028 + 0.983 \log_{10} K_{ow}$$ **Equation 4.3-12**

The EqP method assumes that pore water is in equilibrium with sediment and that pore water must meet water-quality standards to be considered nontoxic (O'Connor et al. 1998, 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 the colloids or suspended solids available for direct ingestion by wildlife are not considered, then only the uncomplexed chemical in pore water, in equilibrium with the organic carbon fraction of the sediment, is bioavailable to aquatic organisms. Jones et al. (1997, 62789) stated "for highly hydrophobic chemicals and where there is significant dissolved organic carbon complexing, the solid-phase chemical concentration gives a more direct estimate of the bioavailable pore water COPC concentration than do the pore water concentrations."

The EqP approach requires four major assumptions (Jones et al. 1997, 62789):

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

EPA has concluded that the sensitivities of benthic organisms are sufficiently similar to those of water column species to tentatively permit using WQBs to derive sediment-quality benchmarks (Jones et al. 1997, 62789). Because of complexities associated with metal binding in sediments (e.g., metal binding sites other than organic carbon such as clay surfaces), the EqP approach is inappropriate for use with metals.

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

$$SQC = f_{oc} \cdot K_{oc} \cdot (FCV \text{ or } CCC)$$ **Equation 4.3-13**

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 NAWQC (Tier I toxicity values). See Section 4.4.3, Water ESLs, and EPA (1995, 62787) for details on the calculation of FCVs.

CCC (µg/L) is the "criterion continuous concentration." See EPA (1998, 62791) for recommended CCC values and EPA (1995, 62787) for details on the calculation of CCCs.

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

Using the above relationships, SQCs can be derived for any number of nonionic organic compounds. This method uses 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:

$$SSB1 = \left(f_{oc[site\ specific]} \Big/ f_{oc[normalized]} \right) \cdot SQC \qquad \textbf{Equation 4.3-14}$$

Using 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 because of 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 compounds?
- Do Tier II level WQC exist for the COPC?
- If **yes**, for both of the above, follow the directions outlined below for generating SQBs in the calculation of ESLs.
- If **no**, to either of the above, go to the SSB3 section.

When Tier I NAWQCs are unavailable for calculating SQCs, SQBs are generated using Tier II secondary chronic values (SCVs) for water (see Section 4.3.4, Water ESLs). SQBs are calculated using the identical mathematical relationships of SQCs with the substitution of SCV for the FCV/CCC in Equation 4.3-13. 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, 62792). Since both SQCs and SQBs are directly dependent upon K_{ow} values, reliable sources for this information are necessary (e.g., EPA 1995, 62806; MacKay et al. 1992–1997, 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, 62787).

SSB2 values are calculated as follows:

$$SSB2 = \left(f_{oc[site\ specific]} \Big/ f_{oc[normalized]} \right) \cdot SQB \qquad \textbf{Equation 4.3-15}$$

Jones et al. (1997, 62789) provide some SQBs for ionic organic compounds. As Jones et al. (1997, 62789) indicate, ionic organic compounds have not been well studied for their equilibrium partitioning properties in the water-sediment interface. The Oak Ridge National Laboratory (ORNL) benchmarks (Jones et al. 1997, 62789) are probably conservative, as the fraction of ionic organic compounds sorbed to the organic carbon surfaces is likely to be greater than that for nonionic substances. Other factors may also affect the sorption capacity of sediment for ionic compounds, including pH (Jafvert 1990, 62904).

SSB3: Great Lakes Water-Quality Criteria Sediment Effects Concentrations

- Is the COPC a nonionic organic compound 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 other SSB resources 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, 62877). The 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 threshold effects level (TEL) and probable effects level (PEL) methods, and the State of Washington's apparent effects threshold (AET). The calculation of SECs is considered more robust than using a single benchmark, because multiple benchmarks are used to derive a single SEC value. The Laboratory generally recommends using TELs or ERLs as SECs, 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 as an SSB. The following definitions apply (EPA 1996, 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; and
- AET—the sediment chemical concentration above which statistically significant biological effects always occur.

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 of many intact aquatic systems. Because they are ubiquitious and part of 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 Great Lakes System Stakeholders, and has been found to be adequate to serve as a nominal model for freshwater systems nationwide (Jones et al. 1997, 62789). Jones et al. (1997, 62789)

recommend SECs to be adopted as SSBs for organic COPCs not covered by the SQC or SQB methods, and for metals.

Other SSB Resources

If data in the above resources are lacking for a particular chemical, then the Laboratory considers EPA Region IV's sediment screening values (EPA 1996, 62794) as a source for SSBs. These values are based on sediment toxicity work performed on marine sediments (e.g., Long et al. 1995, 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, 62794; EPA 1996, 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, 62796). In many cases, because the practical quantitation limit (PQL) for identifying organic compounds was greater than the benchmark values, the benchmark value defaults to the PQL. In the case where the PQL is used as a default value, a justification should be provided for using a PQL as a benchmark value. 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.

In the absence of any valid SSB resources, any available information on a given constituent (e.g., from primary literature sources) should be used. The biotic system being evaluated should be considered, as well as the range of concentrations over which there may be information on no effects or observed effects. In context with one another, the biotic system being evaluated and relative effects ranges considered may provide insight into the most appropriate SSB. The Ontario Ministry of the Environment sediment quality guidelines (Persaud et al. 1993, 62875) is an additional resource. Jones et al. (1997, 62789) also provide references for other resources. SSBs adopted from sources other than those recommended in SSB1-SSB3 should be discussed in the 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 4.3-7, which is the insectivore soil ESL model described in Section 4.3.1, Soil ESLs. The model shown in Equation 4.3-16 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 in this model are the bat and the swallow, and the exposure information for these receptors is provided in Table 4.2-5. Transfer to higher level carnivores is not accounted for by these ESLs and should be addressed in the uncertainty analysis.

$$ESL_{ij} = \frac{NOAEL_{ij}}{I_i \cdot TF_{invert,j}}$$ **Equation 4.3-16**

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])

$TF_{invert,i}$ is a transfer factor from sediment to invertebrate for COPC j (pCi/g fresh invertebrate weight per pCi/g dry soil weight)

Summary of Sediment ESLs Derivation

Sediment ESLs may be derived from a variety of sources (Table 4.3-6), and more than one SSB may be available for any given constituent (see Table 4.3-5). 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 to ensure that bioaccumulation concerns are addressed by the final ESL.

Table 4.3-6
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, 62792] using Tier I AWQC from EPA [1998, 62791] or EPA [1995, 62787]) 2. SQB (EPA [1996, 62792] a using Tier II SCVs from EPA [1995, 62787]) 3. SEC (EPA 1996, 62877) 4. EPA Region IV (EPA 1996, 62794) 5. Other (e.g., Jones et al. 1997, 62789; Long et al. 1995, 62793)	Equation 4.3-16 with NOAEL—primary toxicity literature for each COPC TF—based on site-specific information, literature values, or simple models I—as provided in Table 4.2-5
Inorganic chemicals	The following are used in order of preference: 1. SEC (EPA 1996, 62877) 2. EPA Region IV (EPA 1996, 62794) 3. Other (e.g., Jones et al. 1997, 62789; Long et al. 1995, 62793)	

The main parameters introduced in Section 4.3.3 are summarized in Table 4.3-7. Jones et al. (1997, 62789) discuss uncertainties and limitations associated with the basis and generation of sediment ESLs. Of particular importance to the Laboratory is the fraction of organic carbon in Laboratory sediments, an environmentally variable parameter that can be measured directly on a site-specific basis.

4.3.4 Water ESLs

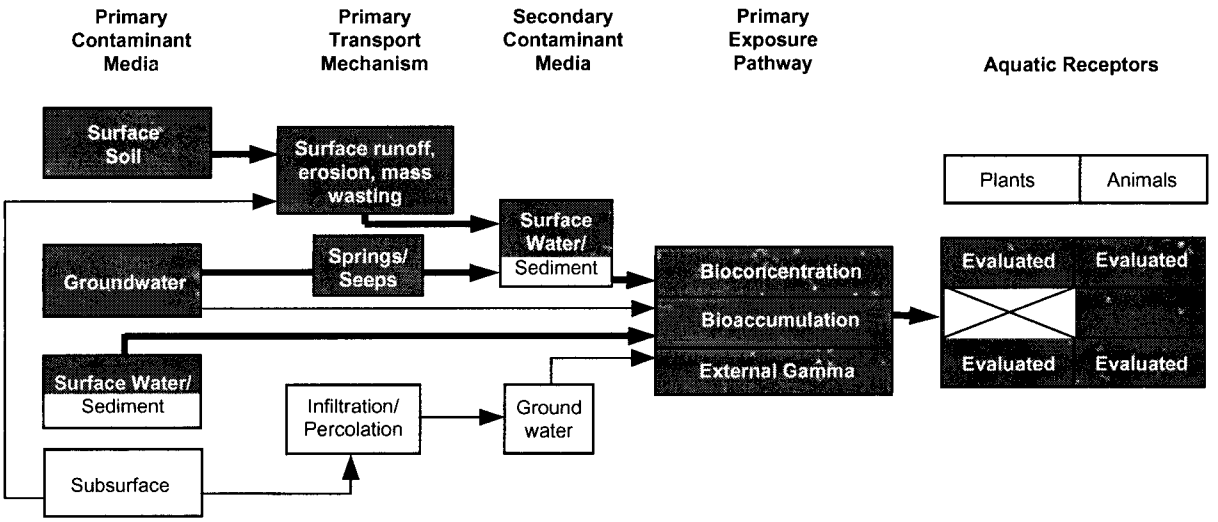
Water of potential concern to ecological receptors at the Laboratory includes surface water and shallow groundwater. For the purposes of ecological screening, only exposure pathways related to surface water and groundwater that emerges at the surface are evaluated. For those sites where exposure to shallow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

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

Table 4.3-7
Summary of Variables Used in the Sediment ESL Models

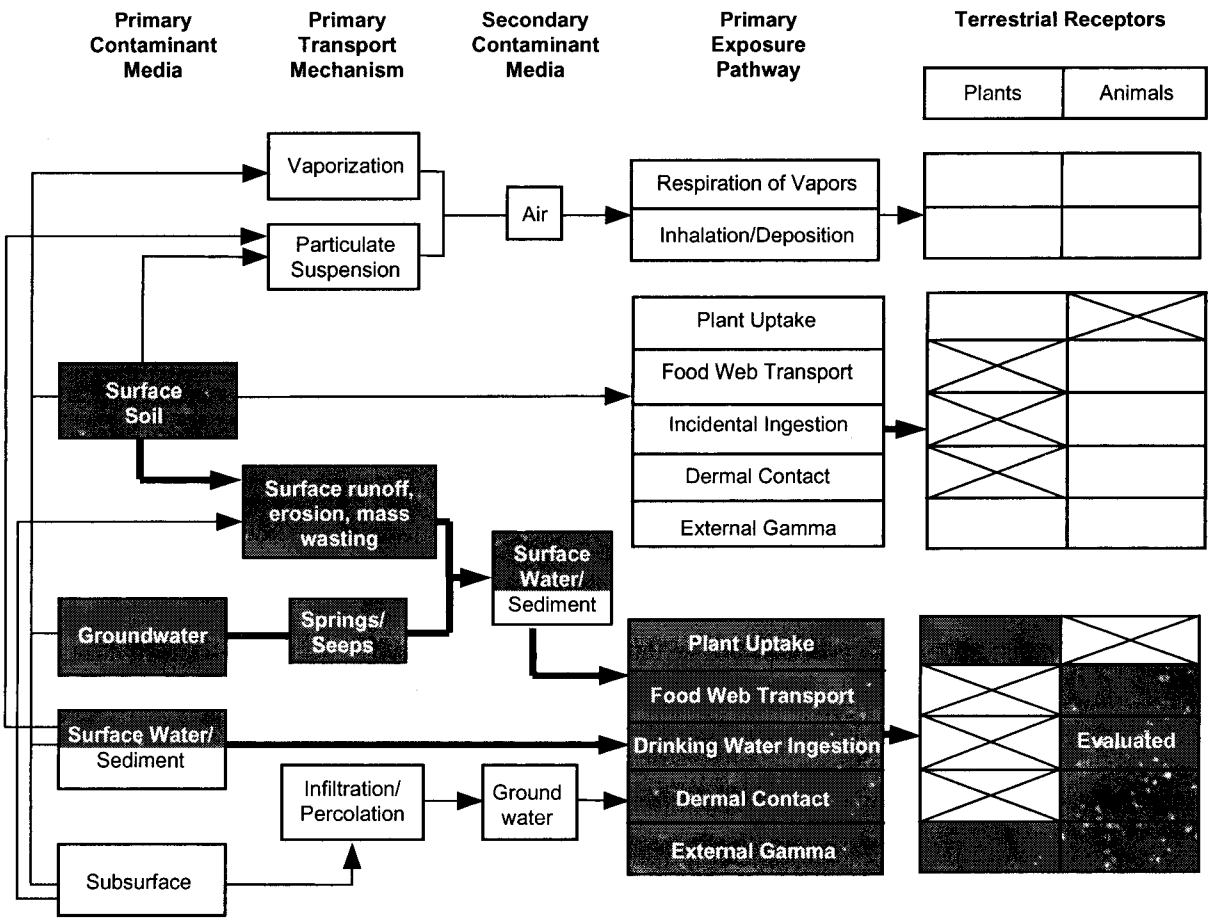
Variable	Source
SSB	Sediment screening benchmark (µg/kg). SSBs are generic and may be derived from variable sources (e.g., SQC from EPA [1996, 62792] methods) ranked according to criteria outlined above.
SQC	Sediment quality criteria (µg/kg). SQCs are based on NAWQC or their equivalent (Tier I NAWQC). SQCs are intended to prevent significant toxic effects in most chronic exposures.
SQB	Sediment quality benchmark (µg/kg). SQBs are based on Tier II WQC. 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 (mg/L octanol) / (mg/L water). This value may 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 value 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 (LANL 2004, 87386; LANL 2004, 87437, or latest version). 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 (LANL 2004, 87386; LANL 2004, 87437, or latest version).
I_i	Body weight normalized food intake for wildlife receptors, see values provided in Table 4.2-5. Body weight is an implicit component of this variable. For this reason Table 4.2-5 provides body weight for each receptor. 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 derived from site-specific studies, other empirical literature studies, and models. Values for specific COPCs are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).

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 associated with benthic surfaces and the free water column of both lentic and lotic systems. To be broadly protective of aquatic plant and animal species, EPA has developed methods (EPA 1995, 62787; EPA 1996, 62792) intended to protect 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 4.3-4. To evaluate potential effects of contaminated water on terrestrial receptors, a wildlife exposure model is developed (Figure 4.3-5). The terrestrial conceptual model is based on exposure to contaminated drinking water. Inclusion of this model addresses bioaccumulation concerns not addressed directly by the EPA methods.



Note: Boxes with Xs indicate incomplete pathways. Open boxes indicate complete pathways. Complete pathways for water exposure to aquatic receptors are gray; evaluated pathways are included in the water ESL calculations.

Figure 4.3-4. Aquatic conceptual model for water pathways



Note: Boxes with Xs indicate incomplete pathways. Open boxes indicate complete pathways. Complete pathways for water exposure to terrestrial receptors are gray; evaluated pathways are included in the terrestrial ESL calculations.

Figure 4.3-5. Terrestrial conceptual model for water pathways

The consideration of impacts from waterborne contamination to aquatic receptors requires the evaluation of a number of WQC, which come from a variety of sources, all based upon toxicological information from primary studies; only values endorsed by various EPA-approved sources have been considered in the SLERA. These criteria differ in the methods for their development and/or in the rigor of their development. Consequently, WQC 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, WQCs are chosen in the following order:

1. Chronic NAWQC set forth by EPA guidance (EPA 2002, 82653) and Sections 20.6.4.12 and 20.6.4.900 of the New Mexico Administrative Code (NMED 2002, 87103);
2. Great Lakes methodology Tier I final chronic value (FCV) (EPA 1995, 62787);
3. Great Lakes methodology Tier II chronic value (CV) (EPA 1995, 62787); and
4. Other sources, including Suter and Tsao (1996, 59838) and Suter (1996, 62805).

Justification for selecting the above order is provided in greater detail in various EPA documents (EPA 1995, 62787; EPA 1996, 62792; EPA 2002, 82653) and in Sections 20.6.4.12 and 20.6.4.900 of the New Mexico Administrative Code (NMED 2002, 87103). If more than one WQC exists for a given COPC, then the information implicit in these values is discussed in the uncertainty analysis.

Water is screened utilizing WQCs in the rank order presented above. For example, if New Mexico and/or EPA chronic ambient WQC are available for a given constituent, then the New Mexico and/or EPA value respectively is selected as the most relevant WQC for screening. If there is no New Mexico and/or EPA ambient WQC, but a Tier I WQC is calculable (EPA 1995, 62787), then the Tier 1 WQC becomes the preferred criterion. Lastly, if no Tier I WQC is calculable, then a Tier II value will be used. All of the acceptable WQC and wildlife ESLs that are available and applicable to the site-specific conditions may be evaluated in the uncertainty analysis. Table 4.3-8 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 is retained as a COPEC and discussed in the uncertainty section.

Table 4.3-8
Method for Obtaining Final Nonradionuclide Water ESL

COPC	WQC (µg/L)	Minimum Wildlife ESL* (µg/L)	Final Water ESL (µg/L)
T	Chronic NAWQC (WQC1)	Value	Minimum of WQC1 and wildlife ESL
U	Tier I value (WQC2)	Value	Minimum of WQC2 and wildlife ESL
V	Tier II value (WQC3)	Value	Minimum of WQC3 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

*Based on wildlife exposure calculation.

The following paragraphs describe the methods used to select water ESLs, from the most to the least preferable. The discussion uses a question-and-answer approach to aid in calculating water ESLs. The

sources cited in the discussion provide more comprehensive information on the calculations used to determine the various criteria and benchmarks. Suter (1996, 62805) presents a list of many NAWQC, Tier II values, and other TRVs.

WQC1: Chronic Ambient Water-Quality Criteria

- Are chronic NAWQCs available, as set forth by EPA (2002, 82653) and 20.6.4.12 and 20.6.4.900 of the New Mexico Administrative Code (NMED 2002, 87103)?
- If **yes**, use the New Mexico criterion as the WQC1, if available, or the EPA criterion if no New Mexico value is available.
- If **no**, go to the WQC2 section.

NAWQCs are developed by EPA's OW under the Clean Water Act, Section 304 (EPA 2002, 82653). New Mexico has developed similar criteria for "high quality coldwater fisheries" as listed in "Standards for Interstate and Intrastate Streams," Sections 20.6.4.12 and 20.6.4.900 of the New Mexico Administrative Code (NMED 2002, 87103). The development of NAWQCs is outlined in EPA guidance (EPA 1995, 62787). NAWQC values are considered applicable or relevant and appropriate requirements (ARARs), and, therefore, should be considered foremost for water ESL adoption (Sample et al. 1998, 62807). NAWQCs have been developed for chronic exposure of aquatic organisms to some waterborne chemicals (EPA 1996, 62792; NMED 2002, 87103). Metals are often water hardness-dependent and should be adjusted for site-specific conditions (see EPA [1996, 62792], EPA [2002, 82653], and Sections 20.6.4.12 and 20.6.4.900 of the New Mexico Administrative Code [NMED 2002, 87103] for explanations/delineation of methods, as methods require analyte-specific information).

WQC2: Tier I Water-Quality Criteria

- Can Tier 1 WQCs be derived? (See criteria for derivation below.)
- If **yes**, use the methods described here for deriving WQC2.
- If **no**, go to the WQC3 section.

EPA recommends that Tier I WQCs be developed in the absence of NAWQCs. Tier I CVs can be determined using methods of the Great Lakes Water Quality Initiative as detailed in EPA guidance (EPA 1995, 62787). A discussion of the acceptability criteria and details of the determination of Tier I values can be found in an EPA report (EPA 1995, 62787, pp. 15395–15399). In general, the Laboratory uses the methods described in the EPA report, while using information from the AQUIRE database (AQUIRE 1997, 62898). Similar information may be available in the primary literature. The fundamental requirements and methods for deriving Tier I criteria are outlined below.

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

- the family *Salmonidae* in the class *Osteichthyes*,
- one other family in the class *Osteichthyes*,
- a third family in the phylum *Chordata* (e.g., fish, amphibian),
- a planktonic crustacean (e.g., a cladoceran, copepod),
- a benthic crustacean (e.g., ostracod, isopod, amphipod, crayfish),
- an insect (e.g., mayfly, dragonfly, damselfly, stonefly, caddisfly, mosquito, midge),

- a family in a phylum other than *Chordata* or *Arthropoda* (e.g., *Annelida*, *Molluska*, *Rotifera*), and
- 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 EPA definition of lower chronic limit (EPA 1995, 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,

- order the GMCVs from low to high;
- assign ranks, r, to the GMCVs from "1" for the lowest to "n" for the highest. If two or more GMCVs are identical, they are assigned successive ranks;
- calculate an empirical cumulative probability (P) for each GMCV as r/(n+1);
- select the four GMCVs with cumulative probabilities closest to 0.05; and
- using the four selected GMCVs and Ps, calculate a FCV (EPA 1995, 62787).

FCVs are calculated using the following mathematical relationships:

The slope (S) of the FCV calculation equation, is derived using equation 4.3-17.

$$S^2 = \frac{\left[\sum_1^4 (\ln GMCV)^2\right] - \frac{\left[\sum_1^4 (\ln GMCV)\right]^2}{4}}{\left[\sum_1^4 P\right] - \frac{\left[\sum_1^4 \sqrt{P}\right]^2}{4}}$$ **Equation 4.3-17**

The value for the intercept or location parameter of the equation, L, is calculated using equation 4.3-18.

$$L = \frac{\left[\sum_1^4 \ln GMCV\right] - \left[S \cdot \sum_1^4 \sqrt{P}\right]}{4}$$ **Equation 4.3-18**

The calculated slope and intercept are summed as shown in equation 4.3-19 to obtain A, which represents the natural log of the FCV. The FCV is therefore given by equation 4.3-20.

$$A = L + S \cdot \sqrt{0.05}$$ **Equation 4.3-19**

$$FCV = e^A$$ **Equation 4.3-20**

Tier I FCVs are also calculable based on acute values. This method calculates 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 4.3-17 through 4.3-20 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 (e.g., LC_{50}/CV) 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 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:

- at least one is a fish;
- at least one is an invertebrate; and
- 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:

- 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 rigorous [see EPA 1995, 62787, p. 15398]).
- Calculate the FACR as the geometric mean of the SMACRs.
- Calculate an FAV.
- Calculate the FCV by dividing the FAV by the FACR.

Generally, the final Tier I criterion is considered the FCV; however, if one-half the FAV is lower than the FCV, then one-half the FAV 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, 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 is 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 below.

WQC3: Tier II Water-Quality Criteria

If no NAWQC exists, and a Tier I WQC cannot be derived, the following methods are used to evaluate a Tier II WQC.

If three or more experimentally determined ACRs (see above) are available for the COPC, the FACR should be determined as described above. If fewer than three ACRs may be calculated, it is assumed that each "missing" ACR value is equal to 18, so the total number of "ACRs" equals three. A secondary acute-chronic ratio (SACR) is calculated as the geometric mean of the three ACRs. The secondary chronic value (SCV) is calculated using one of the following equations: Equation 4.3-21 using the SACR and FAV as calculated under Tier I methods; Equation 4.3-22 using methods for calculating a secondary acute value (SAV) as outlined below, and the FACR, as outlined under Tier I methods; and Equation 4.3-23 using the SAV and SACR.

$$SCV = \frac{FAV}{SACR}$$ **Equation 4.3-21**

$$SCV = \frac{SAV}{FACR}$$ **Equation 4.3-22**

$$SCV = \frac{SAV}{SACR}$$ **Equation 4.3-23**

SAVs are presented in detail in the Great Lakes document (EPA 1995, 62787, p. 15400). To calculate a SAV a minimum of one genus mean acute value (GMAV) for a daphnid (*Crustacea*: *Cladocera*) must be used, 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 4.3-9). The requirement of at least one daphnid GMAV has been criticized for restricting the number of benchmarks that can be calculated (Suter 1996, 62805). However, Suter and Tsao (1996, 59838) provide SMAVs for calculating SAVs when no daphnid GMAVs can be calculated. These values are also presented in Table 4.3-9.

Table 4.3-9
Secondary Acute Value Factors (SAVFs) for Estimation of Tier II Values

Number of 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, 62787).
[b] Factors taken from Suter and Tsao (1996, 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 NAWQCs in no more than 20% of all cases. The Tier II CV is then adopted as WQC3.

When an NAWQC, a Tier I, or a Tier II value is not available or cannot be calculated, other toxicologically based benchmarks are taken from other sources, particularly primary literature. Suter and Tsao (1996, 59838) and Suter (1996, 62805) provide information on a variety of potential benchmarks and resources. When an NAWQC, a Tier I, or a Tier II value is not available or cannot be calculated for a given COPC, then the COPC should be discussed 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 4.3-1, which is the general wildlife exposure model. To

screen the drinking water pathway, it is assumed that all oral exposure to water is derived from drinking water. Thus, exposure is calculated as follows:

$$E_{water} = C_{water} \cdot I_{water}$$ **Equation 4.3-24**

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])

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

$$ESL_{ij} = \frac{1000 \cdot NOAEL_{ij}}{I_i}$$ **Equation 4.3-25**

Where ESL_{ij} is the water ESL for wildlife species *i* and COPC *j* (µg/L)

1000 is the number of µg per mg

$NOAEL_{ij}$ 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)

The main parameters introduced in Section 4.2 are summarized in Table 4.2-3. Suter (1996, 62805) and Suter and Tsao (1996, 59838) discuss the uncertainties and limitations associated with generating water ESLs.

4.3.5 Multimedia Screening Calculations

This part of the screening evaluation tracks 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. Equation 4.3-26 should be used to calculate the HI for concurrent exposure to chemicals in more than one media:

$$HI_i = \sum_{j=1}^{n} \frac{D_{ij}}{NOAEL_{ij}}$$ **Equation 4.3-26**

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

There are alternate equations to calculate the media-specific HI for each receptor based on the ESLs and the multimedia HI. These equations are presented as Equations 4.3-27 through 4.3-34. Equations 4.3-27 through 4.3-29 present the HQ calculations for each media for a given receptor *i* for COPC *j*. Equations 4.3-30 through 4.3-32 show the calculation summing all COPCs for a given receptor *i* in a particular medium. The remaining two equations demonstrate how these HIs would be combined to produce a multimedia HI. Equation 4.3-33 shows the multimedia HI where a receptor is potentially exposed to contaminated soil and water; these receptors include robin, kestrel, deer mouse, desert

cottontail, shrew, and red fox. Equation 4.3-34 shows the multimedia HI where a receptor is potentially exposed to contaminated sediment and water; these species include the bat and swallow.

$$HQ_{soil,i,j} = \frac{C_{soil,j}}{ESL_{soil,i,j}}$$ **Equation 4.3-27**

$$HQ_{sediment,i,j} = \frac{C_{sediment,j}}{ESL_{sediment,i,j}}$$ **Equation 4.3-28**

$$HQ_{water,i,j} = \frac{C_{water,j}}{ESL_{water,i,j}}$$ **Equation 4.3-29**

$$HI_{soil,i} = \sum_{j=1}^{n} HQ_{soil,i,j}$$ **Equation 4.3-30**

$$HI_{sediment,i} = \sum_{j=1}^{n} HQ_{sediment,i,j}$$ **Equation 4.3-31**

$$HI_{water,i} = \sum_{j=1}^{n} HQ_{water,i,j}$$ **Equation 4.3-32**

$$HI_i = HI_{soil,i} + HI_{water,i}$$ **Equation 4.3-33**

$$HI_i = HI_{sediment,i} + HI_{water,i}$$ **Equation 4.3-34**

Where $C_{soil,j}$ is the concentration of the COPC *j* in soil (mg/kg)

$C_{sediment,j}$ is the concentration of the COPC *j* in sediment (mg/kg)

$C_{water,j}$ is the concentration of the COPC *j* in water (µg/L)

$ESL_{sediment,i,j}$ is the ESL for receptor *i* and the COPC *j* in sediment (mg/kg)

$ESL_{soil,i,j}$ is the ESL for receptor *i* and the COPC *j* in soil (mg/kg)

$ESL_{water,i,j}$ is the ESL for receptor *i* and the COPC *j* in water (µg/L)

$HQ_{soil,i,j}$ is the HQ for receptor *i* and the COPC *j* in soil (unitless)

$HQ_{sediment,i}$ is the HQ for receptor *i* and the COPC *j* in sediment (unitless)

$HQ_{water,i}$ is the HQ for receptor *i* and the COPC *j* in water (unitless)

$HI_{soil,i,j}$ is the HI for receptor *i* in soil (unitless)

$HI_{sediment,i}$ is the HI for receptor *i* in sediment (unitless)

$HI_{water,i}$ is the HI for receptor *i* in water (unitless)

n is the number of COPCs in each media

HI_i is the multimedia HI for receptor *i* (unitless)

Table 4.3-10 provides a summary of the variables used to develop water ESLs. The list of receptors and media to which the ESLs may be applied is given in Table 4.3-11.

Table 4.3-10
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	National ambient water quality criteria. Tier II values were developed so aquatic life criteria could be established with fewer data than 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 4.3-9) 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 NAWQC) 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 (LANL 2004, 87386; LANL 2004, 87437, or latest version). 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 are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).
I_i	Body weight normalized water intake for wildlife receptors, see values provided in Table 4.2-5. Thus, body weight is an implicit component of this variable. For this reason Table 4.2-5 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 4.3-11
List of Wildlife Receptors and Applicable Media for Multimedia Exposure to Nonradiological 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
Aquatic community	No	Yes	Yes

4.4 ESL Calculations for Radiological Constituents

The methods presented in this section were developed before DOE guidance on the ecological evaluation of radionuclides was established in "A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota" (DOE 2002, 85637) and "RESRAD-BIOTA: A Tool for Implementing a Graded Approach to Biota Dose Evaluation, User's Guide, Version 1" (DOE 2004, 85639). However, the methods are consistent with DOE guidance and with the conceptual basis presented by NMED for evaluating ecological effects of radionuclides (NMED 2000, 87104).

The graded approach developed by DOE considers the potential for adverse effects on terrestrial, aquatic, and riparian receptors based on three tiers of assessment (DOE 2002, 85637). The first tier provides only a single screening value for each medium (soil, sediment, or water) and is thus similar to the final ESLs. However, the first tier of the DOE methods does not provide any way to evaluate the set of receptors and trophic levels considered in this document. Thus, the Laboratory has retained the methods described in this section so screening assessments of radionuclides and nonradionuclides are based on the same set of receptors. Using the current Laboratory method for radionuclides, the final ESLs for soil are lower than those developed under Tier I by DOE for most radionuclides. The notable exceptions are cesium-134, cesium-137, and strontium-90; Laboratory final ESLs for these radionuclides exceed the DOE screening levels by at least an order of magnitude. These DOE screening levels and their potential impact on the results of the screening assessment should be discussed in the uncertainty analysis.

Radionuclide ESLs are calculated by the 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

$$Dose = Effective\ Energy \cdot Amount \qquad \textbf{Equation 4.4-1}$$

Much of the confusion in calculating radionuclide dose relates to the units of the terms in Equation 4.4-1. 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 indicates 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 difference between the radionuclide and nonradionuclide wildlife models is that the radionuclide models require calculating the internal concentration or body burden and the nonradionuclide models require calculating the exposure to the contaminant. Conversion factors are also required to account for the effective energy for different types of radionuclides in different media. Table 4.4-1 summarizes the receptors and diets used to model exposure to radionuclides in soil, sediment, and water. The same receptor species are used to model terrestrial exposure to radionuclides and nonradionuclides, but aquatic receptors for radionuclides consist of four specific species that serve as receptors, whereas aquatic ESLs for nonradionuclides are based on standards and benchmarks that are considered to be broadly protective of all aquatic species.

Radionuclide Dose Limits

Radionuclide dose limits are the equivalent of the NOAELs used to develop nonradionuclide ESLs. The International Atomic Energy Agency (IAEA) has concluded that doses protective of human health are protective of ecological resources, except under the following conditions, when doses protective of human health may not provide adequate protection of ecological resources (IAEA 1992, 62802):

- human access is restricted but access by biota is not restricted,
- unique exposure pathways exist,
- threatened or endangered species are present, or
- other stresses are significant.

For these four situations, IAEA recommends a dose limit of 0.1 rad/day. Because this dose limit is considered appropriately conservative and is consistent with the results of the National Council on Radiation Protection (NCRP) reviews (NCRP 1991, 62803) and Eisler (1994, 63043), the Laboratory has adopted 0.1 rad 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

$$\textit{Total Acceptable Dose} = 0.1\ (\text{rad/day}) = \textit{Internal Dose} + \textit{External Dose} \qquad \textbf{Equation 4.4-2}$$

DOE has also recommended 0.1 rad/day as the dose limit for wildlife, but DOE has specified 1 rad/day as the basis for plant and aquatic animal screening values, and DOE has not developed screening levels that are specifically protective of soil invertebrates (DOE 2002, 85637). Thus, the Laboratory has selected a more protective dose limit for plant and aquatic receptors.

4.4.1 Soil ESLs

The operational definition of soil was provided in Section 4.3.1, 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 4.4-2). ESLs for the other nine terrestrial receptors should be used as appropriate in the multimedia assessment (see Section 4.4.4, Multimedia Screening Calculations) or site-specific uncertainty analysis. ESLs are developed to account for dose from a single radionuclide. The HQ is calculated for radionuclides in the same manner as nonradionuclides to identify COPECs. Values for HQs of all radionuclides are summed to produce an HI. If the radionuclide HI exceeds 1.0, then radionuclides with HQ values greater than 0.3 are retained as COPECs.

Table 4.4-1
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
		Montane 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]
		Montane 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 used only to evaluate whether those media may have significant exposure pathways and COPCs because ESLs for one medium do not account for exposure to the same COPC in another medium. In all cases where a site has contaminated water or sediment, a multimedia assessment (see Section 4.3.5) 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 is required to address the potential cumulative effects from soil (or sediment) and water for these receptors.

Table 4.4-2
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)
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.

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 included in the calculations for radionuclides in soil are identical to those for nonradionuclides (Figure 4.3-1). 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 overestimate dose and are used for screening purposes only. The calculations for estimating internal and external doses from radionuclides in soil are derived from the calculations presented in Higley and Kuperman (1996, 62804). The basic model for calculating acceptable dose from soil for radionuclides is

$$Dose_j = 0.1 = C_{organism,j} \cdot DCF_{int,j} + C_{soil,j} \cdot DCF_{ext,j}$$ **Equation 4.4-3**

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 BW)

$C_{soil,j}$ 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 soil)

Internal dose results from exposure to radionuclides through plant uptake, incidental soil ingestion, and food web uptake (Figure 4.3-2). External dose is based on exposure to gamma emitting radionuclides from contaminated soil (Figure 4.3-2).

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

$$C_{plant,j} = C_{soil,j} \cdot TF_{plant,j}$$ **Equation 4.4-4**

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 (pCi/g fresh plant weight per pCi/g soil dry weight)

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

$$C_{invert,j} = C_{soil,j} \cdot TF_{invert,j}$$ **Equation 4.4-5**

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* (pCi/g fresh invertebrate weight per pCi/g soil dry weight)

Values and references for transfer factors are presented in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version). When values are not available in the literature for soil-to-invertebrate transfer, a default value of 1.0 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 4.4-6.

$$C_{wildlife,j} = C_{soil,j} \cdot [I_{soil} + TF_{food,j} \cdot I_{food}] \cdot TF_{blood,j} \cdot R_{t,j}$$ **Equation 4.4-6**

Where $C_{wildlife,j}$ is the body burden of radionuclide *j* in a wildlife species (pCi/g)

$C_{soil,j}$ is the concentration of radionuclide *j* in soil (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* (pCi/g fresh food weight per pCi/g soil dry weight)

$TF_{blood,j}$ is the food to blood transfer factor (pCi/g fresh blood per pCi/g fresh food)

$R_{t,j}$ is the retention time of radionuclide *j* in the organism (days)

Dietary and soil ingestion rates for each receptor are presented in Table 4.2-5. Values and supporting references for all transfer factors used are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or the latest version). 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, depending 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, 62801)

$$R_t = (1 - e^{-\lambda Tc}) / \lambda$$ **Equation 4.4-7**

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)

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 2004, 87386; LANL 2004, 87437, or latest version).

Internal Dose Conversion Factor

The radionuclides uranium, plutonium, americium, thorium, and radium have radioactive daughters. For screening purposes, the sum of average energies per disintegration for the decay chains of all radioactive daughters for any given isotope is 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 given in the units million electron volts (MeV) per disintegration. 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 4.4-8.

Equation 4.4-8

$$5.11\times10^{-5}\,\frac{disintegrations\cdot g\cdot rad}{MeV\cdot pCi\cdot day}=1.6\times10^{-6}\,\frac{ergs}{MeV}\cdot\frac{rad}{100ergs/g}\cdot\frac{disintegration}{27.03pCi\cdot s}\cdot 8.64\times10^{4}\,\frac{s}{day}$$

Where disintegrations is spontaneous disintegration of a radioactive substance along with the emission of ionizing radiation

erg is a unit of energy equal to a force of one dyne acting over one centimeter (equal to 0.642×10^{12} electron volts)

MeV is million electron volts

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

$$DCF_{int,j} = CF_i \cdot (f_a \cdot 20 + [1 - f_a])E_j$$ **Equation 4.4-9**

Where $DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (rad/day per pCi/g)

CF_i is the conversion factor between energy per disintegration and rad/day [value is 5.11×10^{-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 and 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, 62801; Eckerman and Ryman 1993, 62798). Several simplifying assumptions are made in estimating this dose. First, as indicated by the conceptual model diagram (Figure 4.3-1), 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, 62804). To emphasize the protective nature of the screening levels, "worst case" assumptions are 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° 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 are used (Eckerman and Ryman 1993, 62798) to provide a conservative estimate of external dose, as dose resulting from immersion in contaminated soil would be less than dose from water from 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 is used (Eckerman and Ryman 1993, 62798). As larger organisms receive a greater proportion of the external dose, the standard man is used as a default organism to conservatively represent exposure to all terrestrial receptors living on or above the soil surface. Thus, external DCF is modeled by the following equations:

Invertebrates and burrowing mammals:

$$DCF_{ext,j} = DC_{water,skin,j} \cdot CF_{e,w}$$ **Equation 4.4-10a**

Terrestrial receptors on or above the soil surface:

$$DCF_{ext,j} = DC_{soil,skin,j} \cdot CF_{e,s}$$ **Equation 4.4-10b**

Where $DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/day per pCi/g)

$DC_{water,skin,j}$ is the dose coefficient for skin exposed to water contaminated to an infinite depth with radionuclide *j* (from Eckerman and Ryman 1993, 62798)

$CF_{e,w}$ is the conversion factor from Sv/s per Bq/m^3 [6] to rad/day per pCi/g for an organism immersed in water [value is 3.2×10^{11}; see Equation 4.4-11]

$DC_{soil,skin,j}$ is the dose coefficient for skin exposed to soil contaminated to an infinite depth with radionuclide *j* (from Eckerman and Ryman 1993, 62798)

$CF_{e,s}$ is the conversion factor from Sv/s per Bq/m^3 to rad/day per pCi/g for an organism on the soil surface [value is 5.11×10^{11}; see Equation 4.4-12]

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

$$CF_{e,w} = 10^3 \frac{kg}{m^3} \cdot 10^3 \frac{g}{kg} \cdot 100 \frac{rad}{Sv} \cdot \frac{Bq}{27.03 pCi} \cdot 86400 \frac{s}{d}$$ **Equation 4.4-11**

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

$$CF_{e,s} = 1.6 \times 10^3 \frac{kg}{m^3} \cdot 10^3 \frac{g}{kg} \cdot 100 \frac{rad}{Sv} \cdot \frac{Bq}{27.03 pCi} \cdot 86400 \frac{s}{d}$$ **Equation 4.4-12**

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 written as

$$ESL = \frac{0.1}{TF_{plant,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 4.4-13**

[6] Sievert (Sv) is the Standards International (SI) unit for biologically effective dose corresponding to rem. Becquerel (Bq) is the SI unit for activity of source corresponding to curie.

Where 0.1 rad/day is the dose limit

$TF_{plant,j}$ is the soil to plant transfer factor for radionuclide *j* (pCi/g fresh plant weight per pCi/g dry soil weight)

$DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (rad/day per pCi/g) (from Equation 4.4-9)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/day per pCi/g) (from Equations 4.4-10a and 4.4-10b)

For terrestrial invertebrate receptors, the ESL equation is written as

$$ESL = \frac{0.1}{TF_{invert,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 4.4-14**

Where 0.1 rad/day is the dose limit

$TF_{invert,j}$ is the soil to invertebrate transfer factor for radionuclide *j* (pCi/g fresh invertebrate weight per pCi/g dry soil weight)

$DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (rad/day per pCi/g) (from Equation 4.4-9)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/day per pCi/g) (from Equations 4.4-10a and 4.4-10b)

For terrestrial herbivores, the ESL equation is written as

$$ESL = \frac{0.1}{[I_{soil,i} + TF_{plant,j} \cdot I_{plant,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 4.4-15**

Where 0.1 rad/day is the dose limit

$I_{soil,i}$ is the normalized daily soil ingestion rate for organism *i* (g of soil/g of body wt/day)

$TF_{plant,j}$ is the soil to plant transfer factor for radionuclide *j* (pCi/g fresh plant weight per pCi/g dry soil weight)

$I_{plant,i}$ is the normalized daily plant ingestion rate for organism *i* (g of dry plant/g of body wt/day)

$TF_{blood,j}$ is the food to blood transfer factor for radionuclide *j* (pCi/g fresh blood per pCi/g fresh food)

$R_{t,j}$ 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 4.4-9)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/day per pCi/g) (from Equations 4.4-10a and 4.4-10b)

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

$$ESL = \frac{0.1}{[I_{soil,i} + TF_{invert,j} \cdot I_{invert,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 4.4-16**

Where 0.1 rad/day is the dose limit

$I_{soil,i}$ is the normalized daily soil ingestion rate for organism *i* (g of soil/g of body wt/day)

$TF_{invert,j}$ is the soil to invertebrate transfer factor for radionuclide *j* (pCi/g fresh invertebrate weight per pCi/g dry soil weight)

$I_{invert,i}$ is the normalized daily invertebrate ingestion rate for organism *i* (g of dry food/g of body wt/day)

$TF_{blood,j}$ is the food to blood transfer factor for radionuclide *j* (pCi/g fresh blood per pCi/g fresh food)

$R_{t,j}$ 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 4-4-9)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/day per pCi/g) (from Equations 4.4-10a and 4.4-10b)

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

Equation 4.4-17

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

Where 0.1 rad/day is the dose limit

$I_{soil,i}$ is the normalized daily soil ingestion rate for organism *i* (g of soil/g of body wt/day)

$TF_{plant,j}$ is the soil to plant transfer factor for radionuclide *j* (pCi/g fresh plant weight per pCi/g dry soil weight)

$I_{plant,i}$ is the normalized daily plant ingestion rate for organism *i* (g of dry plant/g of body wt/day)

$TF_{invert,j}$ is the soil to invertebrate transfer factor for radionuclide *j* (pCi/g fresh invertebrate weight per pCi/g dry soil weight)

$I_{invert,i}$ is the normalized daily invertebrate ingestion rate for organism *i* (g of dry food/g of body wt/day)

$TF_{blood,j}$ is the food to blood transfer factor for radionuclide *j* (pCi/g fresh blood per pCi/g fresh food)

$R_{t,j}$ is the retention time for radionuclide *j* (days)

$DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (from Equation 4.4-9)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (from Equations 4.4-10a and 4.4-10b)

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 4.4-18

Where 0.1 rad/day is the dose limit

$I_{soil,i}$ is the normalized daily soil ingestion rate for organism *i* (g of soil/g of body wt/day)

$TF_{flesh,j}$ is the soil-to-flesh transfer factor for radionuclide *j* (pCi/g fresh flesh weight per pCi/g dry soil weight)

$I_{flesh,i}$ is the normalized daily flesh ingestion rate for organism *i* (g of dry food/g of body wt/day)

$TF_{blood,j}$ is the food to blood transfer factor for radionuclide *j* (pCi/g fresh blood per pCi/g fresh food)

$R_{t,j}$ is the retention time for radionuclide *j* (days)

$DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (from Equation 4.4-9)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (from Equations 4.4-10a and 4.4-10b)

Table 4.4-3 summarizes the variables used to calculate soil ESLs for radionuclides. Nonradionuclide and radionuclide ESL calculations share many common variables (compare Table 4.3-3 with Table 4.4-3). Thus, much of the discussion concerning uncertainty in the nonradionuclide ESLs is directly relevant to the radionuclide ESLs. Three variables, the retention time, the transfer factor from food to blood, and the dose conversion factors, are unique to radionuclides. The retention time and blood transfer factors vary between species and are based on laboratory experimental data. Thus, some uncertainty in these values exists. 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 the physical properties of each radionuclide and typically have less uncertainty, especially in the screening context where worst-case assumptions are made.

4.4.2 Sediment ESLs

Discussion on the operational definition of sediment was provided in Section 4.3.3, Sediment ESLs. Radionuclide sediment ESLs are based on exposure of contaminated sediment to aquatic receptors and to the bat and swallow through ingestion of contaminated prey. The final radionuclide sediment ESL is the lowest receptor-specific ESL among the four aquatic receptors as well as the bat and swallow as shown in Table 4.4-4. ESLs for the other terrestrial receptors should be used as appropriate in the site-specific uncertainty analysis discussion. ESLs are developed to account for dose from a single radionuclide. The HQ is calculated for radionuclides in the same manner as nonradionuclides to identify COPECs. Values for HQs of all radionuclides are summed to produce an HI. If the radionuclide HI is greater than 1.0, then radionuclides with HQ values exceeding 0.3 are retained as COPECs.

An ESL calculation for aquatic organisms exposed to sediment is based on the models presented by Baker and Soldat (1992, 62801). The radiological dose to aquatic organisms is the external dose from the radionuclide in sediment; the internal dose from sediment radionuclides is accounted for in the water ESL calculations for aquatic organisms for radionuclides (Baker and Soldat 1992, 62801; DOE 2002, 85637). Sediment-based thresholds used for screening values do not exist for radionuclides, so algae, daphnids, and snails and fish have been chosen as surrogates for organisms living in aquatic environments at the Laboratory. Transport pathways from sediment to aquatic organisms are presented in Figure 4.4-1. In addition, to address bioaccumulation and some biomagnification, bats and swallows have been chosen as higher trophic level terrestrial receptors that feed primarily upon insects emerging from sediment in aquatic environments. ESLs calculated for these receptors assume they are feeding 100% upon aquatic invertebrates. The pathways for bat and swallow exposure to sediment are the same as presented in Figure 4.3-3. The basic model for calculating acceptable dose from sediment for radionuclides is

$$Dose_j = 0.1 = C_{sediment,j} \cdot DCF_{ext,j}$$ **Equation 4.4-19**

Where $Dose_j$ is the total acceptable dose from radionuclide *j* (rad/day)

$C_{sediment,j}$ 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)

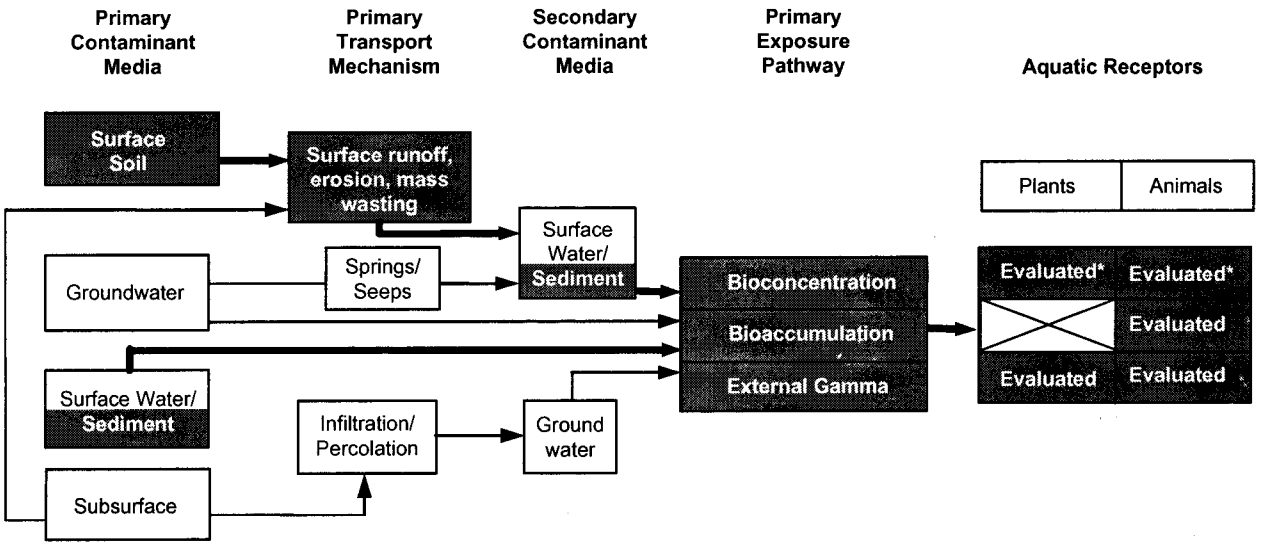
Table 4.4-3
Summary of Variables Used in Soil ESL Calculations for Radionuclides

Variable	Source
I_{soil}	Body weight normalized soil ingestion rate for wildlife receptors (food intake × fraction of soil in diet from Table 4.2-5).
I_{plant}	Body weight normalized plant ingestion rate for wildlife receptors (food intake × fraction of plants in diet from Table 4.2-5).
I_{invert}	Body weight normalized invertebrate ingestion rate for wildlife receptors (food intake × fraction of invertebrates in diet from Table 4.2-5).
I_{flesh}	Body weight normalized flesh ingestion rate for wildlife receptors (food intake × fraction of flesh in diet from Table 4.2-5).
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 4.4-7 for calculation of this variable.
TF_{blood}	The transfer factor from food to blood is a COPC-specific value derived from site-specific studies, other empirical literature studies, and/or models. The transfer factor is based on the beef transfer factor (TF_{beef}) in pCi/g fresh beef per pCi COPC/day and the food ingestion rate. Values for specific COPCs are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).
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/or models. Values for specific COPCs are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).
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/or models. Values for specific COPCs are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).
TF_{flesh}	The transfer from soil to flesh is a COPC-specific value that is derived from three other factors. The first factor is a fresh weight feed to muscle transfer factor (TF_{beef}) derived from studies of beef cattle. The second factor is the maximum of either the moisture content adjusted dry weight TF_{plant} or the MC adjusted dry weight TF_{invert}. This transfer factor term represents the prey with the most contaminated diet. The two transfer factors are multiplied by a food ingestion rate. This rate is based on a composite prey species value developed from the four potential mammalian prey species (robin, deer mouse, cottontail, and shrew). The highest food and soil intake rates among these four potential prey species was used to represent the composite prey species in the equation. Thus, $TF_{flesh} = TF_{beef} \cdot (I_{food} \cdot \text{maximum of } (TF_{plant} \cdot (1\text{-}MC_{plant}), TF_{invert} \cdot (1\text{-}MC_{invert})) + I_{soil})/(1\text{-}MC_{flesh})$ Values for specific COPCs are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).
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 BW and accounts for the increased biological effectiveness of alpha particle deposition over beta or gamma deposition (see Equation 4.4-9).
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 contains the unit conversion factor rad/d per pCi/g soil.

Table 4.4-4
Method for Determining Final Sediment ESL for Radionuclides

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

[a] Lower of snail and daphnid ESLs.
[b] Lower of bat and swallow ESLs.
[c] Value = Value available for that contaminant.
[d] No value = No value available for that contaminant.



Note: Boxes with Xs indicate incomplete pathways. Open boxes indicate complete pathways. Complete pathways for sediment exposure to aquatic receptors are gray; evaluated pathways are included in the sediment ESL calculations for aquatic receptors.

* Bioconcentration is evaluated for sediment for plants and animals using water ESLs.

Figure 4.4-1. 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 as part of the water ESL development described in section 4.4.3 (Baker and Soldat 1992, 62801). Thus, paired data for water and sediment are needed to assess the radionuclide dose.

For terrestrial receptors ingesting sediment invertebrates, however, the internal dose from invertebrate prey is explicitly considered in the sediment calculation, which is consistent with DOE standard DOE-STD-1153-2002 (DOE 2002, 85637). 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 receptors is calculated:

$$C_{organism,j} = C_{sediment,j} \cdot BCF_{invert,j} \cdot I_{food,i} \cdot TF_{blood,j} \cdot R_{t,j}$$ **Equation 4.4-20**

Where $C_{organism,j}$ is the internal concentration of radionuclide *j* (pCi/g of organism

$C_{sediment,j}$ 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* (pCi/g fresh blood per pCi/g fresh food)

$R_{t,j}$ is the retention time of radionuclide *j* in the organism (days) (see Equation 4.4-7)

Values and references for the transfer factors and bioconcentration factors are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).

Dose Conversion Factors

For aquatic receptors, internal dose conversion factors are identical to those used for terrestrial receptors (see Equation 4.4-9). For organisms that reside in, on, or near the sediment (algae, snail, fish), external dose is estimated as for terrestrial receptors living in or on soil (see Equations 4.4-10a and 4.4-10b). 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 have limited contact with sediments, it is assumed that external dose to terrestrial receptors 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 receptor. For receptors that spend at least part of their lives in close association with sediment, the sediment ESL calculation is

$$ESL = \frac{0.1}{DCF_{ext,j}}$$ **Equation 4.4-21**

Where 0.1 rad/day is the dose limit

$DCF_{ext,j}$ 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 4.4-22**

Where 0.1 rad/day is the dose limit

$I_{food,i}$ 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 (pCi/g fresh blood per pCi/g fresh food)

$R_{t,j}$ is the retention time for radionuclide j (days)

$DCF_{int,j}$ is the internal DCF for radionuclide j (rad/day per pCi/g)

Table 4.4-5 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 DCFs. The second group of uncertainties relates to receptors, including feeding rates and diets. The uncertainties associated with the variables in Table 4.4-5 are similar to those identified for radionuclides in soil.

Table 4.4-5
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 4.2-5).
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 4.4-7 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. The transfer factor is based on the beef transfer factor (TF_{beef}) in pCi/g fresh beef per pCi COPC/day and the food ingestion rate. Values for specific COPCs are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).
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 2004, 87386; LANL 2004, 87437, or latest version).
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 4.4-9).
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}.

4.4.3 Water ESLs

Discussion on the operational definition of water was provided in Section 4.3.4, 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 eight wildlife receptors (Table 4.4-6). ESLs for the other receptors should be used as appropriate in the site-specific 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. The HQ is calculated for radionuclides in the same manner as nonradionuclides to identify COPECs. Values for HQs of all radionuclides are summed to produce an HI. If the radionuclide HI is greater than 1.0 then radionuclide with HQ values that exceed 0.3 are retained as COPECs.

Table 4.4-6
Method for Determining Final Water ESL for Radionuclides

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

[a] Lower of snail and daphnid ESLs.
[b] Lower of bat and swallow ESLs.
[d] Value = Value available for that contaminant.
[e] No value = No value available for that contaminant.

Calculation of ESLs for aquatic organisms is based on the models presented by Baker and Soldat (1992, 62801). The radiological dose to aquatic receptors is the sum of the dose from internally deposited radionuclides and the external dose from the same radionuclides in water. In this model, the internal dose calculated for water ESLs for aquatic receptors includes the internal component associated with sediment as well because the bioaccumulation factor considers the partitioning of the radionuclide between sediment and water (Baker and Soldat 1992, 6280; DOE 2002, 85637). Thus, paired data for water and sediment are needed to assess the radionuclide dose. Media-based screening values for radionuclides do not exist, so algae, daphnids, and snails and fish have been chosen as assessment endpoint surrogates for receptors living in aquatic environments at the Laboratory. Transport pathways to aquatic organisms are presented in Figure 4.4-2. The only water exposure pathway considered for terrestrial receptors is ingestion of drinking water (Figure 4.3-5). The basic model for calculating acceptable dose from water for radionuclides is

$$Dose_j = 0.1 = C_{organism,j} \cdot DCF_{int,j} + C_{water,j} \cdot DCF_{ext,j}$$ **Equation 4.4-23**

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

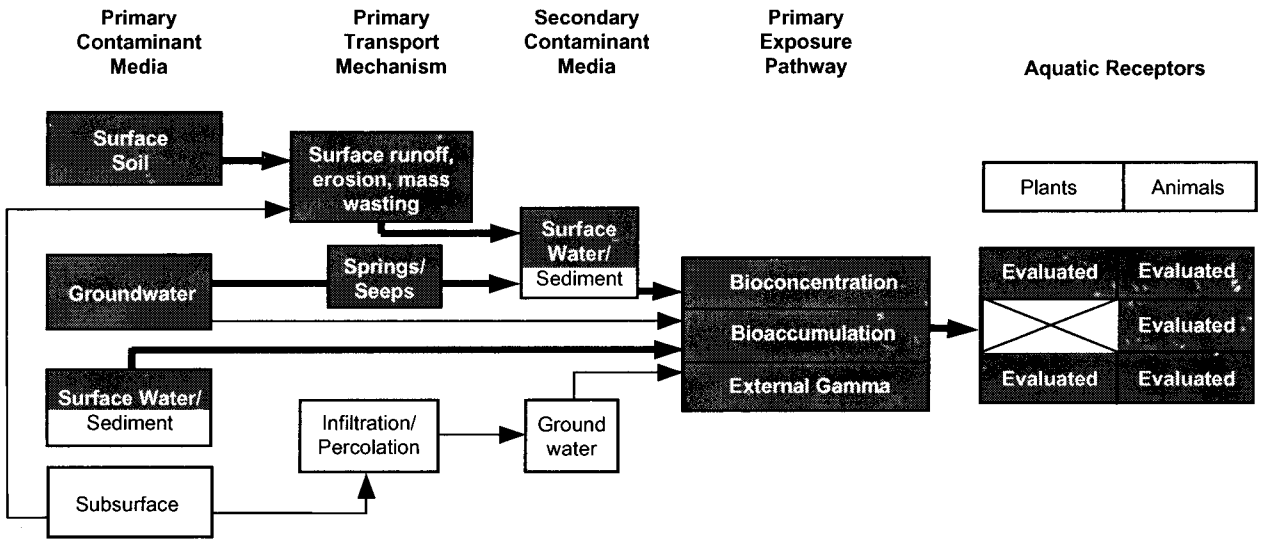
For organisms immersed in water (algae, daphnid, snail, and fish), internal concentration of any radionuclide is modeled by applying a simple bioconcentration factor (Baker and Soldat 1992, 62801):

$$C_{organism,j} = C_{water,j} \cdot BCF_{organism,j}$$ **Equation 4.4-24**

Where $C_{organism,j}$ is the internal concentration of radionuclide j (pCi/g of organism)

$C_{water,j}$ 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)



Note: Boxes with Xs indicate incomplete pathways. Open boxes indicate complete pathways. Complete pathways for water exposure to aquatic receptors are gray; evaluated pathways are included in the water ESL calculations.

Figure 4.4-2. 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 as

$$C_{organism,j} = C_{water,j} \cdot I_{water} \cdot TF_{blood,j} \cdot R_{t,j} \qquad \textbf{Equation 4.4-25}$$

Where $C_{organism,j}$ is the internal concentration of radionuclide j (pCi/g of organism)

$C_{water,j}$ 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 (pCi/g fresh blood per pCi/g fresh water) (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 4.4-7)

Values and references for the transfer factors and bioconcentration factors are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).

Dose Conversion Factors

For aquatic receptors, the internal DCFs are identical to those used for terrestrial receptors (see Equation 4.4-9). For organisms immersed in water (algae, daphnid, snail, fish), the external dose coefficients of Eckerman and Ryman (1993, 62798) are used to estimate external dose. Coefficients used are for skin immersed in water contaminated to an infinite depth. A conversion factor of $3.2 \cdot 10^{11}$ is used to convert the dose coefficients from Sv/s per Bq/m^3 to rad/day per pCi/g (see Equation 4.4-11).

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 receptor. For aquatic receptors 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 4.4-26**

Where 0.1 rad/day is the acceptable dose limit

$BCF_{i,j}$ 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 4.4-27**

Where 0.1 rad/day is the dose limit

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* (pCi/g fresh blood per pCi/g fresh water) (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 4.4-7 summarizes the input variables for the radionuclide water ESL. 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 water) and DCFs. The second group of uncertainties relates to receptors, which includes the water ingestion rate. The uncertainties associated with the variables in Table 4.4-7 are similar to those identified for radionuclides in soil.

4.4.4 Multimedia Screening Calculations

Multimedia calculations for radionuclides use the same principles and equations as for nonradionuclides. Because all radionuclides are evaluated in terms of radiological dose, it is appropriate to combine exposure across all radionuclides and contaminated media. It is particularly important to consider both the water and sediment ESL for a radionuclide because it is assumed that internal dose to aquatic organisms is modeled from exposure to water. In cases where water data are not available for a radionuclide, the internal dose from water can be estimated from the sediment concentration and the soil-water partitioning coefficient; this approach is consistent with the one taken by DOE for RESRAD-BIOTA calculations (DOE 2004, 85639). Table 4.4-8 provides the list of ecological receptors considered in the multiple radionuclide and multimedia assessment.

Table 4.4-7
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 4.2-5).
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 4.4-7 for calculation of this variable.
TF_{blood}	The transfer factor from food to blood is a COPC-specific value derived from site-specific studies, other empirical literature studies, and models. The transfer factor is based on the beef transfer factor (TF_{beef}) in pCi/g fresh beef per pCi COPC/day and the food ingestion rate. Values for specific COPCs are provided in the ECORISK Database (LANL 2004, 87386; LANL 2004, 87437, or latest version).
$BCF_{organism}$	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 2004, 87386; LANL 2004, 87437, or latest version).
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 4.4-9).
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 4.4-8
List of Wildlife Receptors and Applicable Media for Multiple Contaminants and Multimedia Exposure to Radionuclides

	Contaminated Media Evaluated for Receptor		
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

4.5 Screening Evaluation/Uncertainty Analysis

Much of the uncertainty in the screening assessment is addressed by applying exposure and toxicity values designed to be protective of all the receptors. However, the net result is likely to overestimate exposure to ecological receptors from contaminated media. Thus, more accurate estimates of exposure can be evaluated by considering factors such as area use and bioavailability of COPECs (Pastorok et al. 1996, 62784).

Bioavailability is often a key parameter in the evaluation of exposure to wildlife, and mechanistic bioconcentration or bioaccumulation models can be evaluated for their applicability (Jager 1998, 62786). AUFs and population area use factors (PAUFs) may be appropriate to modify the estimate of risk to some receptors at some sites depending on the size of the site. The introduction of area use reduces potential over-estimation of risks to receptors whose home ranges are larger than the area of contamination being evaluated. These AUFs/PAUFs may be applied to either individual organisms or populations. Area use may be particularly important for species that represent both a feeding guild and serve as a surrogate for a T&E species with a different home range than the surrogate. The flesh-eating kestrel, for example, represents both the feeding guild of carnivorous birds (using its normal home range) and serves as a surrogate for the Mexican spotted owl (which has a much larger home range). If Mexican spotted owls are potentially exposed receptors for a site, then the uncertainty analysis should include a discussion of the impact on HQs and HIs of the surrogate species when the home range of the Mexican spotted owl is used instead of the home range of the surrogate. The values in Table 4.5-1 for body weight, food ingestion rate, and home range for the Mexican spotted owl are from Gonzales et al. (2004, 85207). The value for water ingestion rate is developed from the allometric equation for drinking water consumption in birds based on body weight (EPA 1993, 59384, p. 3-8).

Table 4.5-1
Factors for the Mexican Spotted Owl

Receptor Species	Body Weight (kg)	Food Ingestion Rate (kg dry wt/day)	Water Ingestion Rate (L/kg/day)	Home Range (ha)
Mexican spotted owl	0.6	0.019	0.070	410

4.5.1 Development of Population Area Use Factors

Except when evaluating T&E species, EPA guidance recommends evaluating ecological effects at the population rather than at the individual level (EPA 1999, 70086). The initial screening using ESLs generates HQs and HIs designed to estimate the potential for risk to individual ecological receptors, assuming continuous exposure to the representative concentration of the COPC in question. The AUF is calculated based on the ratio of the site area to the home range of an individual receptor to reflect the fact that a receptor actually moves around its home range and does not remain stationary in the contaminated site. Therefore, the individual AUF assesses the level of individual exposure based on the area of the home range. The modification of a HQ or HI with a PAUF also uses the estimated area occupied by the population of a receptor species to assess the likelihood of any individual within the assessment population encountering the contaminated area, while using the same ESL based on effects to individuals to determine the impact of this contact within the contaminated area. The PAUF assumes impacts to some individuals and estimates the average effect on the assessment population of that impact. The AUFs and/or PAUFs may be used to modify an HQ or HI developed from ESLs used in the initial screen; those ESLs are based on adverse effects to an individual. Application of AUF and/or PAUF to the results of the ecological screening is generally beyond the screening level and begins to examine the uncertainty

associated with the estimates of potential risk generated by the screening analysis. PAUF puts exposure from a contaminated site in perspective of possible population impacts and provides a reasonable basis for characterizing ecological risks to wildlife.

PAUFs are developed based on investigations correlating the home range of a receptor with its dispersal distance (the distance an animal moves from its natal home range). The dispersal distance has been shown to affect population structure, demographics, and spacing patterns and can be used to determine the assessment population boundaries (Bowman et al. 2002, 73475). When home range is expressed as its linear dimension (the square root of home range), it has a good linear correlation with dispersal distance for the same species (Bowman et al. 2002, 73745). For mammals with similar home range sizes to the species used as screening receptors at the Laboratory, dispersal distance is equal to 3.5 times the square root of the home range. The relationship holds well for small mammals such as mice and rabbits but may overpredict dispersal distance for fossorial species and slightly underpredict dispersal distance for some large herbivores such as the white-tailed deer (Ryti et al. 2004, 76074). The mathematical relationship between home range and dispersal distance has been estimated only for mammals, but for the calculations at these sites the same methodology was applied to avian receptors. Bird species have higher median and maximum dispersal distances than similar-sized mammals (Sutherland et al. 2000, 73460), so application of the mammalian relationship is protective of bird species because this relationship underestimates the dispersal distance and, therefore, the assessment population area of bird species.

The dispersal distance from the center of the home range can be considered the radius of the animal's population area, with the area likely to be occupied by members of that population (the assessment population area) consisting of the circle described by the area covered by the dispersal distance. The assessment population area would therefore be equal to πr^2, which would be equal to π times (3.5 times the home range)^2. This mathematical relationship can be simplified to 40 times the home range as a representation of the assessment population area in hectares (Ryti et al. 2004, 76074). Once the population area is calculated for each receptor species of interest, the area of the site can be divided by the population area to develop a site-specific PAUF for that population. Home ranges for the screening receptors were given Table 4.2-5.

PAUFs cannot be calculated for the plant and invertebrate screening receptors, because these species do not have a home range that can be related to a population assessment area. Plant and invertebrate receptors are evaluated directly against their ESL media concentrations. Assessment populations of plants and invertebrates are evaluated in a more qualitative manner.

4.5.2 Analysis of Uncertainties in Screening Evaluation

Many factors are incorporated in the development of the ESLs, and uncertainty is associated with values for the factors and the model itself. At a minimum, the uncertainty analysis should focus on the key sources of uncertainty discussed in the two following sections. Examination of the uncertainty can result in adding or deleting COPECs. The uncertainty analysis may include a table that qualitatively discusses which factors may overestimate the potential risk for the site and which factors at the site may underestimate the potential risk to ecological receptors at the site.

4.5.2.1 Exposure-Related Parameters

The conceptual models for terrestrial and aquatic ecosystems presented in Section 4.3 describe the potential pathways that may apply to soil, sediment, or water at sites being evaluated. These models should be reviewed as part of the uncertainty analysis to determine if significant complete pathways exist

at the site under consideration that were not included in the development of the ESLs. The exposure pathways addressed by the ESL and HQ/HI analysis include all complete exposure pathways, with the exception of foliar uptake by plants, inhalation, and dermal exposure. Although the last two pathways contribute to the dose received by animals, the contribution is relatively small and does not interfere with COPEC determination, as described in Section 4.2. Soil ingestion rates, however, can represent one of the more significant source of environmental exposure, up to 18% for grazing species in areas of sparse vegetation, and over 10% for some birds and aquatic insects (Beyer et al. 1994, 62785). Therefore, the exposure pathways considered in developing the ESLs used in the screening assessment for a specific site capture the primary exposures for wildlife receptors at this site. ESLs incorporate all the exposure pathways described above; the ESLs overestimate the dose ingested if some of the pathways are not complete at the site, for example, if the contaminated media was buried at a depth inaccessible to wildlife receptors.

For pathways used in ESL development that are complete at the site, the equations used to calculate ESLs from the TRVs include terms for body weight, water intake, food intake, and inhalation rate. To provide a conservative estimate of the ESL, maximum estimates of intake factors (food, water, air) were combined with lower estimates of body weight. This approach maximizes the weight-specific dose to the receptor and is protective of all species within a feeding guild represented by a screening receptor. It may overestimate potential risk to larger-size species or to small-size species with lower intake rates than those used in the model.

Risk to farther-ranging species may also be overestimated because the area use for development of ESLs is 100%. Depending on the size of the site, this value may be appropriate for small-size species but is likely to overestimate risk for larger-size species with a home range greater than the size of the site. Section 4.5.1 discussed how to apply individual AUFs and PAUFs.

Uncertainty is associated with the values used for the representative exposure concentration. The uncertainty analysis should include some consideration of whether use of the maximum concentration of a COPC as the exposure concentration is likely to overestimate the potential ecological risk to receptors, or whether the value may underestimate the true maximum value of COPCs at a site. Use of the 95% UCL as the exposure concentration is likely to overestimate risk if the receptor has a home range greater than the area over which the 95% UCL was determined. The analysis of uncertainty associated with the representative exposure concentration should also include consideration of the findings of data review (e.g. precision and bias of sample results for environmental media samples) and the impact of the review on the confidence in representative concentration estimate.

4.5.2.2 Toxicity-Related Parameters

Another key uncertainty is the availability of toxicity information for receptor groups (e.g., birds, mammals, plants, and invertebrates). The toxicity data and uncertainty factors used to develop the ESLs may potentially overestimate the actual toxicity of a chemical to a receptor, particularly when those data are extrapolated from one species to another. In addition, the comparison of site concentrations to ESLs assumes that the chemical species or form occurring at the site is identical to the chemical species used in the toxicity analysis. The 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.

The TFs are used to estimate the potential for accumulation of contaminants through the levels of the food chain. TFs based on linear equations are used to generate ESLs. They are not well documented, and many are based on the physical properties of a chemical instead of empirically measured values.

Although the linear TFs are considered conservative, other models available can predict higher levels of accumulation. Equations based on TFs also do not account for any depuration from the organism, which tends to overestimate the concentrations at higher trophic levels. Therefore, the models and TFs used to generate the ESLs may over- or underestimate the actual concentrations within an organism, particularly at higher trophic levels.

Many sites have multiple COPCs; cumulative effects and contaminant interactions may alter the safe threshold for exposure to any or all of these COPCs. However, the ESL calculation is modeled on the assumption of the additive effects of chemicals. This assumption could overestimate or underestimate the actual impact of exposure to multiple contaminants from synergistic or antagonistic effects. No information is available for most chemicals on synergistic or antagonistic effects; therefore, almost all risk assessments assume the effects are additive when multiple chemical contaminants are present.

The ESLs also include the implicit assumption that the chemical form of the COPC is likely to be present in the environment in the same form and with the same bioavailability as the chemical form used in toxicity studies. In general, toxicity studies use readily bioavailable forms of chemicals; the TRVs from these studies may overestimate the toxicity of the chemical form of a COPEC at a site. Because TRVs are derived from toxicity studies with whole animals, the TRVs are based on the potential effects of both the administered chemical and the metabolic products of that chemical. The form of the chemical in the toxicity study may differ from that found in environmental media at the site, however, which means the chemical form at the site could potentially have different metabolic products.

Because of these uncertainties, ESLs for some inorganic chemicals may be below background concentrations of those chemicals. In cases where the background concentration is below the ESL, this issue should be addressed in the uncertainty section. An HQ for the background concentration may be presented to show the contribution of background to the overall estimate of potential risk at the site. If the representative concentration for the site is within the range of background concentrations, the uncertainty analysis should also discuss whether the representative concentration represents an elevated risk or represents an exposure similar to background across the site.

4.6 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 potential 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 result from the screening assessment include the following:

1. There is adequate information to conclude that the ecological risks are negligible and no additional investigation of ecological risk is recommended. For example, no unacceptable risks are inferred if the screening evaluation identifies no COPECs.
2. There are sufficient lines of evidence to document potential or actual adverse ecological effects.
3. Ecological risks are not negligible, but the information is insufficient to indicate that adverse ecological effects are occurring. The recommended risk management strategy is to move to the next level of ecological risk assessment to evaluate the potential for adverse ecological impacts.

This next level of investigation uses the results of the screening assessment to focus the investigation. The approach used in the next level of investigation can vary from collecting more and/or better site-specific information to reduce uncertainties in the screening assessment to conducting a baseline risk assessment.

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

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Whitaker, J. O., Jr., 1980. *The Audubon Society Field Guide to North American Mammals*, Alfred A. Knopf, New York, New York. (Whitaker 1980, 62889)

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 –**
Vegetation class based on GIS vegetation coverage **(Indicate all that apply.)**	**Water –** **Bare Ground/Unvegetated –** **Spruce/fir/aspen/mixed conifer –** **Ponderosa pine –** **Piñon juniper/juniper savannah –** **Grassland/shrubland –** **Developed –** **Burned –**
Is T&E Habitat Present? **If applicable, list species known or suspected of using 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 total score and the run-off subscore (maximum of 46); terminal point of surface water transport; slope; and surface water run-on sources.**	

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 GIS vegetation class to assist in verifying the Arcview information	
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**	

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).**	

No Exposure/Transport Pathways:

If there are no complete exposure pathways to ecological receptors onsite and no transport pathways to off-site receptors, the remainder of the checklist should not be completed. Stop here and provide additional explanation/justification for proposing an ecological No Further Action recommendation (if needed). At a minimum, the potential for future transport should include the 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 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 be 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 through vapors?

- **Volatility of the hazardous substance (volatile chemicals generally have Henry's Law constant >10^{-5} atm-m^3/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 SWMU and/or AOC 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 through 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.**
- **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?

- **The potential for contaminants to migrate to groundwater.**
- **The potential for contaminants to migrate through 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.**
- **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 the respiration of vapors?

- **Contaminants must be present as volatiles in the air.**
- **Consider the importance of the inhalation of vapors for burrowing animals.**
- **Foliar uptake of 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 the deposition of particulates or with animals through the inhalation of fugitive dust?

- **Contaminants must be present as particulates in the air or as dust for this exposure pathway to be complete.**
- **Exposure through the 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 through the 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 through 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 through the 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 suspended or sediment-based 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?

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

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

Aquatic Animals:

Provide explanation:

Question V:

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

- **External irradiation effects are most relevant for gamma-emitting radionuclides.**
- **The water column acts to absorb radiation; therefore, 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):

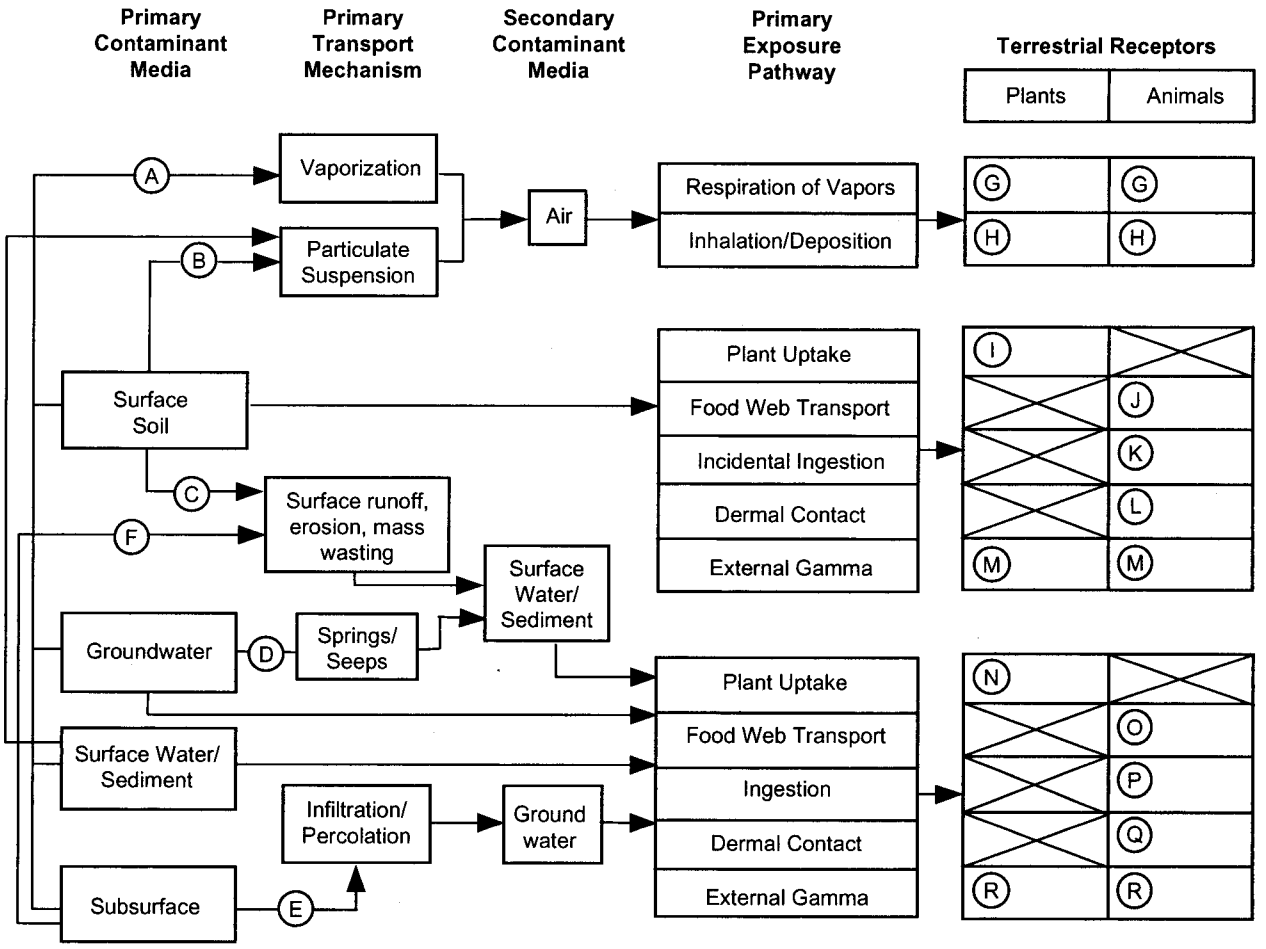
Aquatic Plants:

Aquatic Animals:

Provide explanation:

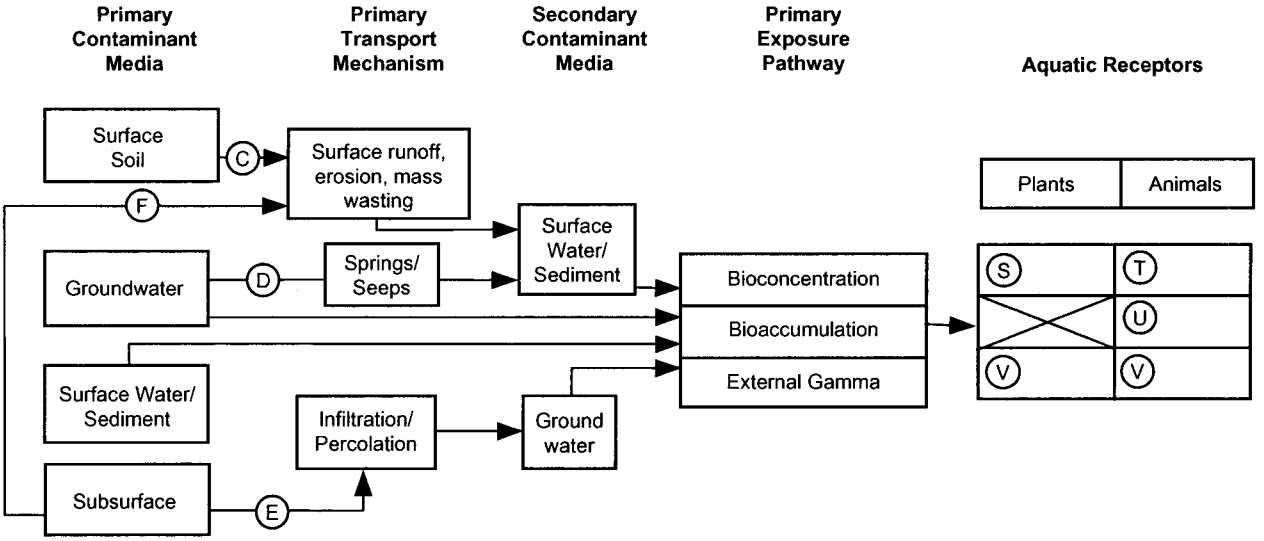
Ecological Scoping Checklist
Terrestrial Receptors
Ecological Pathways Conceptual Exposure Model

NOTE:
Letters in circles refer to questions on the Scoping Checklist



Ecological Scoping Checklist
Aquatic Receptors
Ecological Pathways Conceptual Exposure Model

NOTE:
Letters in circles refer to questions on the Scoping Checklist



Signatures and certifications:

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

Name (printed): _____

Name (signature): _____

Organization: _____

Phone number: _____

Date completed: _____

Verification by another party (provide name, organization and phone number):

Name (printed): _____

Name (signature): _____

Organization: _____

Phone number: _____

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PLATE 7

Vegetation Land Cover Map

Date 12/21/2004

#13894

General Box 18

"No map - can not be found"

13894