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Screening-Level Ecological Risk Assessment Methods, Revision 3

Prepared by the Environmental Programs Directorate

EXECUTIVE SUMMARY

This document provides guidance for screening-level assessments of potential adverse impacts to ecological resources from release of environmental contaminants at the Los Alamos National Laboratory (LANL or the Laboratory). The methods presented are based on two objectives, namely: to provide a basis for reaching consensus with regulators, managers, and other interested parties on how to conduct screening-level ecological risk investigations at the Laboratory; and to provide guidance for ecological risk assessors under the Environmental Programs (EP) Directorate. This guidance promotes consistency, rigor, and defensibility in ecological screening investigations and in reporting those investigation results. The purpose of the screening assessment is to provide information to the risk managers so informed risk-management decisions can be made. This document provides examples of recommendations and possible risk-management strategies.

This document 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. Screening assessments are performed on solid waste management units (SWMUs)[1] or areas of concern (AOCs); 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 purpose of the screening evaluation is to identify chemicals of potential concern (COPCs) that should be retained as chemicals of potential ecological concern (COPECs) by exposure media. The screening evaluation focuses future investigations on important ecological concerns of potentially contaminated sites and identifies those sites that do not have COPECs. Sites with no COPECs do not need further ecological evaluation. The outcome of the screening is expected to be protective of potential adverse ecological effects but is not intended to be predictive of ecological risk. Thus, protective assumptions are made throughout the screening evaluation to ensure that contaminants, exposure pathways, and sensitive species are not missed.

The key components of the screening evaluation are the ecological screening levels (ESLs) that are developed for each chemical and receptor and are media-specific. The ESLs are determined so if a site has levels of a chemical above the ESL in any medium, then this site warrants further consideration because it may pose a potential risk to ecological receptors. To evaluate the potential risk for each COPC, the ESL and the site exposure point 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 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 evaluated (soil, sediment, or water), they do not account for exposure to multiple media. The hazard posed by multiple chemicals is summed as a hazard index (HI) for each wildlife receptor. 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, then the site may pose an ecological risk.

This document describes the HQ and HI calculations and presents general considerations for the basis of ESL calculations. 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. The Laboratory's ECORISK Database provides the necessary information

[1] A SWMU is defined as any discernible 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.

and supporting detailed documentation for TRVs and ESLs and related information. 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. The ECORISK Database also provides a detailed basis for calculating the ESLs.

This document also describes the uncertainty analysis that follows the COPEC identification and the key sources of uncertainty in the screening assessment. This analysis includes a more refined screening assessment using ESLs based on the low observed adverse effect level (LOAEL) rather than on the no observed adverse effect level (NOAEL). The LOAEL analysis is less conservative and is designed to provide a more realistic, but still protective, estimate of potential risk.

This document also includes the ecological scoping checklist, which is a 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. Lastly, this document describes the mathematical basis for deriving ecological screening levels.

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Acronyms

AOC	area of concern
ARCS	Assessment and Remediation of Contaminated Sediments
asl	above sea level
AUF	area use factor
BCF	bioconcentration factor
BV	background value
BW	body weight
CCME	Canadian Council of Ministers of the Environment
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
COPC	chemical of potential concern
COPEC	chemical of potential ecological concern
CSL	cleanup screening level
CSM	conceptual site model
DCF	dose conversion factor
DL	detection limit
DOE	Department of Energy (U.S.)
EP	Environmental Programs (Directorate)
EPA	Environmental Protection Agency (U.S.)
EPC	exposure point concentration
EqP	equilibrium partitioning
ERM	effect range median
FR	Federal Register
GIS	geographic information system
GPS	global positioning system
HI	hazard index
HQ	hazard quotient
HR	home range
IR	inhalation rate
IAEA	International Atomic Energy Agency
K_{oc}	organic carbon partitioning coefficient
K_{ow}	octanol-water partitioning coefficient
LANL	Los Alamos National Laboratory
L-ESL	lowest effect ESL

LOAEL	lowest observed adverse effect level
LOEC	lowest observed effect concentration
MC	moisture content
MPC	maximum permissible concentration
NC	negligible concentration
NCRP	National Council on Radiation Protection
NMAC	New Mexico Administrative Code
NMED	New Mexico Environment Department
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no observed adverse effect level
NOEC	no observed effect concentration
PAUF	population area use factor
PEC	probable effect concentration
PTSE	
RPF	Records Processing Facility
SEL	severe effect level
SLERA	screening-level ecological risk assessment
SMDP	scientific management decision point
SQS	sediment quality standard
SWMU	solid waste management unit
T&E	threatened and endangered
TA	technical area
TEC	threshold effect concentration
TEL	threshold effect level
TF	transfer factor
TL	threshold level
TRV	toxicity reference value
UCL	upper confidence limit
UET	upper effects threshold
VOC	volatile organic compound
WQC	water-quality criteria

1.0 INTRODUCTION

This methodology document describes the approach used by the Los Alamos National Laboratory (LANL or the Laboratory) Environmental Programs (EP) Directorate for screening-level assessments of potential impacts to ecological resources resulting from exposure to contaminants. This approach is consistent with the U.S. Environmental Protection Agency's (EPA's) "Ecological Risk Assessment Guidance for Superfund" (EPA 1997, 059370), the "Guidelines for Ecological Risk Assessment" (EPA 1998, 062809), "Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites" (EPA 1999, 070086), and the "Guidance for Developing Ecological Soil Screening Levels" (EPA 2003, 085643). This guidance incorporates the assessment endpoints developed in "Generic Assessment Endpoints for Ecological Risk Assessment at the Los Alamos National Laboratory" (LANL 1999, 064137). 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, Revision 2" (http://www.nmenv.state.nm.us/HWB/documents/NMED_chemical_ecorisk_guidance_v2_July_2008.pdf). The approach to ecological risk screening for radionuclides provided in this document is also consistent with the U.S. Department of Energy's (DOE's) "Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota" (DOE 2002, 085637) and DOE's "RESRAD-BIOTA: A Tool for Implementing a Graded Approach to Biota Dose Evaluation, User's Guide, Version 1" (DOE 2004, 085639). This version of the document incorporates additional guidance and direction on conducting ecological risk-screening assessments and is consistent with risk assessment procedures developed by the Laboratory (Environmental Programs Directorate Standard Operating Procedure, EP-DIV-SOP-10006, Performing Human and Ecological Risk Screening Assessments).

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 high confidence that all potential adverse impacts to ecological receptors resulting from exposure to contaminants are identified in the initial investigation. 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 document presents the ecological screening process for 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. The approach assesses 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 GENERIC PROBLEM FORMULATION FOR ECOLOGICAL RISK SCREENING ASSESSMENTS

As noted in the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA or Superfund) specific ecological risk guidance (EPA 1997, 059370), problem formulation is the most critical step of an ecological risk assessment. The EPA 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 (section 2.4), which structures information on the feeding relationships among animals and plants 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.

2.1 Environmental Setting

The Laboratory is situated on the Pajarito Plateau, which consists of a series of fingerlike 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. The climate, geographic setting, geology, hydrology, and biology of the Laboratory are described briefly below.

2.1.1 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 northeast of Albuquerque and 20 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 2.1-1). 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.

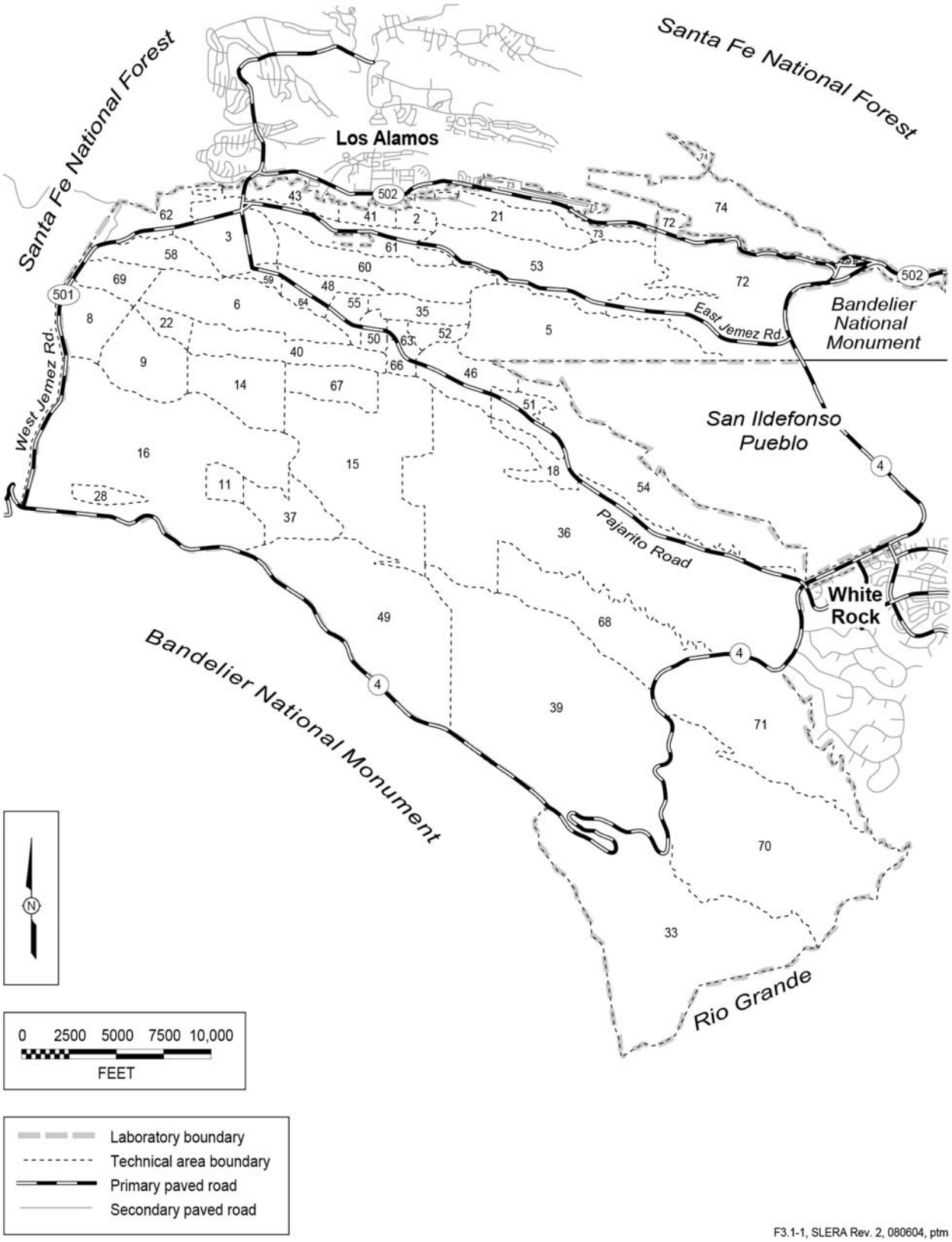


Figure 2.1-1 Laboratory TAs in relation to surrounding landholdings

2.1.2 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 day, to −9°C to −4°C (15°F to 25°F) during the night. Summer temperatures range from 21°C to 31°C (70°F to 88°F) during the day to 10°C to 15°C (50°F to 59°F) during the night. The average annual precipitation (including both rain and water equivalent of frozen precipitation) is 48 cm (19 in.). Details are available at http://weather.lanl.gov/ and are discussed in the "Installation Work Plan for Environmental Restoration Project" (LANL 2000, 066802, p. 2-41).

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 (LANL 2000, 066802, 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.

2.1.3 Geology and Soil

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 (section 4.1). 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 installation work plan (LANL 2000, 066802, 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, 047588).

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 fingerlike mesas in the Los Alamos area (Figure 2.1-2) 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 million years 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, 066802, pp. 2-23 and 2-24).

Figure 2.1-2 Topography of the Los Alamos area

On the western part of the Pajarito Plateau, the Bandelier Tuff overlaps onto the Tschicoma Formation, which consists of older volcanic rock that 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 sediment 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 soil on mesa tops and alluvial deposits of variable thickness on canyon floors.

Soil

Soil erodability is important to understanding the potential for contaminant transport and accurately completion the “contaminant transport information” in site-specific scoping evaluations (section 4.1). Soil on the Pajarito Plateau were initially mapped and described by Nyhan et al. (1978, 005702). A large variety of soil 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 soil (Nyhan et al. 1978, 005702). General patterns of soil erosion rates are summarized by the following text from section 2.2.1.6 of the installation work plan (LANL 2000, 066802, 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 soil and the canyon bottoms consist of relatively coarse sediment. Finer-textured soil of mesa tops is prone to overland runoff whereas soil fines may accumulate in canyon bottoms. The latter are subject to mobilization during flood events.

2.1.4 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, 055622).

The Rio Grande is the highest-order river 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, 066802).

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 installation work plan provides a detailed discussion of this subject (LANL 2000, 066802, pp. 2-28 to 2-37).

2.1.5 Biology

The biota within the Laboratory includes approximately 500 plant species, 29 mammal species, 200 bird species, 19 reptile species, 8 amphibian species, and 1000s of insect species (LANL 2000, 066802). 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, 062887) and five state-protected T&E species (Loftin and Haarmann 1998, 062881) have been identified at the Laboratory (LANL 1999, 062887). 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, 062887). Results of preliminary risk assessments for the Mexican spotted owl, bald eagle, and southwestern willow flycatcher are available in Gallegos et al. (1997, 057915); Gonzales et al. (1997, 062879); Gonzales (1998, 062349); Gonzales (1998, 062350); and Gonzales et al. (2004, 085207). 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 a Laboratory report (LANL 1999, 062887) and in Loftin (1998, 062881*).*

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, 087150). 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. Although the vegetation might have changed since this time due to factors such as fire and drought, the overall cover types remain the same. The version of the vegetation land cover map in Plate 1 is based on the eight taxonomic vegetation classes (Table 2.1-1) 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, 087150). 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, 087150). The approximate areal extent of each cover type on Laboratory property is provided in Table 2.1-1. The risk assessor who conducts scoping verifies the vegetation cover type during the site visit that supports the site-specific problem scoping.

Table 2.1-1
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, 087150, 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, 087150). 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, soil 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 asl (8900 ft to 9950 ft asl). 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 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 colonizing species that invade disturbed areas. 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 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*). An understory of Gambel oak or grasses and bare ground may occur 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*); bare soil may be under the trees or an understory of Basin big sage (*Artemisia tridentate)* and blue grama grass (*Bouteloua gracilis*) may grow.

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 2.1-1.

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, the town of White Rock, and some TAs, are also part of the vegetation land cover class.

2.1.6 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 soil, and the hydrology as described in 10 Code of Federal Regulations 1022, “Compliance with Floodplain and Wetland Environmental Review Requirements.” 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. Wetland 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, 062891). 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 installation work

plan. Figure 2.2-15 in the installation work plan provides a map of the Laboratory showing wetlands by location and type (LANL 2000, 066802, p. 2-40).

2.2 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 2.2-1 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 relevant to a particular site should be discussed in the applicable, site-specific reports.

2.2.1 Hydrologic Transport

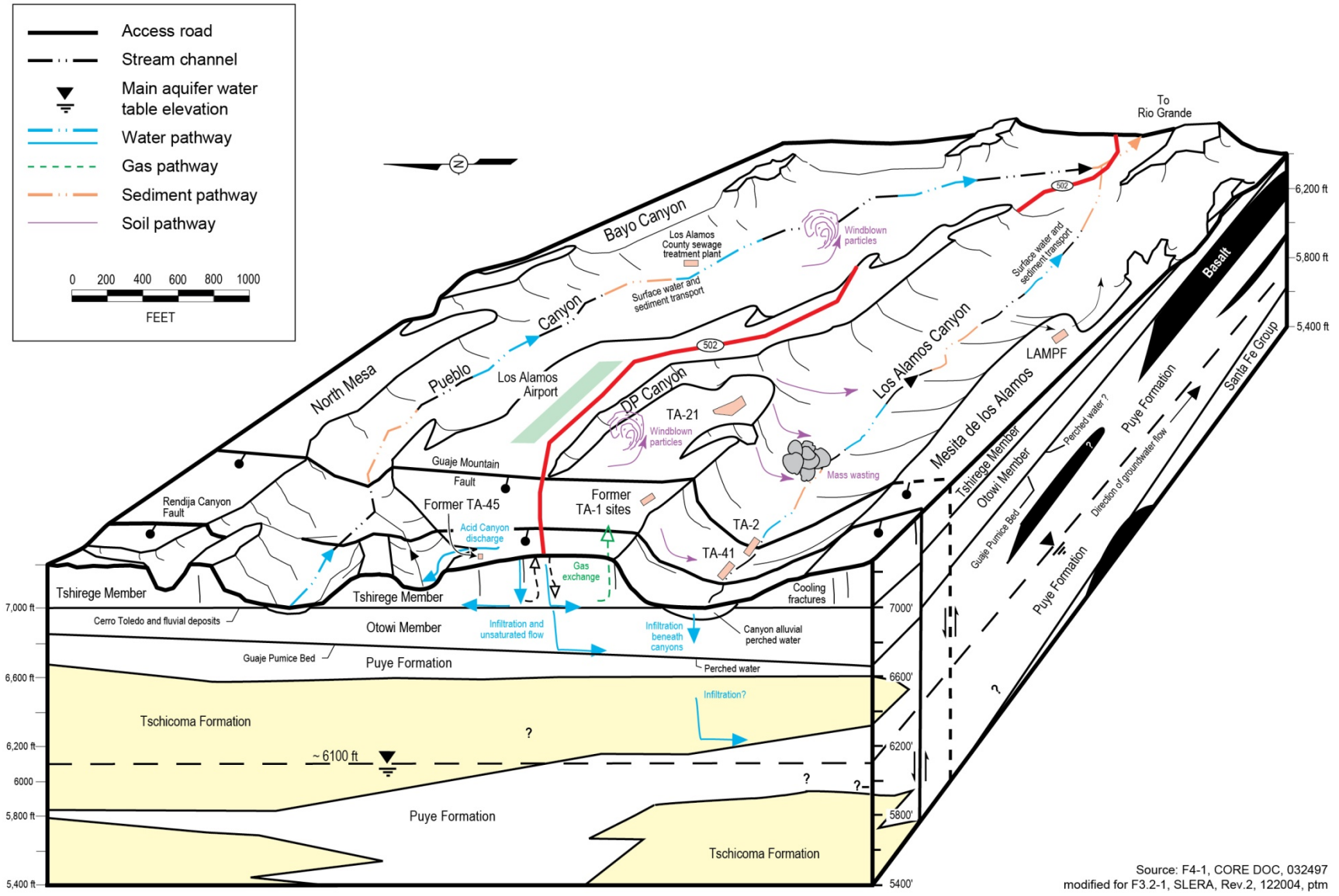
2.2.1.1 Surface Water and Sediment Transport

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 1 million tons of suspended sediment to Cochiti Reservoir (LANL 1997, 055622).

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 Cerro Grande (2000) and Las Conchas (2011) fires. In addition to an increase in the mass of sediment transported in the years following the fires, the concentrations of both nonradionuclides and radionuclides in sediment also increased significantly (e.g., Kraig et al. 2002, 085536). The sediment is transported downstream and 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 the mobilized sediment would mix with post-fire material and other upstream sediment sources.



Source: F4-1, CORE DOC, 032497
modified for F3.2-1, SLERA, Rev.2, 122004, ptm

Figure 2.2-1 Key transport pathways

2.2.1.2 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, 055622). 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 sediment 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.

2.2.2 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.

2.2.3 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.

2.3 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 sediment 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 sediment
- Rain splash or saltation-creep of contaminated sediment onto plants
- Dermal contact with surface water or sediment
- 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 sediment 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

2.4 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 associated relationships and is important for selecting receptors that may be vulnerable to contaminants by virtue of dietary exposure. 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 not sensitive to potentially similar feeding habits among taxa. Figures 2.4-1 and 2.4-2 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.

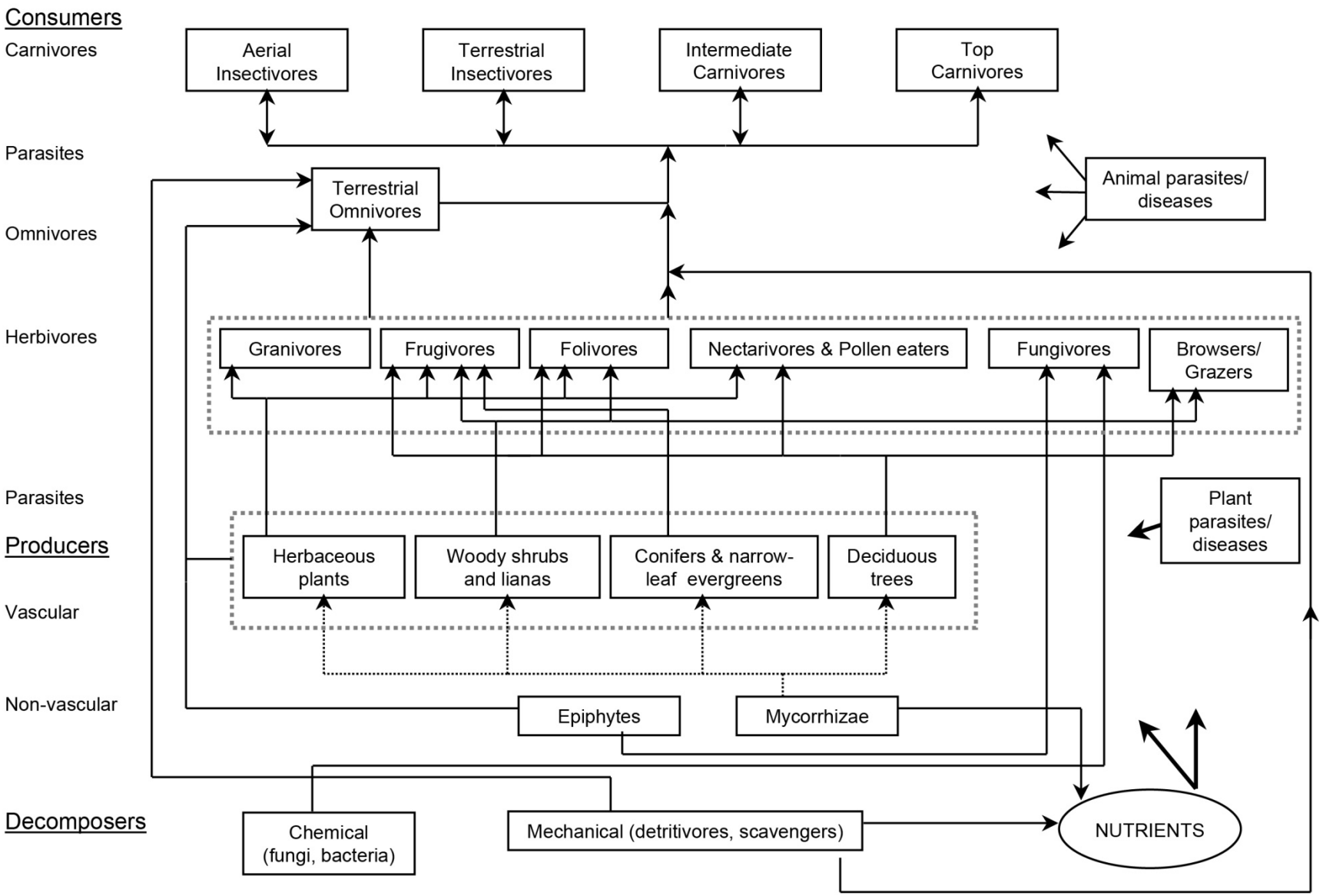


Figure 2.4-1 Terrestrial food web based on feeding relationships of the biota in Los Alamos and on the Pajarito Plateau

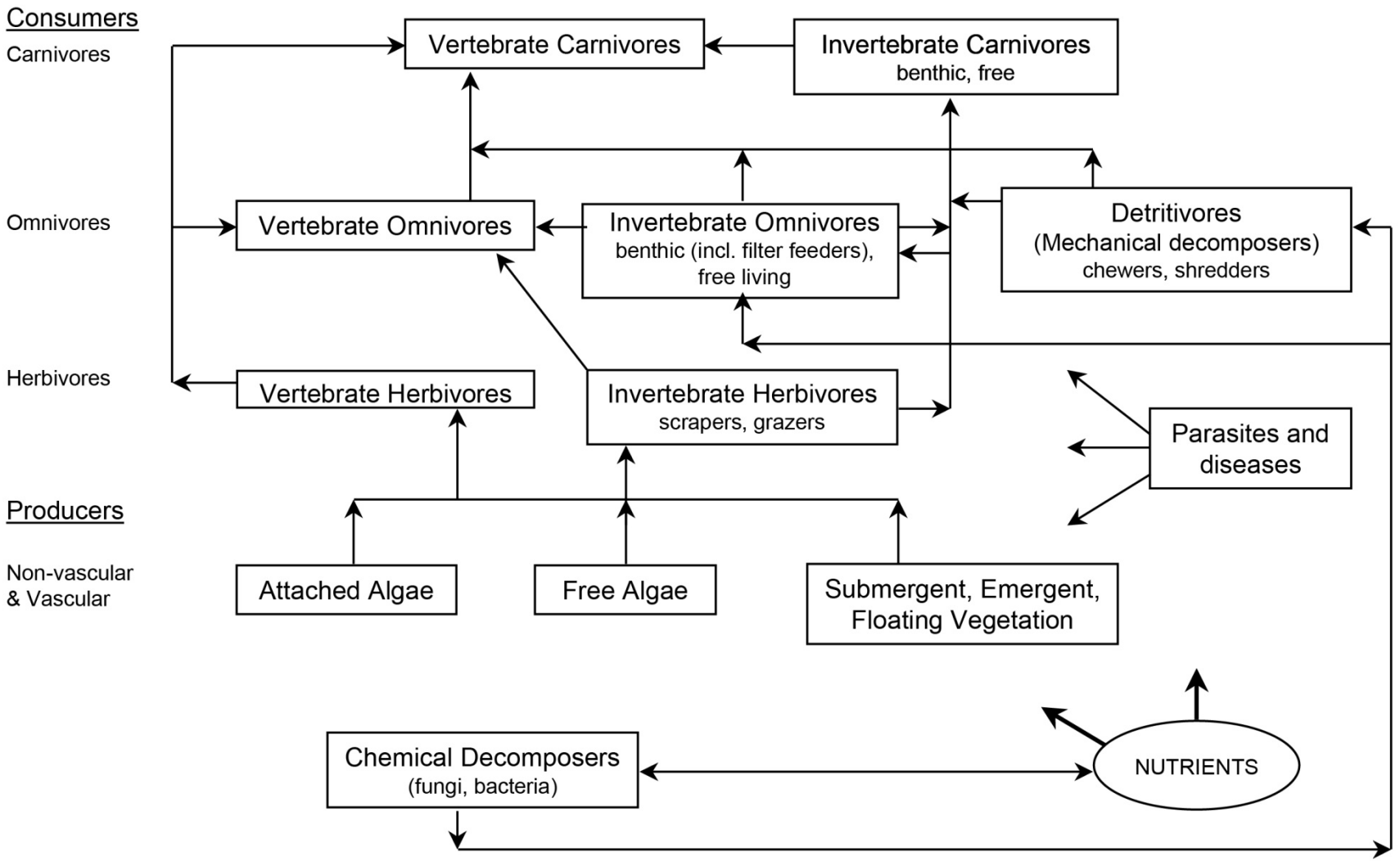


Figure 2.4-2 Aquatic food web based on feeding relationships of the biota in Los Alamos and on the Pajarito Plateau

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 2.4-1 shows a terrestrial food web for the Laboratory, and Figure 2.4-2 is a food web specific to Laboratory aquatic habitats.

As shown on 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 2.6. Separate screening receptors are developed for the terrestrial and aquatic food webs described in section 2.6 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 screening level ecological risk assessment (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 2.4-1, enclosing a number of guilds in a single rectangle, represent broad categories for which a single member may suffice as a screening receptor.

2.5 Assessment Endpoints

To represent the feeding guilds in the food webs as described in section 2.4, 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, 062809). In a screening-level assessment, assessment endpoints are any adverse effects on ecological receptors, where receptors are populations and communities (EPA 1997, 059370).

Superfund guidance also indicates 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 or treaty-protected species (EPA 1999, 070086). 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, 064137) to ensure 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 2.6, are chosen to represent each functional group. The receptor species were chosen based on their presence at the site, their sensitivity to the chemicals of potential concern (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 selection of receptors and assessment endpoints is 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.

2.6 Screening Receptors

As described in section 2.1, 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 2.4. Receptor selection facilitates the determination of potential adverse ecological impacts across the Laboratory and satisfies the following criteria (based on Fordham and Reagan 1991, 063081):

- 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, 059384]).
- The home range (HR) 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 HR 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 with links to 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 selected methods 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. For example, the application of benchmarks for water is targeted at protecting roughly 95% of all aquatic organisms, and thus is inclusive of all trophic guilds illustrated in Figure 2.4-2. The use of benchmarks for screening aquatic environments is recommended in EPA guidance (EPA 1996, 062792).

Terrestrial Receptors

Table 2.6-1 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. 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.

Table 2.6-1
List of Receptors Selected for Screening at the Laboratory

Receptor Category	Receptor Species	Selection Factors
Terrestrial autotroph (producer)	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 available 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 soil Conservative choice for this category, given the food intake to body weight ratio (see section 4.2)

Table 2.6-1 (continued)

Receptor Category	Receptor Species	Selection Factors
Top carnivore	Red fox	Exposure data available Addresses potential biomagnification from soil Conservative choice for this category, given the food intake to body weight ratio (see section 4.2)
Aquatic community, water	Multiple	Typically sensitive organisms so that ecological screening levels (ESLs) are broadly protective of most aquatic species Food source for aquatic animals Ubiquitous and abundant Exposure and toxicity data available
Aquatic community, sediment	Invertebrates	Food source for aquatic animals Ubiquitous and abundant Exposure and toxicity data available
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 higher trophic levels 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 higher trophic levels 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 and 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 as a representative of T&E bird species at the Laboratory, namely 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 sediment 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 because the habitats surrounding water are favorite hunting areas. 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 (small mammals like rabbits and foxes) that may occupy burrows at sites with subsurface vapor-phase COPCs present. Gophers spend most of their time underground. Although small mammals like the deer mouse and shrew have smaller body weights and higher weight-normalized air inhalation rates, these species spend much less time underground relative to the gopher. Thus, pocket gophers are a protective representative for all the burrowing mammal species.

Figure 2.6-1 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., EPA 1993, 059384).

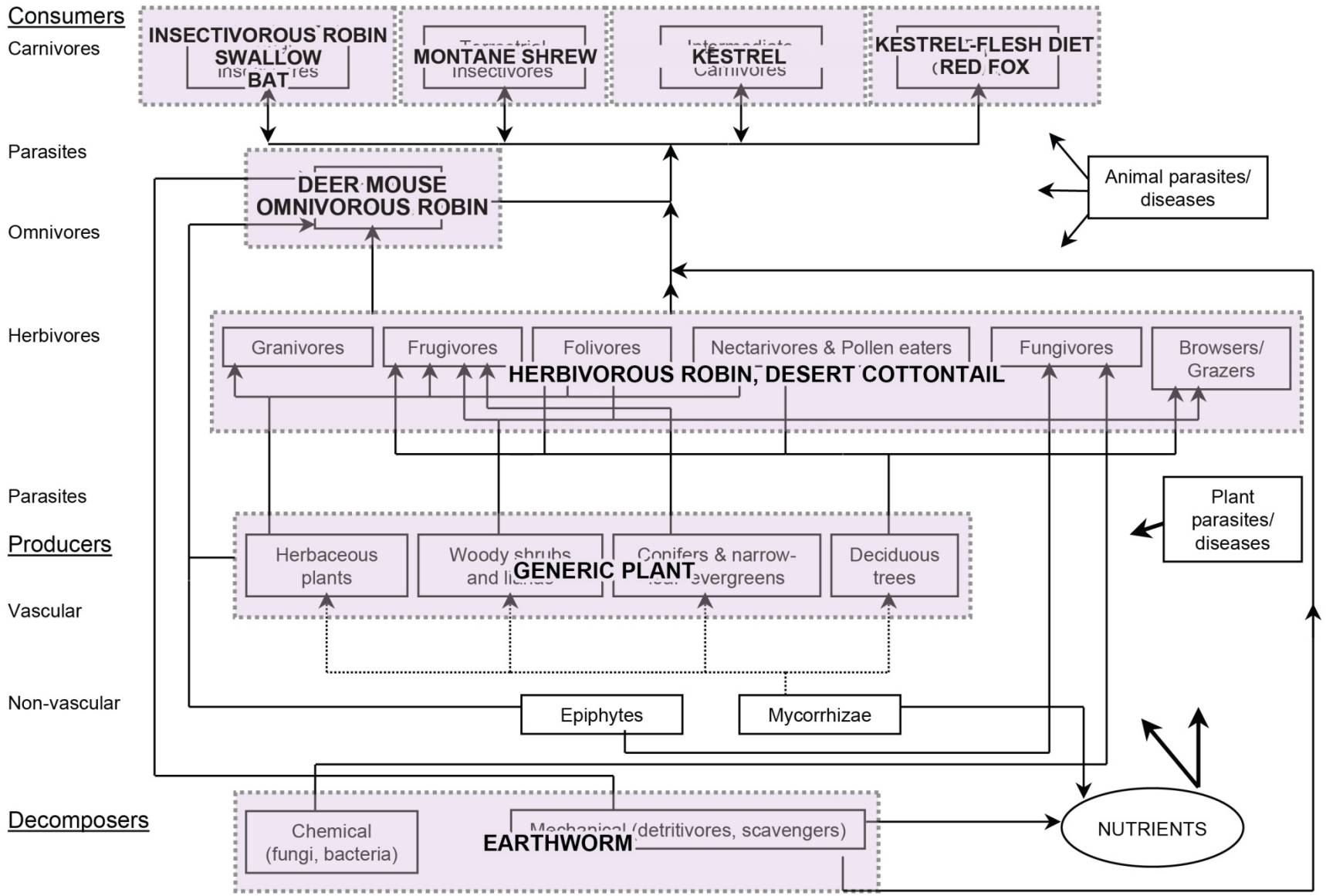


Figure 2.6-1 Screening receptors for terrestrial food web for nonradionuclides and radionuclides at Los Alamos

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., 60 Federal Register 15366, "Final Water Quality Guidance for the Great Lakes System, Final Rule"; EPA 1996, 062792) and others (e.g., Jones et al. 1997, 059813) to be protective of aquatic organisms at large (plants, invertebrates, and vertebrates). Although few vertebrates reside in the aquatic realms of the Laboratory, it was considered prudent to adopt methods that are otherwise considered broadly protective and that include organisms that may be found in the Rio Grande (e.g., fish). The aquatic food web, as shown in Figure 2.4-2, 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.

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, although fish are not found on Laboratory property, a "generic" bony fish was selected to represent intermediate carnivores exposed to contaminants that may potentially enter the Rio Grande.

Figure 2.6-2 shows the aquatic food web with a box representing each screening receptor species superimposed over the feeding guilds represented by that receptor. No direct representative is available 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.

3.0 DERIVATION OF ESLS

Sections 3.4 and 3.5 describe the methods used to derive ESLs for nonradiological and radiological COPCs, respectively, for soil, sediment, and water. 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 2011, 206473, 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. More details on the basis for the derivation of ecological screening levels are presented in Appendix A.

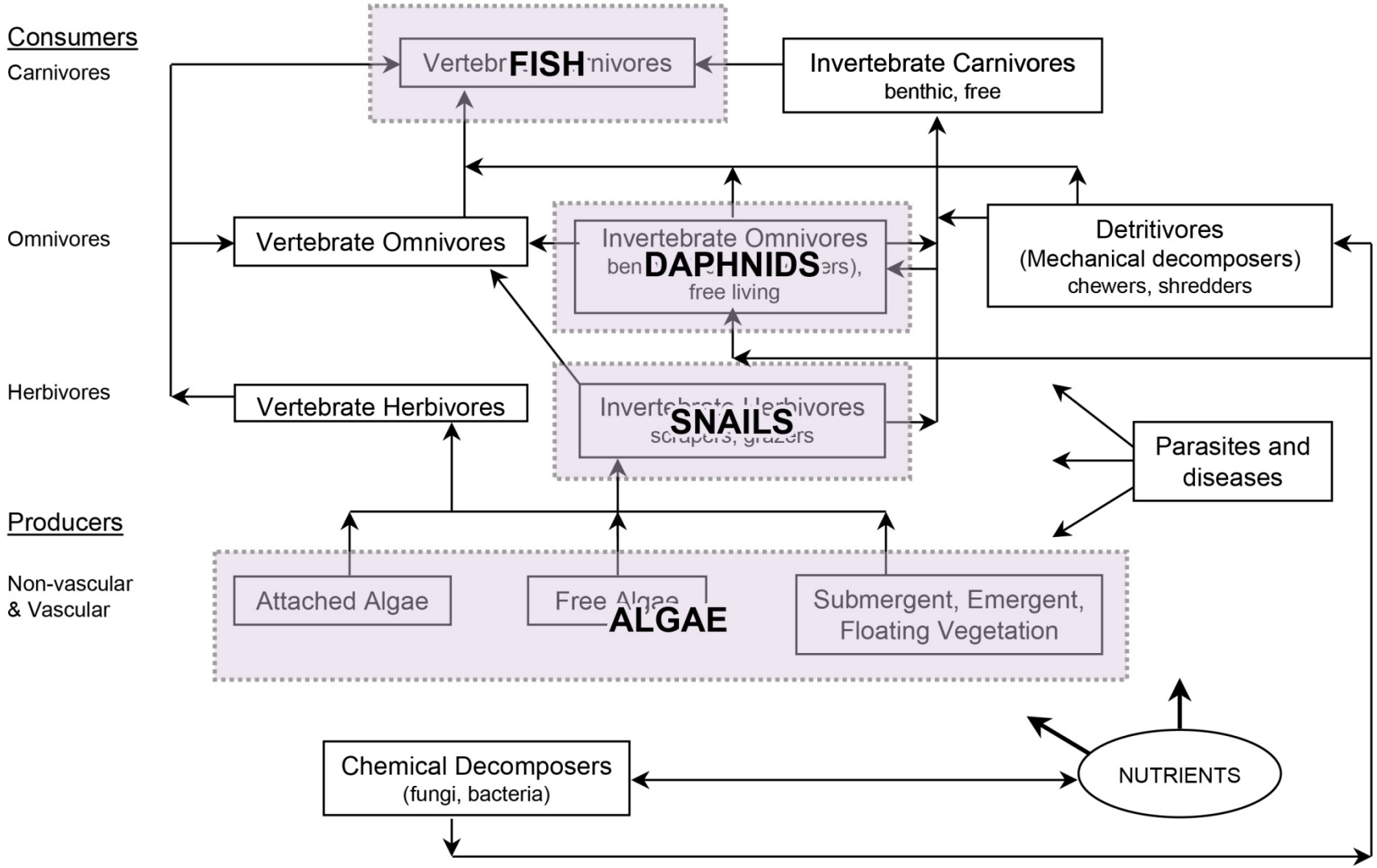


Figure 2.6-2 Screening receptors for aquatic food web for radionuclides

3.1 Ecological Effects of Concern for Screening

The effects of ecological concern or critical life stage effects are those that adversely affect reproduction, development, growth, and survival of organisms. Table 3.1-1 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; rather, it shows only 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 2011, 206473, or latest version).

Table 3.1-1
Relevant Ecological Effects for Terrestrial Receptors

Receptor	Effect Category		
	Reproduction/Development	Survival	Weight Change/ Size Change
Plant	Percent germination, seedling emergence, root or shoot growth	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

The 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. The effects may be morphologically and/or physiologically mediated. The effects on juveniles are associated with exposure to a chemical during pre- or postfertilization and/or during pre- and postembryonic development. The 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 from invertebrates and plants, the developmental effects of chemical exposure are also morphologically and/or physiologically mediated and may directly or indirectly compromise behavior (excepting plants), 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 the effects on growth and survival of the first generation and any other successive generations are considered developmental effects until the organism reaches maturity.

3.2 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, meaning that at some low level, doses of a COPC have no effect on an organism. Conversely, extremely high doses lead to a saturation of effects (e.g., 100% mortality). For radionuclides, a threshold dose is not assumed; it is assumed, however, that a negligible dose exists for which the risk is acceptable.

Most ecological screening assessments use the no observed adverse effect level (NOAEL) or no observed effect concentration (NOEC) as the maximum acceptable exposure value. This value is also used as the TRV. The dose limit for radionuclides is 0.1 rad per day (IAEA 1992, 062802). 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, 059370).

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 most of the COPC exposure to these organisms is not related to dietary pathways. Instead, it is assumed 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.

3.3 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 3.3-1 and is based on EPA's general wildlife exposure models (EPA 1993, 059384).

$$E_{total} = E_{oral} + E_{dermal} + E_{respiration} \qquad \textbf{Equation 3.3-1}$$

Where E_{total} is total exposure to a COPC (units are mg/kg/d)

E_{oral} is oral exposure (diet and direct ingestion of contaminated media, with units of mg/kg/d)

E_{dermal} is dermal exposure (with units of mg/kg/d)

$E_{respiration}$ is exposure through respiration or inhalation (with units of mg/kg/d)

For terrestrial wildlife inhabiting the soil surface, it is assumed most contaminant exposure to nonradiological chemicals is through the oral exposure pathway (Sample et al. 1998, 062807). The dermal contact pathway is not typically assessed quantitatively in ecological risk assessments, based on guidance indicating the ingestion route is most important to terrestrial animals (EPA 1997, 059370). 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, 085643). 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 volatile organic compounds (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 3.3-2.

$$E_{total} = E_{oral}$$ **Equation 3.3-2**

Although the oral pathway is dominant in most cases, the site-specific scoping should assess the potential importance of the dermal and respiration/inhalation pathways. In cases where dermal and respiration may represent significant exposure pathways, the models presented by Hope (1995, 062783) should be used to evaluate these pathways. The oral exposure model used for terrestrial wildlife is from EPA's Wildlife Exposure Factors Handbook (EPA 1993, 059384) and is provided in Equation 3.3-3:

$$E_{oral} = C_{soil} \cdot I_{soil} \cdot AUF_{soil} + C_{water} \cdot I_{water} \cdot AUF_{water} \cdot (1/d_{water}) + C_{food} \cdot I_{food} \cdot AUF_{food}$$ **Equation 3.3-3**

Where E_{oral} is the estimated oral daily dose for a COPC (mg/kg/d)

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 • d], simplified to kg/kg/d 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 • d], simplified to kg/kg/d 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 • d], simplified to kg/kg/d 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. Soil ingestion is calculated from a fraction of the dietary intake of soil (EPA 1993, 059384). 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 a 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, 059384). 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 3.3-1. The information provided in Table 3.3-1 is for the screening receptors adopted by the Laboratory. It is also important to note that exposure parameters provided in Table 3.3-1 represent conservative upper estimates of potential exposure. More realistic exposure information may be considered in the uncertainty analysis. Information about body weight and inhalation rates, which are not required by Equation 3.3-1, is provided to assist with alternate forms of the wildlife exposure model. For example, the exposure models discussed by Hope (1995, 062783) require these additional parameters.

Table 3.3-1
Measures Required for the Wildlife Exposure Model

Receptor	Parameter	Value	Units	Reference	Notes
American kestrel	Body weight	0.103	kg	EPA 1993, 059384, p. 2-112	Lowest male average weight was 103 g used to provide more conservative ESL value
	Food intake[a]	0.099	kg-food dry wt/kg-body wt/d	EPA 1993, 059384, p. 2-112	Used higher of two empirical fresh weight food intake values, 0.31 kg-food fresh wt/kg-body wt/d, multiplied by (100–68)% to account for food moisture content
	Food moisture content	0.68	Proportional	EPA 1993, 059384, p. 4-13	Diet includes insects, birds, mammals, other (see EPA 1993, 059384, p. 2-113) (value assumes mammals, birds)
	Fraction soil in diet	0.02	Unitless	none	Default value
	Soil invertebrate diet[b]	0.5 (0)	Unitless	EPA 1993, 059384, p. 2-113	Rounded EPA value to 50% to equally expose receptor to potentially contaminated invertebrates and flesh
	Flesh diet[b]	0.5 (1)	Unitless	EPA 1993, 059384, p. 2-113	Rounded EPA value to 50% to equally expose receptor to potentially contaminated invertebrates and flesh
	Home range	106	ha	EPA 1993, 059384	Average of all HR data for woods, forests, and agricultural areas
	Population area	4240	ha	Calculated	40 times HR (see text for explanation)
American robin	Body weight	0.077	kg	EPA 1993, 059384, p. 2-197	Lowest weight was 77 g used to provide a conservative ESL
	Food intake[a]	0.35	kg-food dry wt/kg-body wt/d	EPA 1993, 059384, p. 2-197	Higher of two empirical values fresh weight food intake rate for robins feeding primarily on fruits, 1.52 kg-food fresh wt/kg-body wt/d, multiplied by (100–77)% to account for food moisture content
	Food moisture content	0.77	Proportional	EPA 1993, 059384, pp. 4-13,14	Diet includes invertebrates, plants (fruits), assumed fruit

Table 3.3-1 (continued)

Receptor	Parameter	Value	Units	Reference	Notes
American robin (continued)	Fraction soil in diet	0.1	Unitless	Beyer et al. 1994, 062785, Table 1	Used woodcock value, most similar of birds evaluated
	Plant diet[c]	0, 0.5, or 1	Unitless	None	Modeled with three diets: herbivore, omnivore, and insectivore
	Soil invertebrate diet[c]	1, 0.5, or 0	Unitless	None	Modeled with three diets: herbivore, omnivore, and insectivore
	Home range	0.42	ha	EPA 1993, 059384, p. 2-199	HR data represent average territory size in an open, semi urban environment
	Population area	16.8	ha	Calculated	40 times HR (see text for explanation)
Deer mouse	Body weight	0.020	kg	EPA 1993, 059384, p. 2-295	For females that have lower body weights and therefore are used to provide a conservative ESL
	Food intake[a]	0.20	kg-food dry wt/kg-body wt/d	EPA 1993, 059384, p. 2-296	Based on empirical fresh weight food intake of 0.22 kg-food fresh wt/kg-body wt/d (diet of lab chow, 8–10% moisture), multiplied by (100–10)% to account for food moisture
	Food moisture content	0.1	Proportional	EPA 1993, 059384, p. 2-296	Moisture content of lab chow used to determine food intake
	Fraction soil in diet	0.02	Unitless	Beyer et al. 1994, 062785, Table 1	For white-footed mouse, most closely related of species available
	Plant diet	0.5	Unitless	EPA 1993, 059384, 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, 059384, 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, 059384, p. 2-298	Average of data from representative environments
	Population area	3.0	ha	Calculated	40 times HR (see text for explanation)
Eastern cottontail as a surrogate for desert cottontail	Body weight	0.900	kg	EPA 1993, 059384, p. 2-355	Average of range of reported values for desert cottontail
	Food intake[a]	0.093	kg-food dry wt/kg-body wt/d	Nagy 1987, 062782	Estimated as 95% UCL using Nagy (1987, 062782) allometric scaling formula for herbivores
	Fraction soil in diet	0.024	Unitless	Beyer et al. 1994, 062785, Table 1	For meadow vole, most ecologically similar species of those available in table
	Plant diet	1	Unitless	EPA 1993, 059384, p. 2-356	Assume strict herbivore diet
	Home range	3.1	ha	EPA 1993, 059384, p. 2-357	Average of all HR data for a woodlot and for mixed habitats
	Population area	124	ha	Calculated	40 times HR (see text for explanation)

Table 3.3-1 (continued)

Receptor	Parameter	Value	Units	Reference	Notes
Montane shrew	Body weight	0.015	kg	EPA 1993, 059384, p. 2-213	Lowest weight of 15 g used to provide a conservative ESL
	Food intake[a]	0.198	kg-food dry wt/kg-body wt/d	EPA 1993, 059384, p. 2-213	Higher of two empirical fresh weight food intakes, 0.62 kg–food fresh wt/kg-body wt/d, multiplied by (100–68)% to account for food moisture in diet of beef liver
	Food moisture content	0.68	Proportional	EPA 1993, 059384, p. 4-13	Laboratory feeding study used beef liver
	Fraction soil in diet	0.1	Unitless	Beyer et al. 1994, 062785, Table 1	Used woodcock
	Soil invertebrate diet	1	Unitless	EPA 1993, 059384, p. 2-214	Assume strict insectivore diet
	Home range	0.39	ha	EPA 1993, 059384, p. 2-214	Reported average HR for one environment.
	Population area	15.6	ha	Calculated	40 times HR (see text for explanation)
Pocket gopher	Body weight	0.104	kg	Gonzales et al. 2000, 085653	Laboratory-specific minimum measured field value used to provide a conservative ESL
	Inhalation rate	0.089	m^3/d	EPA 1993, 059384, p. 3-12	Calculated from body weight by Equation 3-20 in EPA (1993, 059384)
	Home range	0.06	ha	EPA 1993, 059384, p. 2-214	Reported HR of up to 700 yd^2 (Controlling Pocket Gophers in New Mexico, New Mexico State University Guide L-109; http://aces.nmsu.edu/pubs/_l/L-109.pdf)
	Population area	2.4	ha	Calculated	40 times HR (see text for explanation)
Red fox	Body weight	3.94	kg	EPA 1993, 059384, p. 2-224	Lowest of four mean values used to provide a conservative ESL
	Food intake[a]	0.045	kg-food dry wt/kg-body wt/d	EPA 1993, 059384, p. 2-224	Female after whelping, empirical fresh weight food intake is 0.14 kg–food fresh wt/kg-body wt/day for an unknown diet, multiplied by assumed food moisture content (100–68)%
	Food moisture content	0.68	Proportional	EPA 1993, 059384, p. 4-13	Mean value for mammals and passerine birds
	Fraction soil in diet	0.03	Unitless	Beyer et al. 1994, 062785, Table 1	For red fox
	Flesh diet	1	Unitless	EPA 1993, 059384, p. 2-224	Rounded diet to 100% flesh
	Home range	1038	ha	EPA 1993, 059384, p. 2-226	Average of all HR data over a variety of unspecified environments
	Population area	41520	ha	Calculated	40 times HR (see text for explanation)
Violet-green swallow	Body weight	0.0139	kg	Dunning 1993, 073795	Average body weight of females for *Tachycineta thalassina*
	Food intake[a]	0.268	kg-food dry	Nagy 1987, 062782	Estimated as 95% UCL using Nagy

Table 3.3-1 (continued)

Receptor	Parameter	Value	Units	Reference	Notes
			wt/kg-body wt/d		(1987, 062782) allometric scaling formula for passerines
	Fraction soil or sediment in diet	0	Unitless	None	Assume no soil or sediment exposure for aerial insectivores
	Invertebrate diet	1	Unitless	None	Assume 100% invertebrate diet
	Home range	0.68	ha	Bowman 2003, 087148	Using general allometric equation of 10^[1.8+log(BW) × 1.06]
	Population area	27.2	ha	Calculated	40 times HR (see text for explanation)
Occult little brown myotis bat	Body weight	0.0088	kg	Whitaker 1980, 062889	Used midpoint of reported body weight range for *Myotis lucifugus* (3.1 g to 14.4 g)
	Food intake[a]	0.159	kg-food dry wt/kg-body wt/d	Nagy 1987, 062782	Estimated as 95% UCL using Nagy (1987, 062782) allometric scaling formula for all mammals
	Food moisture content	0.69	Proportional	EPA 1993, 059384, p. 4-13	Used value for grasshoppers and crickets as surrogate for emergent aquatic insects
	Fraction soil or sediment in diet	0	Unitless	None	Assume no soil or sediment exposure for aerial insectivores
	Invertebrate diet	1	Unitless	None	Assume 100% invertebrate diet
	Home range	100	ha	Menzel et al. 2003, 087151	Minimum of 100- to 500-ha HR given for southeastern myotis bat
	Population area	4000	ha	Calculated	40 times HR (see text for explanation)

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

[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 one modeled as a strict insectivore.

Table 3.3-1 presents information on the spatial scales for exposure to the representative receptors. The HR reflects the area from which individuals may be exposed to contamination. However, EPA guidance is to manage the ecological risk to populations rather than to individuals, with the exception of T&E species (EPA 1999, 070086). One approach to addressing the potential effects on populations is to estimate the spatial extent of the area inhabited by the local population that overlaps with the contaminated area. The population area for each receptor is based on the individual receptor HR and its dispersal distance (Bowman et al. 2002, 073475). Bowman et al. (2002, 073475) estimate that the median dispersal distance for mammals is 7 times the linear dimension of the HR (i.e., the square root of the HR area). If only the dispersal distances for the mammals with HRs within the range of the screening receptors are used, the median dispersal distance becomes 3.6 times the square root of the HR ($R^2 = 0.91$) (Bowman et al. 2002, 073475). If it is assumed the receptors can disperse over the same distance in any direction, the population area is circular and the dispersal distance is the radius of the circle. Therefore, the population area for each receptor can be derived by $\pi(3.6\sqrt{HR})^2$ or approximately 40HR. Table 3.3-1 presents receptor population areas based on 40HR.

3.4 ESLs for Chemicals

This section provides an overview of the approach used to develop ESLs for nonradionuclides for soil, burrow air, sediment, and water. Table 3.4-1 summarizes the receptors and diet compositions used in equations for ESL development for each exposure medium.

Table 3.4-1
Ecological Screening Receptors for Chemicals

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	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 used only 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.

[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.

[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.

3.4.1 TRV and ESL Development

ESLs are used to evaluate potential hazards associated with chemicals and radionuclides. The Laboratory has developed chemical-, media-, and receptor-specific ESLs using a tiered TRV development approach, as described in the ECORISK Database guidance (LANL 2011, 206473). ESLs are developed and maintained by the Laboratory as part of the ECORISK Database, which archives the ESLs, TRVs, associated exposure parameters, and all supporting documentation.

The development of an ESL is a two-step process. The first step involves identifying or developing a TRV. In the second step, the TRV and exposure parameters are used to calculate ESLs for chemicals and ecological receptors representative of the ecosystems at the Laboratory. Eleven different receptors were selected to be representative of mammals, birds, plants, and invertebrates inhabiting terrestrial and aquatic ecosystems at the Laboratory (Table 3.4-1). At the time of this publication, 159 analytes, including inorganic chemicals, organic chemicals, and radionuclides, have ESLs documented in the database.

A TRV represents an exposure rate associated with an acceptable risk from chronic exposure of an ecological receptor to a specific contaminant via a specific exposure pathway. In other words, exposures exceeding the TRV may pose adverse effects to wildlife species, while exposures below the TRV are not expected to result in adverse effects (EPA 2005, 089448).

TRVs are important parameters in ESL calculations because “they represent the component of the model that determines whether a contaminant in a media may present potential harm to ecological receptors in the area” (Podolsky et al. 2001, 072586). For any given chemical, TRV values vary among government agencies and private sectors because the methods used to develop them vary according to the site-specific concerns of the organization that developed them (i.e., receptor species, chemical, type of exposure pathway, type, and magnitude of uncertainty factors applied).

The ideal TRV for ecological risk-screening assessments is one that is based on literature representing the most ecologically relevant effects (reproduction/development, survival and/or adult weight/size change); exposure routes (oral ingestion via food or drinking water for birds and mammals, inhalation for mammals, uptake via seed coat and/or roots for plants, and direct contact exposure for invertebrates and aquatic community organisms); exposure media (food and drinking water for birds and mammals, air for mammals, soil for plants and invertebrates, and water and sediment for aquatic community organisms); exposure period (chronic); and effect levels (NOAEL for vertebrates or NOEC for plants and invertebrates). A TRV based on these characteristics is considered protective of the wildlife; aquatic community, plant, and invertebrate populations; and sensitive individuals because it represents an exposure that is not associated with adverse impacts of low-level, long-term chemical effects (i.e., adverse effects on ability of individuals to develop into viable organisms, search for mates, breed successfully, and produce live and equally viable offspring). Therefore, NOAELs and NOECs are used for the ESL meant to be protective of all receptors and levels of biological organization (e.g., individuals and populations). To provide information for a bounding analysis of the potential for ecological risks, low effect ESLs or L-ESLs are also presented in the ECORISK Database. L-ESLs are based on lowest observed adverse effect levels [LOAELs] for vertebrates or lowest observed effect concentrations [LOECs] for plants and invertebrates.

Laboratory guidance (LANL 2010, 110623) includes guidelines for the literature search, data extraction, default value assignment, and exception ruling for various fields of data entry in customized PTSE databases, PTV calculation, and TRV derivation. Before performing a PTSE, the primary toxicity literature for the organism and for the exposure pathway and chemical scenario of concern must be identified and collected. As a result, the appendix begins with guidelines for literature searches and retrieval.

ESLs are chemical- and medium-specific screening levels pertaining to a given receptor (e.g., avian omnivore, earthworm) and medium (sediment, soil, water, and/or air). The TRV is used in the receptor specific ESL calculation. This equation converts the toxicity value from a dose (mg-contaminant/kg body weight/d) to an environmental concentration (e.g., mg-contaminant/kg-soil) using factors to estimate the transfer of chemical from soil, sediment, or water to dietary media (e.g., soil-to-plant transfer factor [TF]) and receptor specific exposure parameters (e.g., ingestion/inhalation rates and body weight). In the case of plants, earthworms, and aquatic organisms, the TRV is equal to the ESL because the toxicity value is already in environmental concentration units.

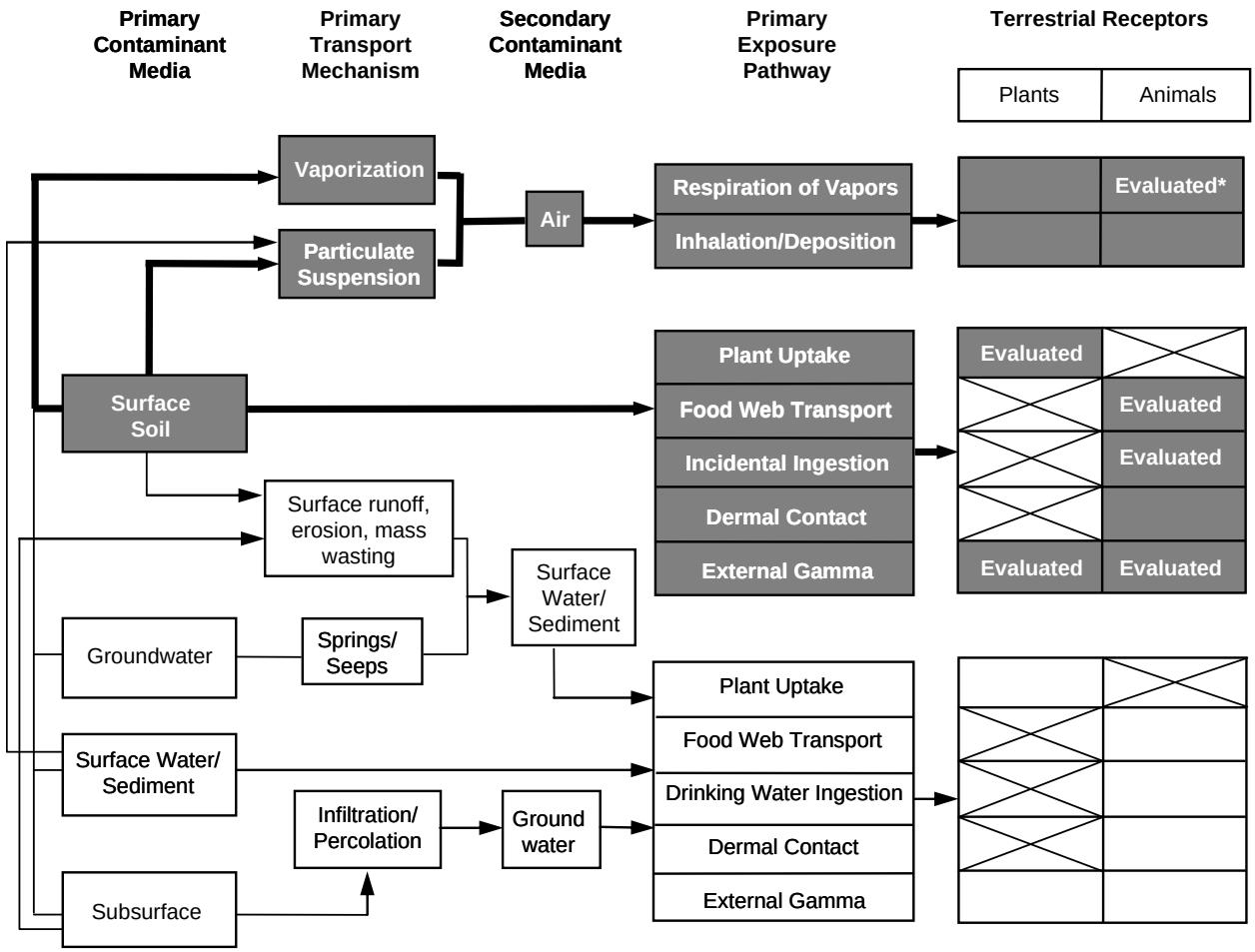
3.4.2 Soil ESLs

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-size particles and by development of ferric hydroxides (LANL 1998, 059730). For the purposes of ecological risk screening, imported fill or disturbed soil is evaluated as well-developed soil because the exposure and transport pathways are similar. 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, 073791).

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, the 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, 059384). The transport and exposure pathways likely to be complete for sites with soil contamination are shown in Figure 3.4-1. Pathways included in all the ESL calculations are designated as “evaluated” in this figure. The pathway for respiration of air vapors 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 soil.

The minimum soil ESL for each COPC is the lowest receptor-specific soil ESL value available among plants, invertebrates, robin, kestrel, shrew, mouse, cottontail, and fox. For plants and invertebrates, the soil ESL is the NOEC and the L-ESL is the LOEC. Information supporting the selected effect level is provided in the ECORISK Database (LANL 2011, 206473, 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 wildlife L-ESL is the soil concentration of the COPC that results in an exposure dose equal to the LOAEL. The full derivation of soil ESLs is presented in Appendix A.



Notes: 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 3.4-1 Ecological CSM for soil pathways

The mathematical basis for calculating wildlife ESLs for herbivore, omnivore, insectivore, and carnivore functional groups is shown in Equations 3.4-1 through 3.4-5.

$$ESL_{ij} = \frac{TRV_{ij}}{I_i \cdot [\, fs_i + fp_i \cdot TF_{plant,\,j}\,]} \qquad \textbf{Equation 3.4-1}$$

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

TRV_{ij} is the toxicity reference value (NOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/kg-body wt/d)

I_i is the normalized daily dietary ingestion rate for herbivore *i* (kg-food dry wt/kg body wt/d])

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

fp_i is the fraction of plants in diet for 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)

$$ESL_{ij} = \frac{TRV_{ij}}{I_i \cdot [\, fs_i + fp_i \cdot TF_{plant,j} + fi_i \cdot TF_{invert,j} \,]}$$ **Equation 3.4-2**

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

TRV_{ij} is the toxicity reference value (NOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/kg-body wt/d)

I_i is the normalized daily dietary ingestion rate for omnivore *i* (kg –food dry wt/kg body wt/d)

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,* expressed as a fraction of the dietary intake

$TF_{plant,i}$ 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 in diet for omnivore *i,* expressed as a fraction of the dietary intake

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

$$ESL_{ij} = \frac{TRV_{ij}}{I_i \cdot [\, fs_i + fi_i \cdot TF_{invert,j} \,]}$$ **Equation 3.4-3**

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

TRV_{ij} is the toxicity reference value (NOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/kg-body wt/d)

I_i is the normalized daily dietary ingestion rate for insectivore *i* (kg-food dry wt/kg body wt/d)

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

fi_i is the fraction of invertebrates in diet for insectivore *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 dry weight soil)

$$ESL_{ij} = \frac{TRV_{ij}}{I_i \cdot [\, fs_i + ff_i \cdot TF_{flesh,j} \,]}$$ **Equation 3.4-4**

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

TRV_{ij} is the toxicity reference value (NOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/kg-body wt/d)

I_i is the normalized daily dietary ingestion rate for carnivore *i* (kg –food dry wt/kg body wt/d)

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

ff_i is the fraction of flesh in diet for 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 dry weight soil)

$$ESL_{ij} = \frac{TRV_{ij}}{I_i \cdot [\, fs_i + ff_i \cdot TF_{flesh,j} + fi_i \cdot TF_{invert,j}\,]}$$ **Equation 3.4-5**

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

TRV_{ij} is the toxicity reference value (NOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/kg-body wt/d)

I_i is the normalized daily dietary ingestion rate for carnivore/insectivore *i* (kg-food dry wt/kg-body wt/d)

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

ff_i is the fraction of flesh in diet for carnivore/insectivore *i,* expressed as a fraction of the dietary intake

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

fi_i is the fraction of invertebrates in diet for carnivore/insectivore *i,* expressed as a fraction of the dietary intake

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

The wildlife ESL models (functional group–specific Equations 3.4-1 through 3.4-5) show the ESL as proportional to the effect level. Thus, larger values of the TRV lead to larger ESL values, which indicate 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 TRV value as a receptor with lower exposure. The wildlife L-ESLs are calculated with Equations 3.4-1 through 3.4-5 and using the LOAEL for the TRV term.

3.4.3 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 vapor-phase contaminants are at much lower concentrations in surface air (Markwiese et al. 2003, 087149), 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. Appendix A provides additional information on the basis for the burrow air ESL. It is assumed the gopher stays in its burrow 100% of the time; the exposure through air is described by Equation 3.4-6:

$$E_{air} = \frac{C_{air} \cdot I_{air}}{BW}$$ **Equation 3.4-6**

Where E_{air} is the estimated inhalation daily dose for a COPC (mg-COPC/kg-body wt/d)

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

I_{air} is the daily inhalation rate for the pocket gopher (m^3/d)

BW is the body weight for the pocket gopher (kg)

Therefore, the ESL can be expressed as shown in Equation 3.4-7:

$$ESL_j = \frac{TRV_j \cdot BW}{I_{air}}$$ **Equation 3.4-7**

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

TRV_j is the NOAEL for burrow animal inhalation and COPC *j* (mg-COPC/kg-body wt/d)

BW is the body weight for the pocket gopher (kg)

I_{air} is the daily inhalation rate for the pocket gopher (m^3/d)

The wildlife L-ESLs can be calculated with Equation 3.4-7 and using the LOAEL for the TRV term.

3.4.4 Sediment ESLs

Geomorphologists define sediment 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, 059730). This definition includes sediment in active channels, inactive channels, and floodplain geomorphic settings. Sediment can also be found in lentic systems (ponds or lakes), but no lakes and few ponds exist on Laboratory property. Inactive channel and floodplain sediment typically have associated terrestrial ecological communities and, therefore, are more akin to soil from an ecological risk evaluation perspective. Thus, soil ESLs apply to inactive channel and floodplain sediment. Aquatic ecological communities are often associated with perennial and seasonally intermittent aquatic environments and, therefore, sediment-based ESLs are applicable to active channel and pond geomorphic settings with developed aquatic communities.

Because of the typical association of sediment 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 sediment, 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 sediment on aquatic organisms. Only limited modeling is needed to develop sediment ESLs. Modeling is used to evaluate potential effects of contaminated sediment 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 soil; infiltration of surface water into shallow and/or deep groundwater; mass wasting; and wind-driven transport of soil-borne COPCs into water courses/bodies. Of primary concern are the first three transport mechanisms, which are included in Figures 3.4-2 and 3.4-3. Rare instances where mass wasting or wind-blown soil 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.

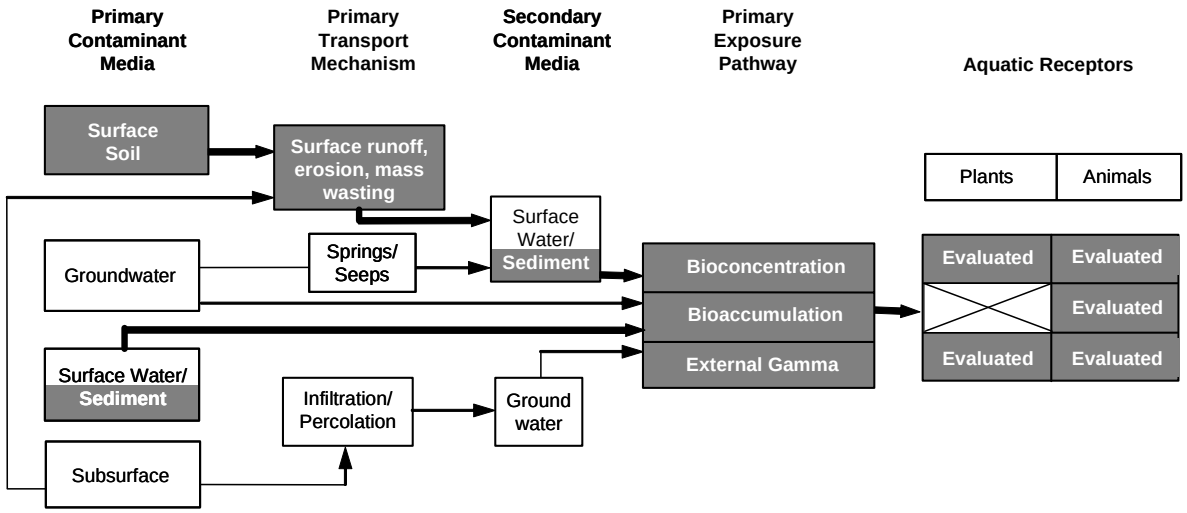
Protecting sediment quality is increasingly viewed as a logical extension of water-quality protection, which helps to emphasize the interrelationship between sediment and water as exposure media. Chapman (1989, 062902) cites several reasons for the requirement of sediment ESLs, including

- various toxic contaminants found only in trace amounts in the water column that accumulate in sediment to elevated levels;
- sediment that serves as both a reservoir and a source of contaminants to the water column;
- sediment that integrates 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
- sediment that is 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 sediment conform to those proposed by the EPA for developing ecotoxicity thresholds (EPA 1996, 062792). Methods for screening sediment 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 sediment). Thus, the exposure pathways for aquatic receptors (using EPA methods) include bioconcentration and, for radionuclides only, external gamma exposure (Figure 3.4-2). 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 of species found in aquatic environs at large.

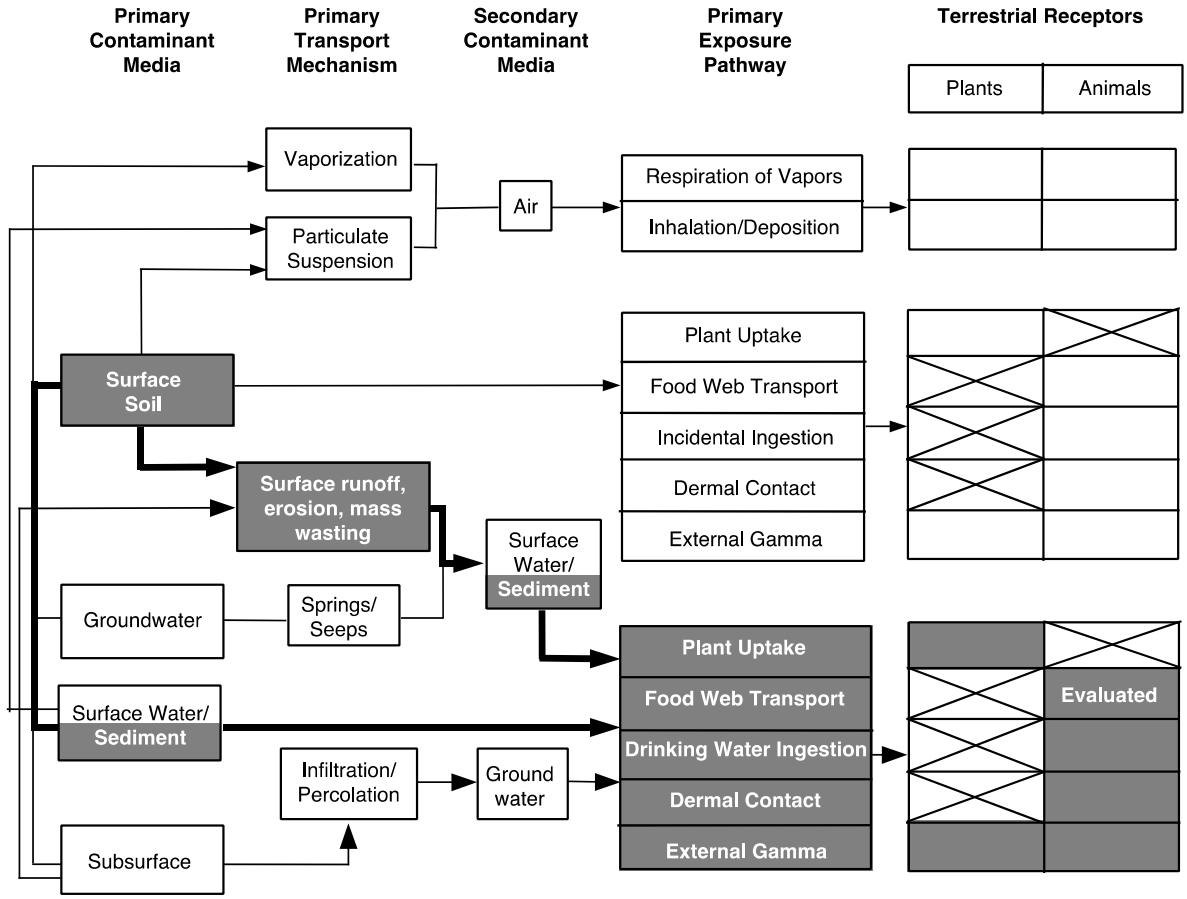
Although sediment ESLs are primarily developed to protect against potential effects on aquatic receptors, pathways from sediment to terrestrial receptors are also evaluated to ensure that bioaccumulation concerns have been addressed. A simple wildlife exposure model is developed to evaluate bioaccumulation potential of COPCs in sediment to aerial insectivores (bat and swallow) via emergent insects. The terrestrial receptor exposure model for sediment pathways is provided in Figure 3.4-3. This conceptual model indicates several exposure pathways are complete, but only the food web transport pathway is evaluated because other pathways make only minor contributions. Additionally, the uptake of COPCs from sediment 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 3.4-2) and screening methods.

Sediment ESLs come from a variety of sources but not all of the benchmarks are equal because they may be derived from different measurement endpoints. Further information on sediment benchmark selection for the aquatic community is provided in Appendix A.



Notes: 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 3.4-2 Aquatic CSM for sediment pathways



Notes: 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 3.4-3 Terrestrial CSM for sediment pathways (to account for bioaccumulation concerns)

3.4.4.1 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 3.4-3, which is the insectivore soil ESL model described in section 3.4.1. The model shown in Equation 3.4-7 is based on the transfer of contamination from sediment 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 Appendix A. Contaminant transfer to higher level carnivores is not accounted for by these ESLs and should be addressed in the uncertainty analysis.

$$ESL_{ij} = \frac{TRV_{ij}}{I_i \cdot fi_i \cdot TF_{invert,j}} \qquad \textbf{Equation 3.4-8}$$

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

TRV_{ij} is the toxicity reference value (NOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/kg-body wt/day)

I_i is the normalized daily dietary ingestion rate for insectivore *i* (kg-food dry wt/kg-body wt/day])

fi_i is the fraction of invertebrates in diet for insectivore *i*

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

The aerial insectivore sediment L-ESLs are calculated with Equation 3.4-8 and using the LOAEL for the TRV term.

3.4.4.2 Summary of Sediment ESLs Derivation

Sediment ESLs may be derived from a variety of sources and more than one ESL may be available for any given constituent (Appendix A). Additionally, ESLs are developed for aerial insectivores based on models that differ from those used to derive ESLs for the sediment aquatic community in general. In screening, the ESL for a given constituent is compared with the ESL derived for all sediment receptors and the lowest of the values is used as the sediment ESL to ensure that bioaccumulation concerns are addressed by the minimum ESL.

3.4.5 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 because this is considered the bioavailable fraction of these constituents in water (EPA 1996, 062792).

Methods for screening water are based on exposure pathways to aquatic and terrestrial organisms. For aquatic organisms, the screening approach assumes they 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 to calculate water-quality criteria 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 3.4-4. To evaluate potential effects of contaminated water on terrestrial receptors, a wildlife exposure model is developed (Figure 3.4-5). 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.

The consideration of impacts from waterborne contamination to aquatic receptors requires the evaluation of a number of water-quality criteria or benchmarks, which come from a variety of sources, all based upon toxicological information from primary studies. These criteria differ in the methods for their development and/or in the rigor of their development. Consequently, water-quality criteria or benchmarks must be evaluated in a hierarchical fashion, based upon evaluation of their conservatism or certainty for the protection of approximately 95% of aquatic species. More information on water ESLs for the aquatic community is provided in Appendix A.

3.4.5.1 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 3.4-1, 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 3.4-9**

Where E_{water} is the estimated oral daily dose for a COPC (mg-COPC/kg-body wt/d)

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 • d])

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

$$ESL_{ij} = \frac{1000 \cdot TRV_{ij}}{I_i}$$ **Equation 3.4-10**

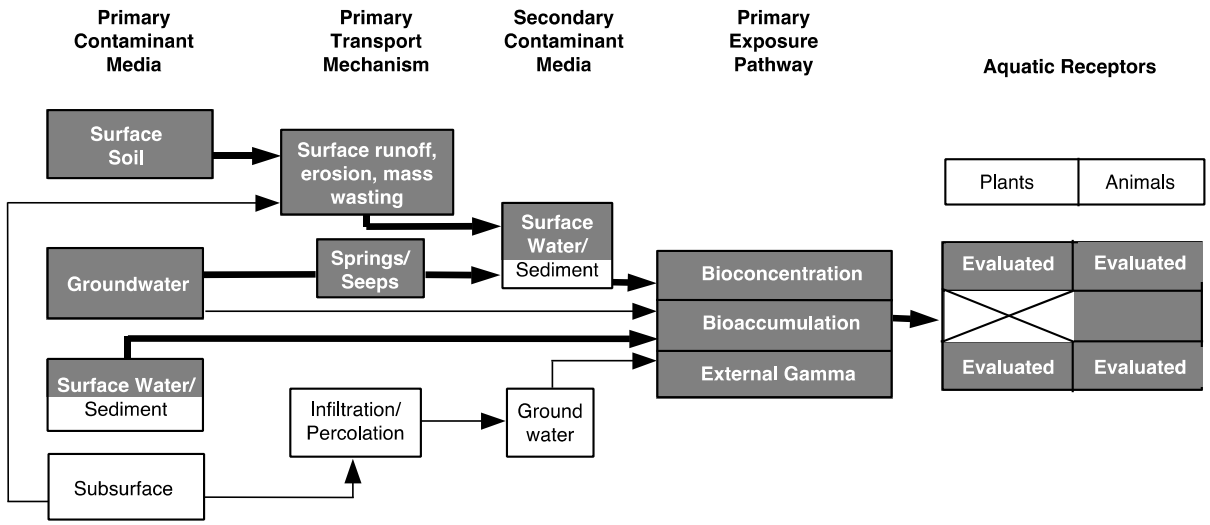
Where ESL_{ij} is the water ESL for wildlife species *i* and COPC *j* (µg/L)

1000 is the number of µg per mg

TRV_{ij} is the toxicity reference value (NOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/kg-body wt/d)

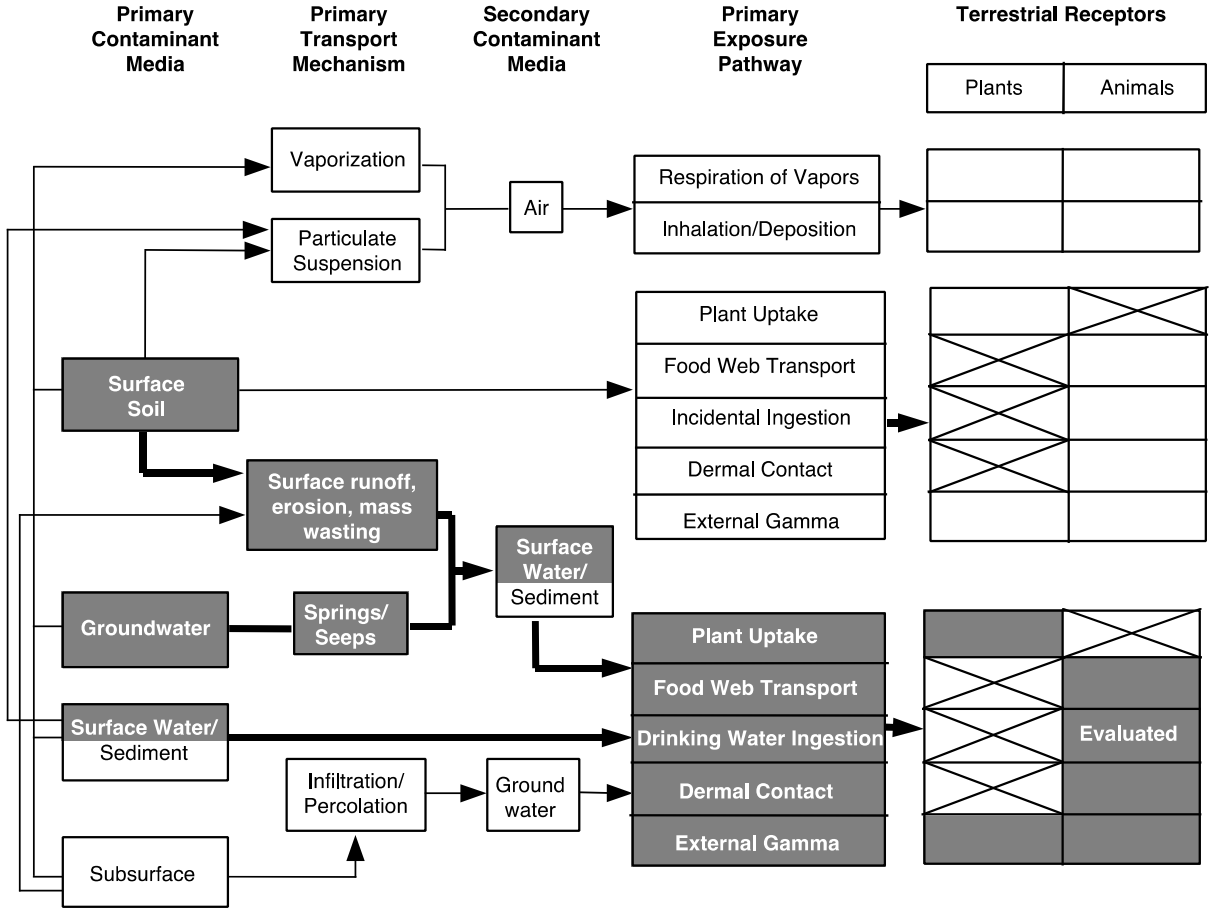
I_i is the daily water ingestion rate for wildlife species *i* (L of water/kg body weight/d)

The main parameters are summarized in Table 3.3-1. The wildlife water L-ESLs are calculated with Equation 3.4-10 and using the LOAEL for the TRV term.



Notes: 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 3.4-4 Aquatic CSM for water pathways



Notes: 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 3.4-5 Terrestrial CSM for water pathways

3.4.5.2 Summary of Water ESLs Derivation

Water ESLs are selected using water-quality criteria or benchmarks in the order presented below:

1. Section 20.6.4.900 of the New Mexico Administrative Code (NMAC) 20.6.4.900;
2. Ambient Water-Quality Criteria set forth by EPA (2009, 109328);
3. National Oceanic and Atmospheric Administration Screening Quick Reference Tables (Buchman 2008, 206414);
4. Other sources.

Values reported as chronic are used for the ESLs and those reported as acute are used for L-ESLs.

3.5 ESLs for Radionuclides

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, 085637) and "RESRAD-BIOTA: A Tool for Implementing a Graded Approach to Biota Dose Evaluation, User's Guide, Version 1" (DOE 2004, 085639). However, the methods are consistent with DOE guidance and with the conceptual basis presented by NMED for evaluating ecological effects of radionuclides (NMED 2000, 087104).

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, 085637). The first tier provides only a single screening value for each medium (soil, sediment, or water) and is thus similar to the minimum 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 minimum 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; the minimum 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

$$Dose = Effective\ Energy \bullet Amount \qquad \textbf{Equation 3.5-1}$$

Much of the confusion in calculating radionuclide dose relates to the units of the terms in Equation 3.5-1. For calculating radionuclide ESLs, "dose" is expressed in units of rad/d, 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/d 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. The same receptor species are used to model terrestrial exposure to radionuclides and nonradionuclides with the exception that aquatic receptors for radionuclides consist of four specific groups (algae, daphnids, snails, and fish); aquatic ESLs for nonradionuclides are based on standards and benchmarks that are considered to be broadly protective of all aquatic species.

3.5.1 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, 062802):

- 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/d. 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, 062803) and Eisler (1994, 063043), 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

$$\textit{Total Acceptable Dose} = 0.1 \text{ (rad/d)} = \textit{Internal Dose} + \textit{External Dose} \qquad \textbf{Equation 3.5-2}$$

DOE has also recommended 0.1 rad/d as the dose limit for wildlife, but DOE has specified 1 rad/d 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, 085637). Thus, the Laboratory has selected a more protective dose limit for plant and aquatic receptors as the no effect level for the ESL. For the L-ESLs, the laboratory increases the DOE's dose limit by a factor of 10.

3.5.2 Soil ESLs

The operational definition of soil was provided in section 3.4.1. Radionuclide soil ESLs are based on exposure of terrestrial receptors to contaminated soil. The minimum radionuclide soil ESL is the lowest receptor-specific ESL among the 11 terrestrial receptors. ESLs are developed to account for dose from a single radionuclide.

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 3.4-1). 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, 062804). The basic model for calculating acceptable dose from soil for radionuclides is

$$Dose_j = C_{organism,j} \cdot DCF_{int,j} + C_{soil,j} \cdot DCF_{ext,j}$$ **Equation 3.5-3**

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

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

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

Internal dose results from exposure to radionuclides through plant uptake, incidental soil ingestion, and food web uptake (Figure 3.4-2). External dose is based on exposure to gamma-emitting radionuclides from contaminated soil (Figure 3.4-2). The basis for calculating internal and external dose is provided in Appendix A.

3.5.2.1 Calculations of Soil ESLs

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

$$ESL = \frac{Dose\ Limit}{TF_{plant,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 3.5-4**

Where Dose limit is 0.1 rad/d

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

$DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (rad/d per pCi/g-fresh tissue) (Appendix A)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g dry soil, assuming 360-degree exposure) (Appendix A)

For terrestrial invertebrate receptors, the ESL equation is written as

$$ESL = \frac{Dose\ Limit}{TF_{invert,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 3.5-5**

Where Dose limit is 0.1 rad/d

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

$DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (rad/d per pCi/g-fresh tissue) (Appendix A)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g dry soil, assuming 360-degree exposure) (Appendix A)

For terrestrial herbivores, the ESL equation is written as

$$ESL = \frac{Dose\ Limit}{[I_{soil,i} + TF_{plant,j} \cdot I_{plant,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 3.5-6**

Where Dose limit is 0.1 rad/d

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

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

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

$TF_{blood,j}$ is the food to blood transfer factor for radionuclide *j* (unitless)

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

$DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (rad/d per pCi/g-fresh tissue) (Appendix A)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g dry soil, assuming 180- or 360-degree exposure) (Appendix A)

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

$$ESL = \frac{Dose\ Limit}{[I_{soil,i} + TF_{invert,j} \cdot I_{invert,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 3.5-7**

Where Dose limit is 0.1 rad/d

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

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

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

$TF_{blood,j}$ is the food to blood transfer factor for radionuclide *j* (unitless)

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

$DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (rad/d per pCi/g-fresh tissue) (Appendix A)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g dry soil, assuming 180- or 360-degree exposure) (Appendix A)

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

Equation 3.5-8

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

Where Dose limit is 0.1 rad/d

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

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

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

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

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

$TF_{blood,j}$ is the food to blood transfer factor for radionuclide *j* (unitless)

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

$DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (rad/d per pCi/g fresh tissue) (Appendix A)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g dry soil assuming 180- or 360-degree exposure) (Appendix A)

For terrestrial carnivores, the ESL is calculated as

$$ESL = \frac{Dose\ Limit}{[I_{soil,i} + TF_{flesh,j} \cdot I_{flesh,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation 3.5-9**

Where Dose limit is 0.1 rad/d

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

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

$I_{flesh,i}$ is the normalized daily flesh ingestion rate for organism *i* (g of flesh-fresh wt/g of body wt/d)

$TF_{blood,j}$ is the food to blood transfer factor for radionuclide *j* (unitless)

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

$DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (rad/d per pCi/g fresh tissue) (Appendix A)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g-dry soil, assuming 180- or 360-degree exposure) (Appendix A)

Nonradionuclide and radionuclide ESL calculations share many common variables. Thus, much of the discussion concerning uncertainty in the nonradionuclide ESLs is directly relevant to the radionuclide ESLs. Three variables, the retention time, the TF from food to blood, and the dose conversion factors, are unique to radionuclides. The retention time and blood TFs 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. The soil radiological L-ESLs are calculated with Equations 3.5-4 through 3.5-9 and using 1 rad/d as the dose limit.

3.5.3 Sediment ESLs

Discussion on the operational definition of sediment was provided in section 3.4.3. 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 minimum radionuclide sediment ESL is the lowest receptor-specific ESL among the four aquatic receptors as well as the bat and swallow. ESLs are developed to account for dose from a single radionuclide.

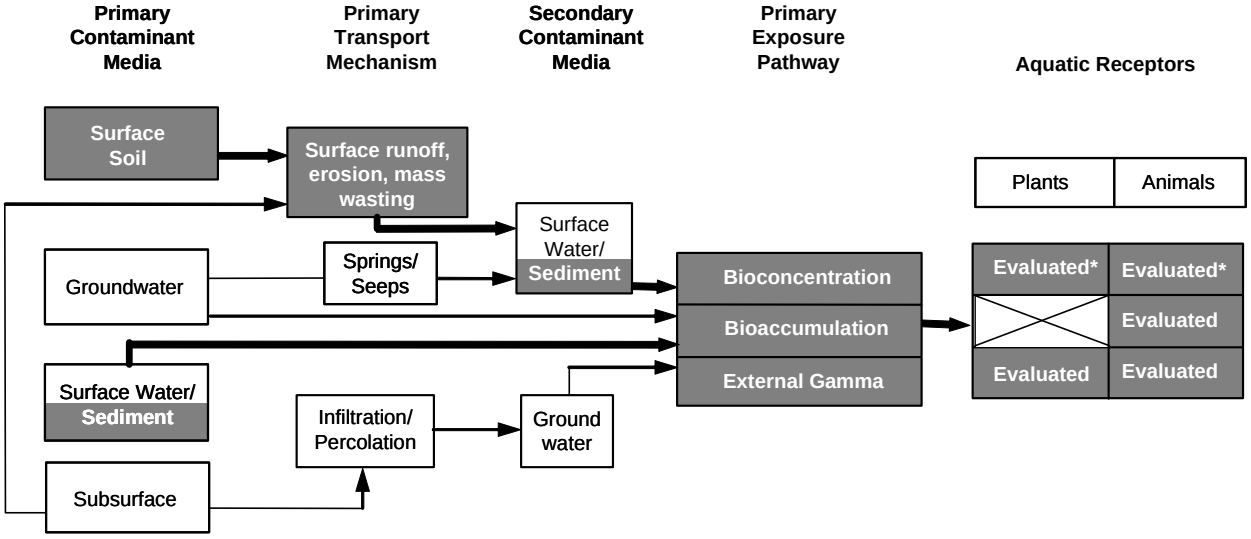
An ESL calculation for aquatic organisms exposed to sediment is based on the models presented by Baker and Soldat (1992, 062801). The radiological dose to aquatic organisms is the external dose from the radionuclide in sediment; the internal dose from sediment radionuclides is accounted for in the water ESL calculations for aquatic organisms for radionuclides (Baker and Soldat 1992, 062801; DOE 2002, 085637). Sediment-based thresholds used for screening values do not exist for radionuclides, so algae, daphnids, and snails and fish have been selected as surrogates for organisms living in aquatic environments at the Laboratory. Transport pathways from sediment to aquatic organisms are presented in Figure 3.5-1. 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 3.4-3. The basic model for calculating acceptable dose from sediment for radionuclides is

$$Dose_j = C_{sediment,j} \cdot DCF_{ext,j} \qquad \textbf{Equation 3.5-10}$$

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

$C_{sediment,j}$ is the concentration of radionuclide *j* in sediment (pCi/g dry sediment)

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



Notes: 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 3.5-1 Aquatic CSM for sediment pathways

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

$$ESL = \frac{Dose\ Limit}{BCF_i \cdot DCF_{int,i} + DCF_{ext,j}}$$ **Equation 3.5-11**

Where Dose limit is 0.1 rad/d

BCF_i is the bioconcentration factor for sediment for organism *i* (pCi/g-fresh weight per pCi-COPC/g dry sediment)

$DCF_{int,i}$ is the internal dose conversion factor for sediment and is set to zero for sediments as internal dose is modeled via water exposures

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/day per pCi/g dry sediment assuming 180-degree exposure)

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

$$ESL = \frac{Dose\ Limit}{I_{food,i} \cdot BCF_{invert,j} \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j}}$$ **Equation 3.5-12**

Where Dose limit is 0.1 rad/d

$I_{food,i}$ is the normalized daily dietary ingestion rate for organism *i* (g of invertebrate-fresh wt/g of body wt/d)

$BCF_{invert,j}$ is the invertebrate bioconcentration factor for radionuclide *j* (pCi/g invertebrate-fresh wt per pCi/g dry sediment)

$TF_{blood,j}$ is the food to blood transfer factor for radionuclide *j* (unitless)

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

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

More information on the basis of deriving radionuclide ESLs in sediment is provided in Appendix A. The sediment radiological L-ESLs are calculated with Equations 3.5-11 and 3.5-12 and using 1 rad/d as the dose limit.

3.5.4 Water ESLs

The operational definition of water was discussed in section 3.4.4. Radionuclide water ESLs are based on exposure of contaminated surface water to aquatic and terrestrial receptors. The minimum radionuclide water ESL is the lowest receptor-specific ESL among the four aquatic and eight wildlife receptors. ESLs are developed to account for dose from a single radionuclide. Calculation of ESLs for aquatic organisms is based on the models presented by Baker and Soldat (1992, 062801). 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, 062801; DOE 2002, 085637). 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 selected as assessment endpoint surrogates for receptors living in aquatic environments at the Laboratory. Transport pathways to aquatic organisms are presented in Figure 3.5-2. The only water exposure pathway considered for terrestrial receptors is ingestion of drinking water (Figure 3.4-5). The basic model for calculating acceptable dose from water for radionuclides is

$$Dose_j = C_{organism,j} \cdot DCF_{int,j} + C_{water,j} \cdot DCF_{ext,j} \qquad \textbf{Equation 3.5-13}$$

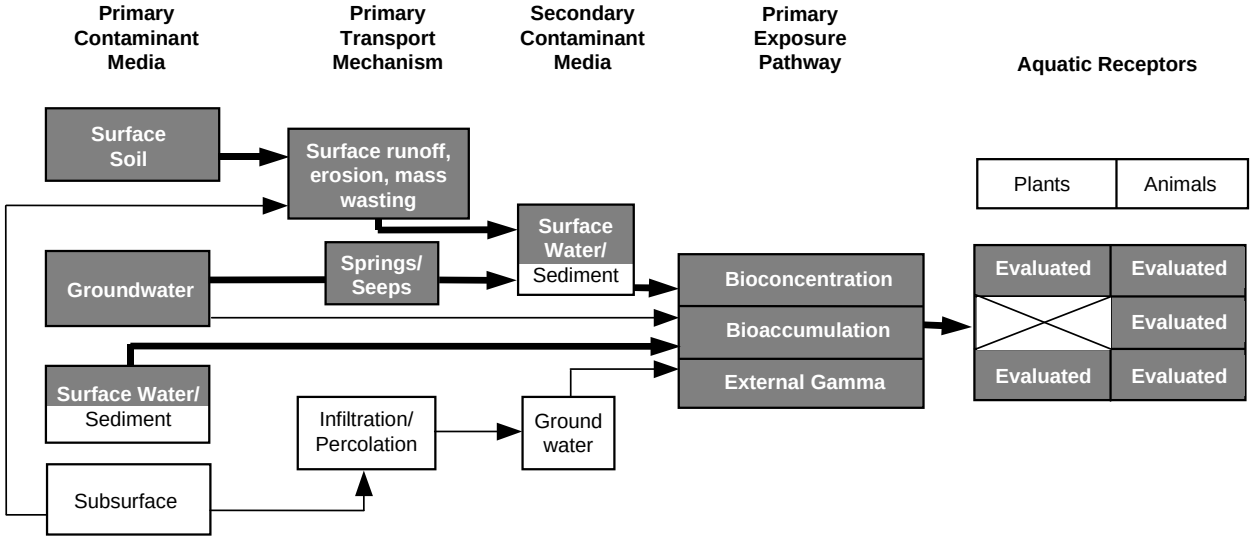
Where $Dose_j$ is the total acceptable dose from radionuclide *j* (rad/d)

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

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



Notes: 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 3.5-2 Aquatic CSM for water pathways

3.5.4.3 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/d to a particular receptor. For aquatic receptors that spend at least part of their lives immersed in water, the ESL calculation is

$$ESL = \frac{Dose\ Limit}{(BCF_{i,j} \cdot DCF_{int,j} + DCF_{ext,j})/1000} \qquad \textbf{Equation 3.5-14}$$

Where Dose limit is 0.1 rad/d

$BCF_{i,j}$ is the bioconcentration factor for organism *i* and radionuclide *j* (pCi/g fresh weight per pCi/mL water)

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

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/mL water, assuming 360-degree exposure)

1000 is the number of mL/L

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

$$ESL = \frac{Dose\ Limit}{(I_{water,i} \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j}) / 1000}$$ **Equation 3.5-15**

Where Dose limit is 0.1 rad/d

I_{water} is the normalized daily water ingestion rate (mL of water/g of body weight per day)

$TF_{blood,j}$ is the water to blood transfer factor for radionuclide *j* (unitless)

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

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

1000 is the number of mL/L

More information on the basis of deriving radionuclide ESLs in water is provided in Appendix A. The water radiological L-ESLs are calculated with Equations 3.5-14 and 3.5-15 and using 1 rad/d as the dose limit.

4.0 SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT

The SLERA 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 and consequently a recommendation to halt further ecological investigation at these sites is included in the applicable risk assessment report.

The SLERA consists of three steps:

1. The scoping evaluation (or problem-formulation) described in section 4.1;
2. The screening evaluation (or the screening-level risk and uncertainty-analysis) described in sections 4.2 and 4.3; and
3. Risk interpretation (or screening-level risk characterization) described in section 4.4.

During the initial step, the ecological risk assessor should determine if COPCs are known or expected to occur at the site. If not, the site should be recommended as requiring no further ecological evaluation and justified in the risk assessment. 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. The information presented in this section is an overview of the SLERA. Assessors are referred to EP-DIV-SOP-10006, Performing Human and Ecological Risk Screening Assessments, or the equivalent procedure for steps involved in performing a SLERA.

4.1 Scoping Evaluation

Sites being investigated by the EP projects 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 and completion of the ecological scoping checklist (Appendix B). The ecological exposure CSM 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 with neighboring SWMUs or AOCs for the purposes of the SLERA. 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. Any aggregation of the SWMUs and/or AOCs under consideration should be established before the SLERA begins.

After the scoping evaluation, if the ecological risk assessor determines the site poses no threat to the environment because 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.

During scoping, a decision is made about the adequacy of the data and the CSM for the screening evaluation. 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. It should be noted that when data are adequate[2] and appropriately distributed, the upper confidence limit (UCL) of the mean concentration may be used instead of the maximum detected concentration in calculations and comparisons. Calculation of UCLs of the mean concentration is done using the EPA ProUCL program (http://www.epa.gov/osp/hstl/tsc/software.htm), which is based on EPA guidance (EPA 2002, 085640).

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 Ecological 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;

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

- Prepare for screening evaluation by determining whether screening should encompass terrestrial and/or aquatic receptors; and
- Gather information to develop the CSM (e.g., what are the dominant/important transport pathways, exposure routes, and receptors).

Completion of the ecological scoping checklist consists of three steps:

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 CSM. 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 CSM 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 and/or TA); (2) information on site erosion potential; (3) investigation work plans or reports that provide information on contamination source, sampling locations, analytical suites, and sampling results; (4) GIS maps that show (if applicable) neighboring SWMUs and AOCs, sampling locations, vegetation types, watershed name, and wetlands; and (5) historical and current aerial photographs to help document changes in site operations and conditions.

Before 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 B). 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, shrub land, 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 (discussion of whether the site should be aggregated with additional SWMUs and/or AOCs for screening); and (7) documentation of other scoping meeting notes (as appropriate).

4.1.1.2 Checklist Step 2: Site Visit

The main objective of the site visit is to confirm 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 sampling 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 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 resampling or reanalysis to obtain detection limits that are appropriate and usable in the ecological screening evaluation. Similarly, if vertical and/or lateral 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.

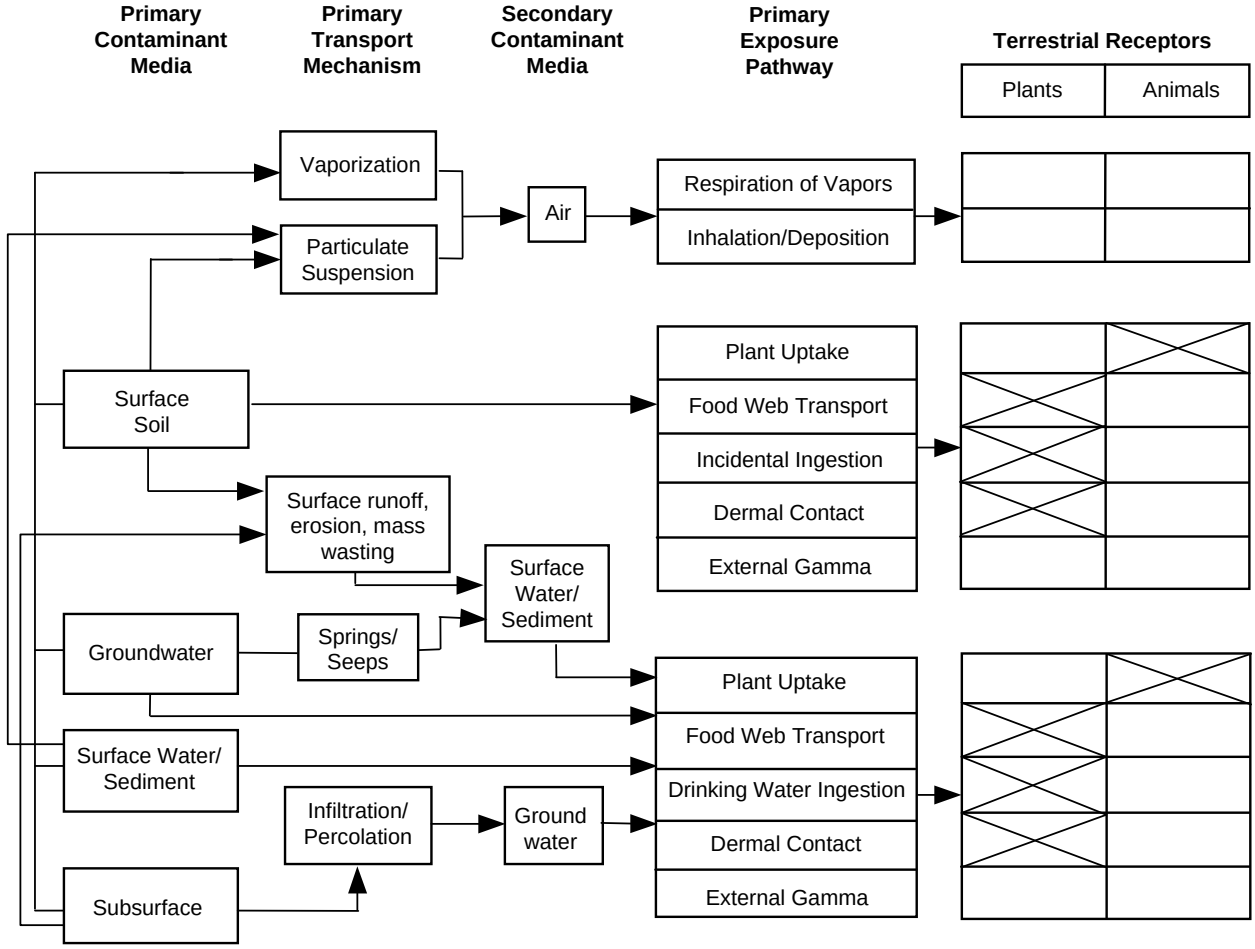
Completion of Part B also includes additional field notes on the site setting and potential ecological receptors to document other site observations relevant to the ecological screening evaluation of the site. Such information may include observations on the variability in the type and density of ecological 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 CSM

Part C of the checklist relates to the CSM 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 (Appendix B). Answers to questions in Part C are used to complete the CSM. 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 CSM 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 soil. Figure 4.1-1 also illustrates the transport pathways that may lead to contaminated air, surface

water/sediment, or groundwater as secondary contaminated media. Two exposure routes are available 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. Five possible exposure routes are available to terrestrial receptors from contaminated soil: plant uptake, food web transport, incidental ingestion, dermal contact, and external gamma. Five possible exposure routes are available 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.

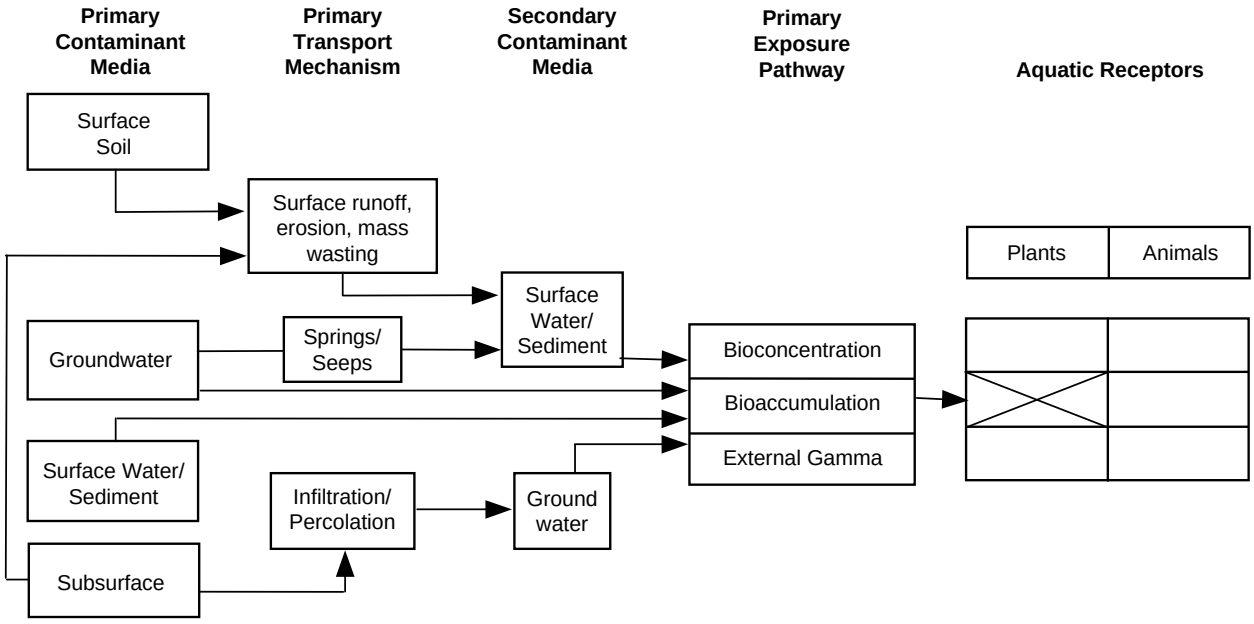


Note: Boxes with Xs indicate incomplete pathways. Open boxes indicate complete pathways.

Figure 4.1-1 Terrestrial receptor conceptual exposure and transport model

The generic aquatic receptor CSM 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 sediment 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 all contaminant exposure pathways are evaluated. Three possible exposure routes are

available 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

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) by exposure medium, 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.

Identification of COPECs first requires assembling exposure point concentrations (EPCs) and ESLs for all media, receptors, and COPCs. All the ESLs for the receptors in a chemical-medium combination are obtained from the Laboratory's ECORISK Database (LANL 2011, 206473, or latest version); the lowest ESL for that chemical in that medium becomes the minimum ESL used for the ecological screening.

The minimum ESLs are specific to the medium and include values for soil, sediment, and water, as appropriate. Each medium and COPC has a minimum ESL. The minimum ESL is the lowest applicable ESL value for a COPC in soil, sediment, and water and is intended to be protective for all ecological receptors in a given functional group for exposure to that single medium. The site EPC and the minimum ESL are used to calculate the COPC and medium-specific hazard quotient (HQ).

The HQ is a ratio between an EPC and a concentration in a medium corresponding to a potential indicator of effects, i.e., the ESL. The HI is a sum of HQs for COPECs with similar toxicological modes of

action. The following equations show how the HQ and hazard index (HI) are calculated, and are based on EPA (1997, 059370):

$$HQ_{ij} = \frac{exposure_{ij}}{effect_{ij}} \qquad \textbf{Equation 4.2-1}$$

$$HI_i = \sum_{j=1}^{n} HQ_{ij} \qquad \textbf{Equation 4.2-2}$$

Where HQ_{ij} is the hazard quotient for receptor *i* to COPC/COPEC *j* (unitless)

$exposure_{ij}$ is the EPC for COPEC *j* for receptor *i* (units are mg of COPEC per kg medium)

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

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

The ESLs and the toxicity and other parameter information required for their calculation are maintained in the Laboratory's ECORISK Database (LANL 2011, 206473, or latest version). The ECORISK Database is available to anyone performing or reviewing ecological screening assessments for the Laboratory, and updates to this database are issued as new information becomes available. The current version of the database is available for downloading at http://www.lanl.gov/environment/cleanup/ecorisk.shtml.

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 main components of the uncertainty analysis are described in section 4.3.

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 4.4.

4.3 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, 062784).

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. Examination of the uncertainty can result in adding or deleting COPECs. The uncertainty analysis may qualitatively discuss factors that may overestimate the potential risk for the site and factors that may underestimate the potential risk to ecological receptors at the site.

Uncertainties associated with ESLs fall into two main categories. The first group is associated with COPCs, including toxicity and bioavailability (or TFs between soil/sediment/water and food). The second group relates to receptors, including feeding rates, the amount of incidental soil/sediment/water ingestion and diets. These uncertainties are addressed by selecting inputs to the 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[3]. Uncertainties are also addressed by using the lowest receptor-specific ESL as the minimum ESL for each COPC to ensure the screening evaluation is protective and inclusive of all COPCs. ESLs only screen individual COPCs, and section 4.2.1 describes how multiple exposure media are evaluated.

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, 062786). 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 has 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, 062782) 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, 062788). The energy value of the food consumed by the animal also shows a relationship to food intake (Nagy 1987, 062782). For example, an animal consuming a low-energy food source must consume a greater quantity to support its basal metabolism. Thus, interrelationships exist between diet composition, body weight and food intake. Relationships also exist between body weight and HR because small animals tend to have smaller HRs (Cotgreave 1993, 062905). Thus, screening receptors were selected to be relatively small species within a feeding guild, which will tend to have smaller HRs 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.

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 overestimation of risks to receptors whose HRs 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 HR than the surrogate. Because T&E species must be assessed on an individual basis (EPA 1999, 070086), the AUF is used for the Mexican spotted owl. The flesh-eating kestrel represents both the feeding guild of carnivorous birds (using its normal HR) and serves as a surrogate for the Mexican spotted owl (which has a much larger HR).

[3] 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 1, it is conservative to assume that TF_{beef} applies to any vertebrate flesh.

4.3.1 Development of Area Use Factors

EPA guidance recommends evaluating ecological effects at the population rather than at the individual level (EPA 1999, 070086), except when evaluating T&E species. 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 HR of an individual receptor to reflect the fact that a receptor actually moves around its HR 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 HR. 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.

As discussed in section 3.3, PAUFs are developed based on investigations correlating the HR of a receptor with its dispersal distance (the distance an animal moves from its natal HR). 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, 073475). When HR is expressed as its linear dimension (the square root of HR), it has a good linear correlation with dispersal distance for the same species (Bowman et al. 2002, 073745). For mammals with similar HR sizes to the species used as screening receptors at the Laboratory, dispersal distance is equal to 3.5 times the square root of the HR. 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, 076074). The mathematical relationship between HR 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, 073460), so application of the mammalian relationship is protective of bird species because this relationship underestimates the dispersal distance and, therefore, the avian assessment population area.

The dispersal distance from the center of the HR 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 HR)^2. This mathematical relationship can be simplified to 40 times the HR as a representation of the assessment population area in hectares (Ryti et al. 2004, 076074). 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. HRs for the receptors are presented in Table 3.3-1.

PAUFs cannot be calculated for the plant and earthworm because these receptors do not have an HR that can be related to a population assessment area. The plant and earthworm are evaluated directly against their EPCs. Assessment populations of plants and earthworms are evaluated in a more qualitative manner. area use factors (AUFs) are used to account for the amount of time that a receptor is likely to

spend within the contaminated areas based on the size of the receptor's home range (HR). The AUFs for individual organisms were developed by dividing the size of the site by the HR for that receptor. Because T&E species must be assessed on an individual basis (EPA 1999, 070086), the AUF is used for the Mexican spotted owl based on an HR of 366 ha. The kestrel (top carnivore) is used as the surrogate receptor for the Mexican spotted owl.

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 HR of the Mexican spotted owl is used instead of the HR of the surrogate. The values in Table 4.3-1 for body weight, food ingestion rate, and HR for the Mexican spotted owl are from Gonzales et al. (2004, 085207). The value for water ingestion rate is developed from the allometric equation for drinking water consumption in birds based on body weight (EPA 1993, 059384, p. 3-8).

Table 4.3-1
Exposure Factors for the Mexican Spotted Owl

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

4.3.2 Exposure-Related Parameters

The CSMs for terrestrial and aquatic ecosystems 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. Soil ingestion rates, however, can represent one of the more significant sources 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, 062785). 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 HR greater than the size of the site.

Uncertainty is associated with the values used for the EPC. The uncertainty analysis should include some consideration of whether use of the maximum concentration of a COPC as the EPC 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 UCL as the EPC is likely to overestimate risk if the receptor has an HR greater than the area over which the UCL was determined. The analysis of uncertainty associated with the EPC should also include consideration of the findings of the data review (e.g., precision and bias of sample results for environmental media samples) and the impact of the review on the confidence and representativeness of the concentration estimate.

The uncertainty analysis discusses aspects of the conservative risk-screening process that over- or underestimate potential risk to receptors and thereby affect site decisions. In the case of the SLERA, one uncertainty is related to the exposure of receptors to COPEC concentrations not likely to result in adverse impacts. This overestimation of risk to receptors exposed either to naturally occurring levels or to exposure that cannot be distinguished from naturally occurring levels is described and put in the context of whether an increased risk to receptors exists. Therefore, the discussion and analysis are appropriate when determining whether COPECs are contributing to increased potential risk at a site.

The EPCs (either the UCL, the maximum detected concentration, or the maximum detection limit [DL]) are evaluated relative to the concentrations measured in samples of soil and tuff from uncontaminated areas of the Pajarito Plateau (LANL 1998, 059730). This uncertainty discussion and analysis is not related to whether an inorganic chemical was detected above background and is a COPC, but rather whether COPCs identified and retained as COPECs result in a potential increased risk to receptors at the concentration designed to represent exposure at the site. Furthermore, the presence of a concentration or concentrations above the background values (BVs) that resulted in the identification of a COPC does not mean the level of exposure to the COPC poses an increased risk.

The EPCs represented by the maximum detected concentrations or the maximum DLs are a deliberate overestimate of the exposure (and therefore the risk). If the EPC is the same as, or cannot be distinguished from, exposure to naturally occurring levels, then the risk to receptors (if present) is no different than would result from exposure to naturally occurring levels, that is, whatever risk may be potentially present is the same as that found in uncontaminated areas of the Pajarito Plateau. If the EPC is a UCL, then the concentration represents a reasonable estimate of the concentration the receptor is likely to come in contact with over time, (i.e., the mean concentration). If the reasonable estimate of the exposure concentration cannot be distinguished from exposure to naturally occurring levels, then any risk to receptors also cannot be distinguished from risk that may be from naturally occurring levels, that is, the potential risk from uncontaminated areas. For example, if the UCL is 8.3 mg/kg for copper and the measured background concentrations range from 0.25 mg/kg to 16 mg/kg for soil and 0.25 mg/kg to 6.2 mg/kg for tuff, the mean exposure to copper across the site is the same as if the receptor were exposed on average to a naturally occurring level of copper. In addition, because the UCL for copper background concentrations is 6.4 mg/kg and the UCL for site concentrations is 8.3 mg/kg, the difference in the potential risk associated with these concentrations is negligible (if any risk exists at all). Thus, risk from copper to ecological receptors cannot be distinguished from, or does not incrementally increase above, that associated with naturally occurring levels, making any further assessment of copper and risk unnecessary. If, on the other hand, the EPC for copper is 117 mg/kg, exposure across the site is above naturally occurring levels of copper and may pose a potential risk to ecological receptors. In this case, further assessment of copper is conducted to determine if a potential risk exists at this mean exposure level.

4.3.3 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 suggests an elevated risk or represents an exposure similar to background across the site.

4.3.3.1 COPECs without ESLs

Some COPECs do not have ESLs for any receptor in the ECORISK Database because literature searches for relevant toxicity data for these chemicals either have not been completed or no usable toxicity data exists. In an effort to address this uncertainty and provide a quantitative assessment of potential ecological risk, several online toxicity databases have been or can be searched to determine if any relevant toxicity information is available. The online databases typically searched include the EPA

Ecotox Database, EPA Office of Pesticide Programs Aquatic Life Benchmarks, U.S. Army Corps of Engineers/EPA Environmental Residue-Effects, California Cal/Ecotox Database, Pesticide Action Network Pesticide Database, U.S. Army Wildlife Toxicity Assessment Program, U.S. Department of Agriculture Integrated Pesticide Management Database, American Bird Conservancy Pesticide Toxicity Database, and Oak Ridge National Laboratory Risk Assessment Information System. Although some COPECs still do not have any relevant toxicity data in the online databases listed above, a search of the literature continues in an effort to determine if any relevant toxicity information exists.

In the absence of a chemical-specific ESL, COPEC concentrations can be compared with ESLs for a surrogate chemical. Comparison to surrogate ESLs provides an estimate of potential effects of a chemically related compound and a line of evidence to indicate the likelihood that ecological receptors are potentially impacted.

Some COPECs without ESLs do not have chemical-specific toxicity data or surrogate chemicals to be used in the screening assessments and cannot be assessed quantitatively for potential ecological risk. These COPECs are often infrequently detected across the site. In these cases, comparisons with residential human health SSLs are presented as part of a qualitative assessment. The comparison of COPEC concentrations to residential human health SSLs is a viable alternative for several reasons. Animal studies are used to infer effects on humans and constitute the basic premise of modern toxicology (EPA 1989, 008021). In addition, toxicity values derived for the calculation of human health SSLs are often based on potential effects that are more sensitive than the ones used to derive ESLs (e.g., cellular effects for humans versus survival or reproductive effects for terrestrial animals). The EPA also applies uncertainty factors or modifying factors to ensure that the toxicity values are protective (i.e., they are adjusted by uncertainty factors to values much lower than the study results). COPEC concentrations compared with these values are an order of magnitude or more below the SSLs, which corresponds to uncertainty factors of 10 or more. Therefore, it is assumed the differences in toxicity would not be more than an order of magnitude for any given chemical. The relative difference between values provides a weight of evidence that the potential toxicity of the COPC is likely to be low or very low to the receptor(s). The COPECs without ESLs may be common to many of the sites and are discussed separately for each site.

4.3.4 L-ESL Analysis

Sites may have adjusted HIs (using the comparison to background concentrations and/or the PAUF) greater than 1 for one or more receptors. To address these HIs and reduce the associated uncertainty, a LOAEL/LOEC analysis is conducted using L-ESLs calculated based on a LOAEL/LOEC rather than a NOAEL/NOEC. The L-ESLs are calculated based on toxicity information in the ECORISK Database and are presented in a table along with the basis for each LOAEL/LOEC used in the ESL calculations. This information has been incorporated into the ECORISK Database (LANL 2011, 206473) and can be referenced rather than presented in a separate table in the risk appendix. The analysis addresses some of the uncertainties and conservativeness of the ESLs used in the initial screening assessments. The HI analyses are conducted using the LOAEL/LOEC-based ESLs. The HQs and HIs calculated for this subset of receptors and COPECs are also adjusted using the PAUFs, if applicable, if the wildlife receptor HIs exceeded 1 using the LOAEL/LOEC-based ESLs.

4.3.5 Comparison with Previous Investigations

As another line of evidence and to reduce the uncertainty related to HIs greater than 1, a comparison of COPEC concentrations reported in the canyon investigations where field and/or laboratory studies and tests have been conducted to provide empirical data to site data may be presented. The premise for this

comparison is that if the field and laboratory studies/tests have not found any ecological effects on receptors, then the concentration(s) found at a site would also not impact ecological receptors even though the screening HI is greater than 1. The opposite is also true: the elevated HI indicates adverse impacts to ecological receptors has occurred or may occur.

Biota investigations have been conducted in canyon reaches in Los Alamos and Pueblo Canyons (LANL 2004, 087390); Mortandad Canyon (LANL 2006, 094161; LANL 2007, 098279); Pajarito Canyon (LANL 2009, 106939); and Sandia Canyon (LANL 2009, 107453). Field and laboratory studies included collection and analysis of soil, sediment, and water samples; cavity-nesting bird monitoring and analysis of eggs; small mammal trapping and analysis of whole organisms; earthworm bioaccumulation tests (measures of growth and survival, and analysis of whole organisms); laboratory testing of sensitive organisms; and seedling germination tests.

4.4 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 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 not adequate information to make a risk management decision. The result would be to identify data needs, based on the results of the screening, and to develop a plan to collect additional data.
2. There is adequate information to conclude 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.
3. Ecological risks are not negligible, but the information is insufficient to indicate adverse ecological effects are occurring. The risk management strategy is to reduce uncertainties in the screening assessment by conducting a baseline risk assessment.
4. There are sufficient lines of evidence to document potential or actual adverse ecological effects such that remediation is warranted.

5.0 REFERENCES

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Appendix A

Basis for the Derivation of Ecological Screening Levels

A-1.0 BASIS AND DERIVATION OF CHEMICAL ECOLOGICAL SCREENING LEVELS

This appendix provides the justification and basis for media-specific ecological screening levels (ESLs) for plants and invertebrates as well as the detailed equations that are the foundation for calculating wildlife screening levels.

A-1.1 Soil ESLs

The general wildlife exposure model is presented in section 3.3. This section of the appendix presents the parameters used to estimate exposure of ecological receptors to contaminants in soil and food. 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 A-1.1-1.

$$C_{food} = C_{soil} \cdot TF_{food} \qquad \textbf{Equation A-1.1-1}$$

Where C_{food} is the concentration of the chemical of potential concern (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 dry weight food per mg/kg dry weight soil)

Thus, the general wildlife exposure model can be rewritten in the following form, after setting the area use factor (AUF) to 1 and using the relationship between C_{soil} and C_{food} shown in Equation A-1.1-2.

$$E_{oral} = C_{soil} \cdot I_{soil} + C_{soil} \cdot TF_{food} \cdot I_{food} \qquad \textbf{Equation A-1.1-2}$$

Where E_{oral} is the estimated oral daily dose for a COPC (mg-COPC/kg-body wt/d)

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-soil dry wt/kg-body wt/d)

I_{food} is the normalized daily dietary ingestion rate (kg-food dry wt/kg-body wt/d)

TF_{food} is a transfer factor from soil to food (mg/kg dry weight food per mg/kg dry weight soil)

Because the intake of soil can be related to the intake of food, Equation A-1.1-2 can be further simplified to Equation A-1.1-3. 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.

$$E_{oral} = C_{soil} \cdot I_{food} \cdot [\, fs + TF_{food} \,] \qquad \textbf{Equation A-1.1-3}$$

Where E_{oral} is the estimated oral daily dose for a COPC (mg-COPC/kg-body wt/d)

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-food dry wt/kg-body wt/d)

TF_{food} is a transfer factor from soil to food (mg/kg dry weight food per mg/kg dry weight soil)

Solving Equation A-1.1-3 for the COPC and wildlife receptor-specific ESL yields Equation A-1.1-4:

$$ESL_{ij} = \frac{TRV_{ij}}{I_i \cdot [\, fs_i + TF_j \,]}$$ **Equation A-1.1-4**

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

TRV_{ij} is the toxicity reference value (no observed adverse effect level [NOAEL]) for wildlife receptor *i* and COPC *j* (mg-COPC/kg-body wt/d)

I_i is the normalized daily dietary ingestion rate for wildlife receptor *i* (kg-food dry wt/ kg-body wt/d)

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)

Equations for calculating wildlife ESLs for herbivore, omnivore, insectivore, and carnivore functional groups are shown in Equations A-1.1-5 through A-1.1-8, respectively.

$$ESL_{ij} = \frac{TRV_{ij}}{I_i \cdot [\, fs_i + fp_i \cdot TF_{plant,\,j} \,]}$$ **Equation A-1.1-5**

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

TRV_{ij} is the toxicity reference value (NOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/kg-body wt/d)

I_i is the normalized daily dietary ingestion rate for herbivore *i* (kg-food dry wt/kg-body wt/d)

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

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

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

$$ESL_{ij} = \frac{TRV_{ij}}{I_i \cdot [\, fs_i + fp_i \cdot TF_{plant,\,j} + fi_i \cdot TF_{invert,\,j} \,]}$$ **Equation A-1.1-6**

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

TRV_{ij} is the toxicity reference value (NOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/ kg-body wt/d)

I_i is the normalized daily dietary ingestion rate for omnivore *i* (kg-food dry wt/kg-body wt/d)

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 weight plant per mg/kg dry weight soil)

fi_i is the fraction of invertebrates 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 for COPC *j* (mg/kg dry weight flesh per mg/kg dry weight soil)

$$ESL_{ij} = \frac{TRV_{ij}}{I_i \cdot [\, fs_i + fi_i \cdot TF_{invert,\,j}\,]}$$ **Equation A-1.1-7**

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

TRV_{ij} is the toxicity reference value (NOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/ kg-body wt/d)

I_i is the normalized daily dietary ingestion rate for insectivore *i* (kg-food dry wt/kg-body wt/d)

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

fi_i is the fraction of invertebrates in diet for insectivore *i*

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

$$ESL_{ij} = \frac{TRV_{ij}}{I_i \cdot [\, fs_i + ff_i \cdot TF_{flesh,\,j}\,]}$$ **Equation A-1.1-8**

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

TRV_{ij} is the toxicity reference value (NOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/ kg-body wt/d)

I_i is the normalized daily dietary ingestion rate for carnivore *i* (kg-food dry wt/kg-body wt/d)

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

ff_i is the fraction of flesh in diet for carnivore *i*

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

The wildlife ESL model (Equation A-1.1-4 and the functional group-specific Equations A-1.1-5 through A-1.1-8) shows the ESL as proportional to the effect level. Thus, larger values of the toxicity reference value (TRV) lead to larger ESL values, which indicate 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 TRV value as a receptor with lower exposure. The wildlife lowest effect ESLs (L-ESLs) are calculated with Equations A-1.1-5 through A-1.1-8 using the lowest observed adverse effect level (LOAEL) for the TRV term. Table A-1.1-1 summarizes the input variables for the wildlife exposure models and indicates the general sources used for these variables.

Table A-1.1-1
Summary of Variables Used in the Nonradionuclide Wildlife ESL Models

Variable	Source
TRV	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 2011, 206473, or latest version). The wildlife L-ESLs can be calculated using the LOAEL for the TRV term.
fs	Receptor-specific values are provided in Table 3.3-1 of the main text.
I	Body weight normalized food intake for wildlife receptors (see values provided in Table 3.3-1). Body weight is an implicit component of this variable. For this reason, Table 3.3-1 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	The fraction of plants in diet is provided in Table 3.3-1.
fi	The fraction of invertebrates in diet is provided in Table 3.3-1.
ff	The fraction of flesh in diet is provided in Table 3.3-1.
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 2011, 206473, 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 2011, 206473, or latest version).
TF_{flesh}	The transfer from soil to flesh is a COPC-specific value derived from three other factors (LANL 2002, 072641). 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, $TF_{flesh} = TF_{beef} \cdot (I_{food} \bullet \text{maximum of } [TF_{plant} \bullet (1\text{-}MC_{plant}], TF_{invert} \bullet [1\text{-}MC_{invert}]) + I_{soil})/(1 - MC_{flesh})$ Values for specific COPCs are provided in the ECORISK Database (LANL 2011, 206473, or latest version).

A-1.2 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, air exposure is potentially a significant exposure pathway for burrowing mammals at some solid waste management units and areas of concern at Los Alamos National Laboratory (the Laboratory). Gaseous or otherwise airborne contaminants can build up in burrows because the potential for dilution with the atmosphere is much more limited compared to surface conditions. Exposure parameters for the pocket gopher are provided in Table 3.3-1 of the main text.

The gopheris inhalation rate (IR) is based on body weight (BW) according to the allometric equation from Stahl (1967, 063119) shown in Equation A-1.2-1:

$$IR = 0.5458 \cdot BW^{0.80}$$ **Equation A-1.2-1**

It is assumed the gopher stays in its burrow 100% of the time; therefore, the exposure through air is described by Equation A-1.2-2:

$$E_{air} = \frac{C_{air} \cdot I_{air}}{BW}$$ **Equation A-1.2-2**

Where E_{air} is the estimated inhalation daily dose for a COPC (mg-COPC/kg-body wt/d)

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

I_{air} is the daily air inhalation rate (m^3/d)

BW is the body weight for the pocket gopher (kg)

Therefore, the ESL can be expressed as shown in Equation A-1.2-3:

$$ESL_j = \frac{TRV_j \cdot BW}{I_{air}}$$ **Equation A-1.2-3**

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

TRV_j is the NOAEL for burrow animal inhalation and COPC *j* (mg-COPC/kg-body wt/d)

BW is the body weight for burrow animal (kg)

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

The wildlife L-ESLs can be calculated with Equation A-1.2-3 and using the LOAEL for the TRV term.

A-1.3 Sediment ESLs Protective of the Aquatic Community

In selecting sediment ESLs that are protective of the aquatic community, it is first determined if a benchmark is available in the National Oceanic and Atmospheric Administration (NOAA) Screening Quick Reference Tables (SQuiRT) (Buchman 2008, 206414). The SQuiRT represents a nationally recognized compendium for ecological effects values in soil, sediment, and water. Within the SQuiRT, benchmarks are evaluated in the order presented in Figure A-1.3-1, based on the rigor and comprehensiveness of the data source. Preference was given to benchmarks based on publication date (more recent assumed to reflect broader extent of scientific knowledge), chronic direct exposure, and nonlethal endpoint studies designed to be protective of sensitive species.

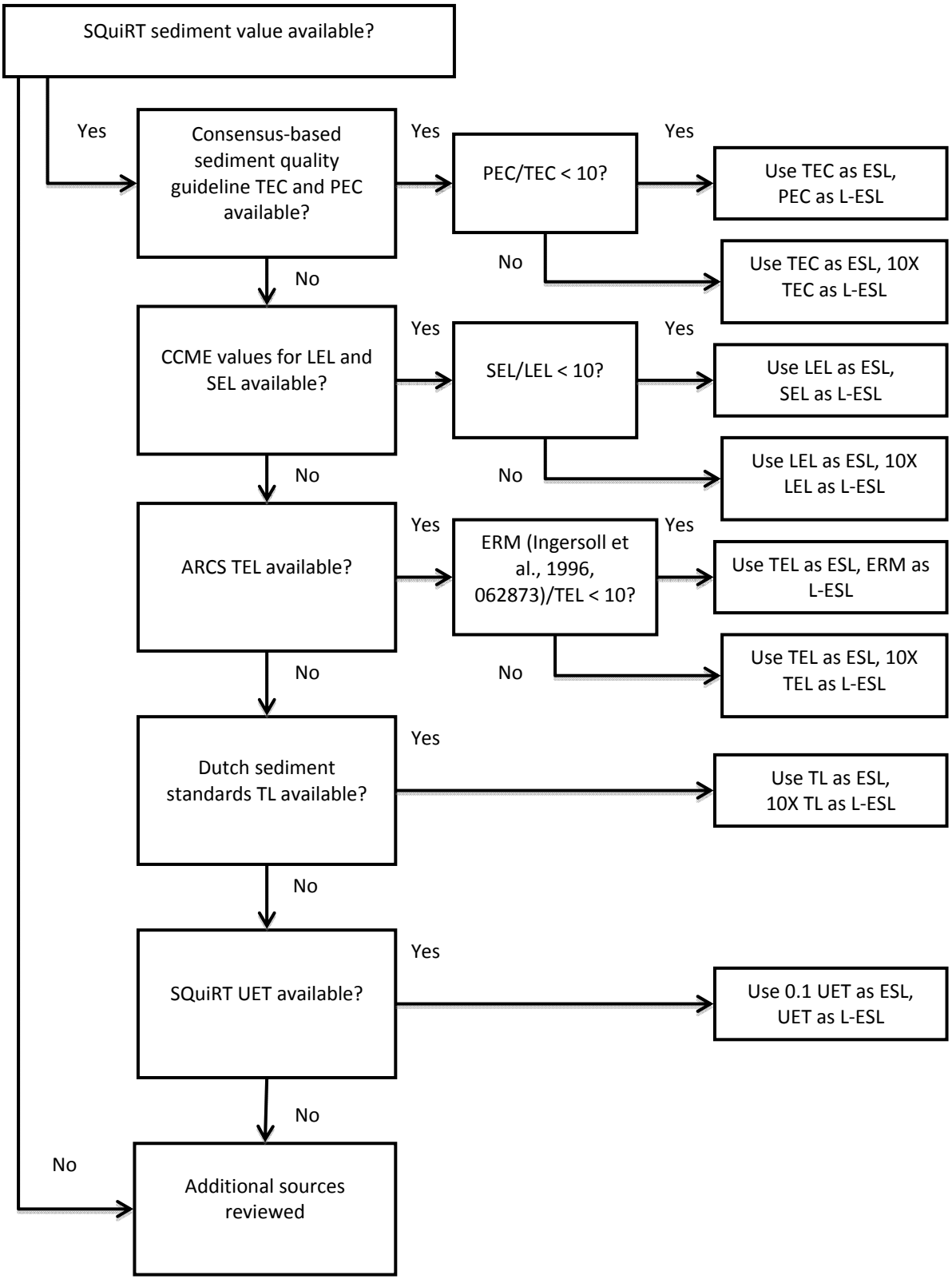

Figure A-1.3-1 Sediment ESL selection process for the aquatic sediment community

Sediment benchmarks from MacDonald et al. (2000, 205266) were selected as the first potential source of sediment ESLs protective of the aquatic community. For some contaminants, MacDonald et al. (2000, 205266) published two consensus-based benchmarks for each contaminant, including a threshold effect concentration ([TEC] the concentration below which adverse effects are not expected to occur) and a probable effect concentration ([PEC] the concentration above which adverse effects are expected to occur more often than not). The predictive ability of these benchmarks was numerically evaluated for accuracy using field data.

If a TEC and/or PEC is not available from MacDonald et al. (2000, 205266), the next potential source for freshwater sediment benchmarks was Persuad et al. (1993, 205250), which form the basis for sediment screening values used by the Canadian Council of Ministers of the Environment ([CCME] see http://st-ts.ccme.ca/). Some of the CCME values have been periodically updated since the since they were first published in the early 1990s. The sediment ESL protective of the aquatic community is based on the CCME concentrations below which are tolerated by the majority of benthic organisms. The L-ESL is based on concentrations that are expected to be detrimental to the majority of benthic species.

If a CCME value was not available in SQuiRT, the Assessment and Remediation of Contaminated Sediments (ARCS) program in the Great Lakes (Ingersoll et al. 1996, 062873) were the next potential source of sediment benchmarks. The ARCS program has sponsored numerous investigations using the amphipod *Hyalella azteca* and the midge *Chironomous riparius* in sediment bioassays. These results, along with those from other freshwater areas, were used to generate a threshold effect level ([TEL] the consensus based concentration of a contaminant below which adverse biological effects are expected to occur rarely) and effect range median ([ERM] concentration of a chemical in sediment above which effects are frequently or always observed or predicted among most species). The ERM values from ARCS represent studies on freshwater species and should not be confused with the marine ERM values. Marine ERM values are not used as the basis for ESLs. The next potential source of sediment benchmarks is the Dutch[1] sediment threshold level (TL) that may be used as a no observed effect concentration (NOEC) for the ESL. The TL represents concentrations that delineate the threshold below which effects are not expected. The last source consulted in the SQuiRT is the upper effects threshold (UET), a sediment toxicity value put forth by NOAA that corresponds to a concentration above which adverse impacts on the benthic community are always expected. A UET is not suitable for a no-effect screening level but can be used for a low-effect level; consequently, the ESL is derived by taking one-tenth of the UET. The L-ESL would be equal to the UET.

If sediment toxicity values are not available in the SQuiRT, Michelsen (2003, 215128) was consulted. Michelsen (2003, 215128) compiled freshwater sediment toxicity results intended for use in the State of Washington and these benchmarks are likely representative of potential for adverse effects in any freshwater stream, including those found at the Laboratory. No other neighboring state has compiled freshwater sediment toxicity values. Sediment quality values were generated using four bioassay endpoints: *H. azteca* 10-d mortality, *Chironomus* 10-d mortality, *Chironomus* 10-d growth, and Microtox 15-min luminescence bioassays. Michelsen (2003, 215128) compiles two relevant sediment benchmarks: the sediment quality standard (SQS) and the cleanup screening level (CSL). The SQS corresponds to the concentration that will result in no adverse effects, including no acute or chronic adverse effects on biological resources, and the CSL corresponds to concentration below which only minor adverse effects would occur and above which more significant adverse effects are expected.

[1] As reported in the SQuiRT, Dutch Standards are "Verbruggen EMJ, Psthumus R, van Wezel AP. 2001. Ecotoxicological serious risk concentrations for soil, sediment, and (ground)water. National Institute of Public Health and the Environment and subsequent updates as published elsewhere."

If benchmarks are not available from any of the preferred sources, values used in the Netherlands (Crommentuijn et al. 2000, 205264; Crommentuijn et al. 2000, 205265) can be considered. These values are grouped in two categories: the negligible concentration (NC) and the maximum permissible concentration (MPC). The NC is associated with negligible risk and the MPC is the concentration in the environment above which the risk of adverse effects was considered unacceptable to sedimentary ecosystems. If Crommentijn (2000, 205264; 2000, 205265) values are unavailable, the equilibrium partitioning (EqP) values are used. The relationship between the octanol-water partitioning coefficient (K_{ow}) and the sediment organic carbon partitioning coefficient (K_{oc}) is described by Equation A-1.3-1 (Di Toro 1985, 062876):

$$log_{10} K_{oc} = 0.00028 + 0.983\ log_{10} K_{ow}$$ **Equation A-1.3-1**

where

K_{oc} is the sediment organic carbon partitioning coefficient

K_{ow} is the octanol/water partition coefficient

If the EqP is used as the ESL, then the ESL is multiplied by a factor of 10 to derive the L-ESL. Because the sediment ESLs are broadly representative of the adverse effects of contaminants on the aquatic community, they are applied to both aquatic plants and to aquatic invertebrates. The sediment ESLs described here are broadly protective of the aquatic environment. Table A-1.3-1 lists the definitions of the sediment effect concentrations.

In addition to selecting sediment ESLs that are protective of the aquatic community, the SLERA describes the methods used to calculate the wildlife ESL (sections 3.4.4.1 and 3.4.4.2 of the main text). The sediment ESL for a given chemical is the lowest of the values available for the aquatic community or wildlife.

A-1.4 Water ESLs Protective of the Aquatic Community

This section describes the selection process for water ESLs or benchmarks protective of the aquatic community. Values reported as chronic are used for the ESLs and those reported as acute are used for L-ESLs. In some cases, study conditions did not match or produce data directly comparable with chronic or acute benchmarks. In these instances, and when the difference between the chronic and acute was more than tenfold, uncertainty factors were applied to the lowest acceptable study data in the order of preferred sources (Figure A-1.4-1) to obtain water benchmarks. Uncertainty factors were used for the conversion of acute values to chronic values and, conversely, when only a chronic value was available, uncertainty factors were applied to derive the acute value. Table A-1.4-1 provides definitions for terms used in development of water ESLs protective of the aquatic community.

For conversion of chronic values to acute an uncertainty factor of ten was applied. This value is consistent with the geometric mean (7.6) of the acute-chronic ratios (ACRs) used by the U.S. Environmental Protection Agency (EPA) in the development of ambient water-quality criteria (AWQC) for primary pollutant metals (EPA 2009, 109328). It is within the range of 1–10 recommended by EPA in the Great Lakes Water Quality Initiative (60 Federal Register [FR] 15366, “Final Water Quality Guidance for the Great Lakes System, Final Rule”), and is supported by EPA Region 10 guidance (EPA 1997, 215127).

Table A-1.3-1
Definitions of Terms Associated with Sediment ESLs Protective of the Aquatic Community

Term	Definition	Description
CCME	Canadian Council and Ministry of the Environment	Canadian environmental standards. Based on Persuad et al. (1993, 205250) and periodically updated (e.g., www.ccme.ca/publications/ceqg_rcqe.html)
CSL	Cleanup screening level	Cleanup screening level, concentration below which only minor adverse effects would occur and above which more significant adverse effects are expected
EqP	Equilibrium partitioning	Method to calculate sediment ESLs based on chemicals toxicity in water and calculated partitioning between sediment and water
ERM	Effect range median	Concentration of a chemical in sediment above which effects are frequently or always observed or predicted among most species
IL	Intervention level	Dutch sediment standard for environmental risk limit for bedded sediment
K_{oc}	Partitioning coefficient	Sediment organic carbon partitioning coefficient
K_{ow}	Partitioning coefficient	Octanol/water partition coefficient
LEL	Lowest effect level	Concentration below which is tolerated by majority of benthic organisms
NC	Negligible concentration	Concentration below which potential for adverse effects are negligible
PEC	Probable effects concentration	Consensus based concentration in sediment to which a plant or animal is directly exposed that is likely to cause an adverse effect
SEL	Severe effect level	Concentrations above SEL expected to be detrimental to the majority of benthic species
SQS	Sediment quality standard	Concentration that will result in no adverse effects, including no acute or chronic adverse effects on biological resources
TEC	Threshold effect concentration	Consensus based concentration of a contaminant above which some effect will be produced and below which it will not
TEL	Threshold effect level	Consensus based concentration of a contaminant below which adverse biological effects are expected to occur rarely
TL	Target level	Dutch sediment standard for acceptable level of chemical in bedded sediment environment
UET	Upper effects threshold	SQuiRT value above which adverse impacts on the benthic community are always expected

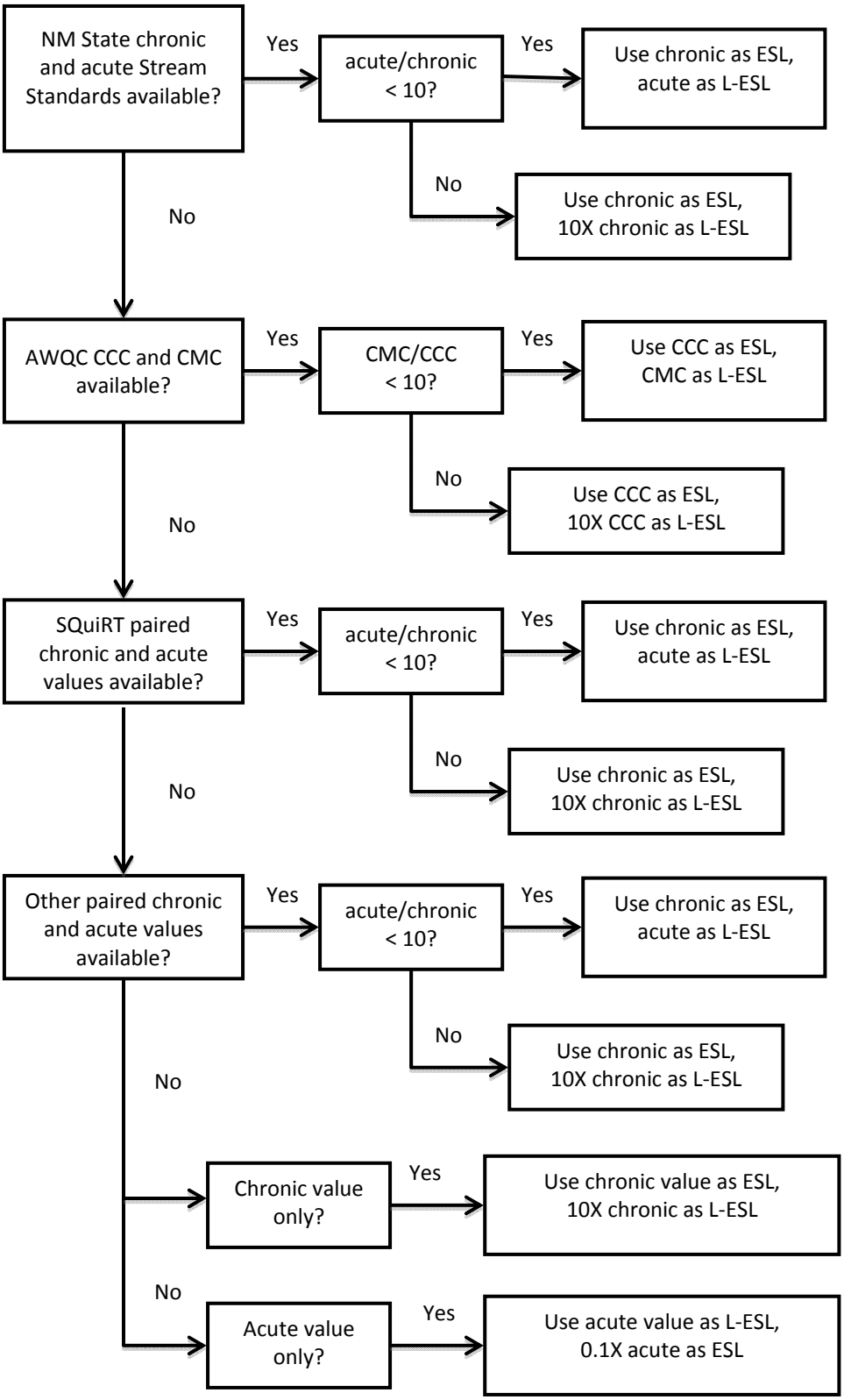

Figure A-1.4-1 Water ESL selection process for the aquatic community

Table A-1.4-1
Definitions of Terms Associated with Water ESLs Protective of the Aquatic Community

Term	Definition	Description
ACR	acute-chronic ratio	Ratio of the acute toxicity of a toxicant to its chronic toxicity, used as a factor for estimating chronic toxicity on the basis of acute toxicity data or for estimating acute toxicity on the basis of chronic toxicity data
AWQC	ambient water quality criteria	U.S. national recommended water-quality criteria broadly protective of aquatic species
CCC	criterion continuous concentration	The concentration in water that is expected to be protective of 95% of aquatic species over chronic exposure
CMC	criterion maximum concentration	The concentration in water that represents a low-level effect on the fifth percentile genus, applied as a limit on the short-term average concentration in the environment. Both the acute and chronic criteria are values that are not to be exceeded more than once in 3 yr. In other words, the criteria specify a magnitude, duration, and frequency to be met in order to provide protection of aquatic life.
LOEC	lowest observed effect concentration	Concentration below which is tolerated by majority of aquatic organisms and above which effects may be expected
NOEC	no observed effect concentration	Concentration below which is expected to result in no effect on aquatic organisms
NOAA	National Oceanic and Atmospheric Administration	A U.S. federal agency focused on the condition of the oceans and the atmosphere

Values are selected from four tiers of data sources with Tier 1 being the most preferred data source. If data cannot be identified for a particular chemical, the NOEC- and lowest observed effect concentration–(LOEC-) based ESLs are designated as data gaps. The selection process followed is shown in Figure B 1.4-1 and the data sources are as follows:

1. Section 20.6.4.900 of the New Mexico Administrative Code (20.6.4.900 NMAC)
2. AWQC set forth by EPA (2009, 109328)
3. NOAA Screening Quick Reference Tables (Buchman 2008, 206414)
4. Other sources

Water ESLs are selected utilizing water-quality criteria or benchmarks in the order presented above. For example, if 20.6.4.900 NMAC criteria are available for a given constituent, then this is selected as the most relevant screening value. If no 20.6.4.900 NMAC criterion is available, the AWQC are evaluated as the next tier. Justification for selecting the above order is provided in greater detail in 20.6.4.900 NMAC and in various EPA documents (60 FR 15366, “Final Water Quality Guidance for the Great Lakes System, Final Rule”;[EPA 2009, 109328]).

AWQCs are developed by EPA’s Office of Water under the Clean Water Act, Section 304 (EPA 2009, 109328). New Mexico has developed similar criteria for “high quality coldwater fisheries” as listed in “Standards for Interstate and Intrastate Streams,” 20.6.4.900 NMAC. The development of AWQCs is outlined in EPA guidance (60 FR 15366, “Final Water Quality Guidance for the Great Lakes System, Final Rule”). Metals are often water hardness-dependent and should be adjusted for site-specific conditions (see EPA guidance [EPA 2009, 109328], and 20.6.4.900 NMAC for explanations/delineation of methods, as methods require analyte-specific information).

If New Mexico state water quality criteria (WQCs) or National AWQCs are unavailable, values from the SQuiRT (Buchman 2008, 206414) should be reviewed for applicability. In some cases more than one chronic value is presented for a chemical in Buchman (2008, 206414), In such instances, the priority was to go with ECOTOX thresholds or Tier II SCVs (http://www.esd.ornl.gov/programs/ecorisk/tools.html) followed by values from Canada or New Zealand with the EPA Region V ecological screening values (http://www.epa.gov/reg5rcra/ca/edql.htm) being the last option. For more information on Buchman (2008, 206414), see section A-1.3, sediment ESLs. If toxicity information is not available in the SQuiRT, other sources were consulted for water benchmarks.

In addition to selecting water ESLs that are protective of the aquatic community, the screening-level ecological risk assessment (SLERA) describes the methods used to calculate the wildlife ESL (sections 3.4.5.1 and 3.4.5.2 of the main text). The water ESL for a given chemical is the lowest of the values available for the aquatic community or wildlife.

A-2.0 ESLs FOR RADIONUCLIDES

Considerations for the derivation of ESLs for radionuclides are presented in section 3.5 of the SLERA. The methods followed for radionuclide ESL development are consistent with DOE guidance (DOE 2002, 085637) and with the conceptual basis presented by NMED for evaluating ecological effects of radionuclides (NMED 2000, 087104). The equations and assumptions underlying radiological ESL development for soil, sediment, and water are presented in the sections that follow.

A-2.1 Soil ESLs

A-2.1.1 Radionuclide Concentrations in Biota

Plants and Invertebrates

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

$$C_{plant,j} = C_{soil,j} \cdot TF_{plant,j} \qquad \textbf{Equation A-2.1-1}$$

Where $C_{plant,j}$ is the internal concentration of radionuclide *j* in plants (pCi/g fresh weight)

$C_{soil,j}$ is the soil concentration of radionuclide *j* (pCi/g dry soil)

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

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

$$C_{invert,j} = C_{soil,j} \cdot TF_{invert,j} \qquad \textbf{Equation A-2.1-2}$$

Where $C_{invert,j}$ is the internal concentration of radionuclide *j* in invertebrates (pCi/g fresh weight)

$C_{soil,j}$ is the soil concentration of radionuclide *j* (pCi/g dry soil)

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

Values and references for transfer factors are presented in the ECORISK Database (LANL 2011, 206473, or latest version). When values are not available in the literature for soil-to-invertebrate transfer, a default value of 1 is used.

Wildlife

The internal dose to wildlife is calculated by multiplying the effective energy of a radionuclide by the body burden of that radionuclide in an organism. Body burden is a measure of the accumulation of a radionuclide in an organism through ingestion. The body burden calculation is presented in Equation A-2.1-3.

$$C_{wildlife,j} = C_{soil,j} \cdot [I_{soil} + TF_{food,j} \cdot I_{food}] \cdot TF_{blood,j} \cdot R_{t,j} \qquad \textbf{Equation A-2.1-3}$$

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/d)

I_{food} is the normalized daily dietary ingestion rate (g of food [fresh wt]/g of body weight/d)

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

$TF_{blood,j}$ 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 3-3-1 of the main text. Values and supporting references for all transfer factors used are provided in the ECORISK Database (LANL 2011, 206473, or 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, 062801)

$$R_t = (1 - e^{-\lambda Tc}) / \lambda \qquad \textbf{Equation A-2.1-4}$$

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 2011, 206473, or latest version).

A-2.1.2 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/d 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/d per pCi/g. This conversion factor is derived in Equation A-2.1-5.

Equation A-2.1-5

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

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, 062802). 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 A-2.1-6**

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

CF_i is the conversion factor between energy per disintegration and rad/d [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)

A-2.1.3 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, 062801; EPA 1993, 062798). Several simplifying assumptions are made in estimating this dose. First, as indicated by the conceptual site model diagram (Figure 3.4-1 of the main text), 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, 062804). 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-degree 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 (EPA 1993, 062798) 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 (EPA 1993, 062798). 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:

$$DCF_{ext,j} = DC_{water,skin,j} \cdot CF_{e,w} \qquad \textbf{Equation A-2.1-7a}$$

Terrestrial receptors on or above the soil surface:

$$DCF_{ext,j} = DC_{soil,skin,j} \cdot CF_{e,s} \qquad \textbf{Equation A-2.1-7b}$$

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

$DC_{water,skin,j}$ is the dose coefficient for skin exposed to water contaminated to an infinite depth with radionuclide *j* (from EPA 1993, 062798)

$CF_{e,w}$ is the conversion factor from Sv/s per Bq/m^3 [2] to rad/d per pCi/g for an organism immersed in water (value is 3.2×10^{11}; see Equation A-2.1-8)

$DC_{soil,skin,j}$ is the dose coefficient for skin exposed to soil contaminated to an infinite depth with radionuclide *j* (from EPA 1993, 062798)

$CF_{e,s}$ is the conversion factor from Sv/s per Bq/m^3 to rad/d per pCi/g for an organism on the soil surface (value is 5.11×10^{11}; see Equation A-2.1-9)

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

$$CF_{e,w} = 10^3 \frac{kg}{m^3} \cdot 10^3 \frac{g}{kg} \cdot 100 \frac{rad}{Sv} \cdot \frac{Bq}{27.03 pCi} \cdot 86400 \frac{s}{d} \qquad \textbf{Equation A-2.1-8}$$

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

$$CF_{e,s} = 1.6 \times 10^3 \frac{kg}{m^3} \cdot 10^3 \frac{g}{kg} \cdot 100 \frac{rad}{Sv} \cdot \frac{Bq}{27.03 pCi} \cdot 86400 \frac{s}{d} \qquad \textbf{Equation A-2.1-9}$$

A-2.1.4 Calculations of Soil ESLs

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/d to any organism. For terrestrial plants this calculation is written as

$$ESL = \frac{Dose\ Limit}{TF_{plant,j} \cdot DCF_{int,j} + DCF_{ext,j}} \qquad \textbf{Equation A-2.1-10}$$

Where Dose limit is 0.1 rad/d

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

$DCF_{int,j}$ is the internal dose conversion factor for radionuclide *j* (rad/d per pCi/g-fresh tissue)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g dry soil, assuming 360-degree exposure)

[2] 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.

For terrestrial invertebrate receptors, the ESL equation is written as

$$ESL = \frac{Dose\ Limit}{TF_{invert,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation A-2.1-11**

Where Dose limit is 0.1 rad/d

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

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

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g dry soil, assuming 360-degree exposure)

For terrestrial herbivores, the ESL equation is written as

$$ESL = \frac{Dose\ Limit}{[I_{soil,i} + TF_{plant,j} \cdot I_{plant,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation A-2.1-12**

Where Dose limit is 0.1 rad/d

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

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

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

$TF_{blood,j}$ 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/d per pCi/g fresh tissue)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g dry soil assuming 180- or 360-degree exposure)

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

$$ESL = \frac{Dose\ Limit}{[I_{soil,i} + TF_{invert,j} \cdot I_{invert,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$ **Equation A-2.1-13**

Where Dose limit is 0.1 rad/d

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

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

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

$TF_{blood,j}$ 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/d per pCi/g fresh tissue)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g dry soil assuming 180- or 360-degree exposure)

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

Equation A-2.1-14

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

Where Dose limit is 0.1 rad/d

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

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

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

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

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

$TF_{blood,j}$ 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/d per pCi/g fresh tissue)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g dry soil assuming 180- or 360-degree exposure)

For terrestrial carnivores, the ESL is calculated as

$$ESL = \frac{Dose\ Limit}{[I_{soil,i} + TF_{flesh,j} \cdot I_{flesh,i}] \cdot TF_{blood,j} \cdot R_{t,j} \cdot DCF_{int,j} + DCF_{ext,j}}$$

Equation A-2.1-15

Where Dose limit is 0.1 rad/d

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

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

$I_{flesh,i}$ is the normalized daily flesh ingestion rate for organism *i* (g of flesh-fresh wt/g of body wt/d)

$TF_{blood,j}$ 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/d per pCi/g fresh tissue)

$DCF_{ext,j}$ is the external dose conversion factor for radionuclide *j* (rad/d per pCi/g dry soil assuming 180- or 360-degree exposure)

The soil radiological L-ESLs are calculated with Equations A-2.1-10 through A-2.1-15 and using 1 rad/d as the dose limit. Table A-2.1-1 summarizes the variables used to calculate soil ESLs for radionuclides.

Table A-2.1-1
Summary of Variables Used in Soil ESL Calculations for Radionuclides

Variable	Source
I_{soil}	BW normalized soil ingestion rate for wildlife receptors (food intake × fraction of soil in diet from Table 3.3-1 of the main text).
I_{plant}	BW normalized plant ingestion rate for wildlife receptors (food intake × fraction of plants in diet from Table 3.3-1).
I_{invert}	BW normalized invertebrate ingestion rate for wildlife receptors (food intake × fraction of invertebrates in diet from Table 3.3-1).
I_{flesh}	BW normalized flesh ingestion rate for wildlife receptors (food intake × fraction of flesh in diet from Table 3.3-1).
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 A-2.1-4 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/d and the food ingestion rate. Values for specific COPCs are provided in the ECORISK Database (LANL 2011, 206473, 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 2011, 206473, 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 2011, 206473, 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 rate among these four potential prey species was used to represent the composite prey species in the equation. Thus, $TF_{flesh} = TF_{beef} \times (I_{food} \bullet \text{maximum of } (TF_{plant} \times (1\text{-}MC_{plant}), TF_{invert} \times (1\text{-}MC_{invert})) + I_{soil})/(1\text{-}MC_{flesh})$ Values for specific COPCs are provided in the ECORISK Database (LANL 2011, 206473, or latest version).
f_a	The fraction of energy deposition in an organism from 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/d per pCi/g BW and accounts for the increased biological effectiveness of alpha particle deposition over beta or gamma deposition (see Equation A-2.1-6).
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 dry soil and is based on assuming 180- or 360-degree exposure.

A-2.2 Sediment ESLs

A-2.2.1 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 3.4.4 of the main text (Baker and Soldat 1992, 062801). 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, 085637). 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:

$$C_{organism,j} = C_{sediment,j} \cdot BCF_{invert,j} \cdot I_{food,i} \cdot TF_{blood,j} \cdot R_{t,j} \qquad \textbf{Equation A-2.2.1}$$

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 of invertebrate-fresh wt/g dry sediment)

$I_{food,i}$ is the normalized daily dietary ingestion rate of organism *i* (g of food [dry wt]/g of body weight /d)

$TF_{blood,j}$ 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 3.5-7 of the main text)

Values and references for the transfer factors and bioconcentration factors are provided in the ECORISK Database (LANL 2011, 206473, or latest version).

A-2.2-2 Dose Conversion Factors

For aquatic receptors, internal dose conversion factors are identical to those used for terrestrial receptors (see Equation 3.5-9 of the main text). 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 A-2.1-7a and A-2.1-7b). 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 sediment, it is assumed the external dose to terrestrial receptors is insignificant and all dose received is internal.

A-2.3 Water ESLs

A-2.3.1 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, 062801):

$$C_{organism,j} = C_{water,j} \cdot BCF_{organism,j} \qquad \textbf{Equation A-2.3-1}$$

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 (pCi/g fresh weight per pCi/mL water)

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

$$C_{organism,\,j} = C_{water,\,j} \cdot I_{water} \cdot TF_{blood,\,j} \cdot R_{t,\,j}$$ **Equation A-2.3-2**

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 /d)

$TF_{blood,j}$ is the water 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 3.5-7 of the main text)

Values and references for the transfer factors and bioconcentration factors are provided in the ECORISK Database (LANL 2011, 206473, or latest version).

A-2.3.2 Dose Conversion Factors

For aquatic receptors, the internal DCFs are identical to those used for terrestrial receptors. For organisms immersed in water (algae, daphnid, snail, fish), the external dose coefficients of EPA guidance (EPA 1993, 062798) are used to estimate external dose. Coefficients used are for skin immersed in water contaminated to an infinite depth. A conversion factor of 3.2×10^{11} is used to convert the dose coefficients from Sv/s per Bq/m^3 to rad/d per pCi/g.

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.

A-3.0 REFERENCES

The following list includes all documents cited in this report. Parenthetical information following each reference provides the author(s), publication date, and ER ID. This information is also included in text citations. ER IDs are assigned by the Environmental Programs Directorate's Records Processing Facility (RPF) and are used to locate the document at the RPF and, where applicable, in the master reference set.

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Appendix B

Ecological Scoping Checklist

B-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 **Indicate terminal point of surface water transport; slope; and surface water run-on sources.**	

B-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.

B-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?
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 soil?

- **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 soil (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 soil?

- **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 soil?

- **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 soil?

- **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 sediment (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 sediment?

- **If sediment are present in an area that is only periodically inundated with water, terrestrial receptors may incidentally ingest sediment.**
- **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 sediment 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 sediment 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):

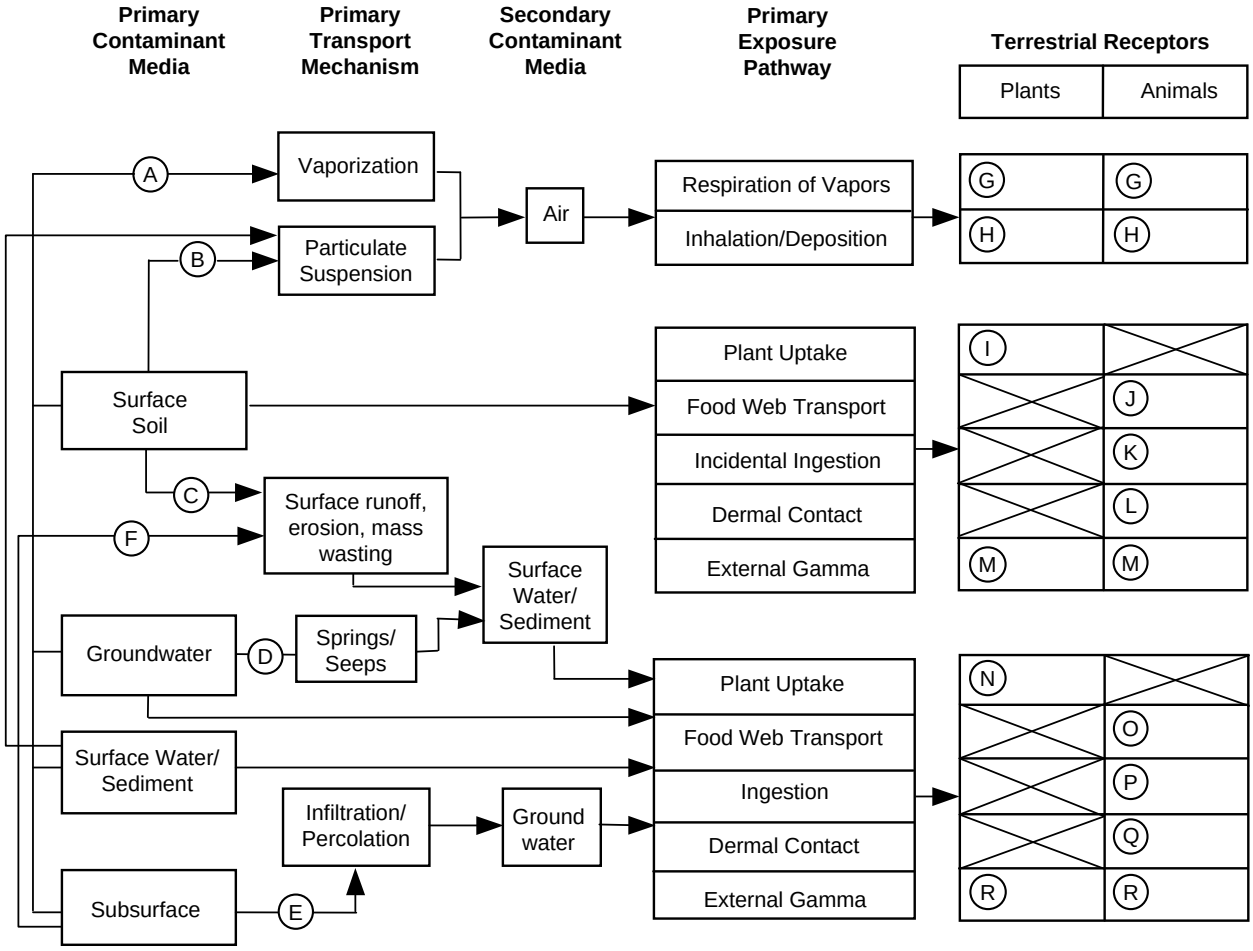
Aquatic Plants:

Aquatic Animals:

Provide explanation:

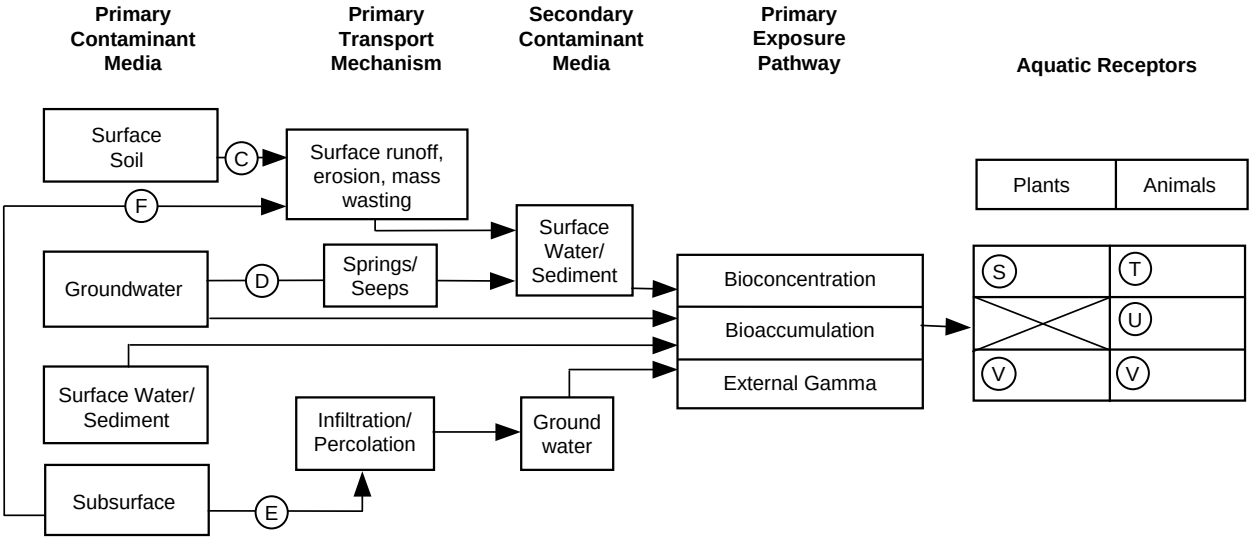
Ecological Scoping Checklist
Terrestrial Receptors
Ecological Pathways Conceptual Exposure Model

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):

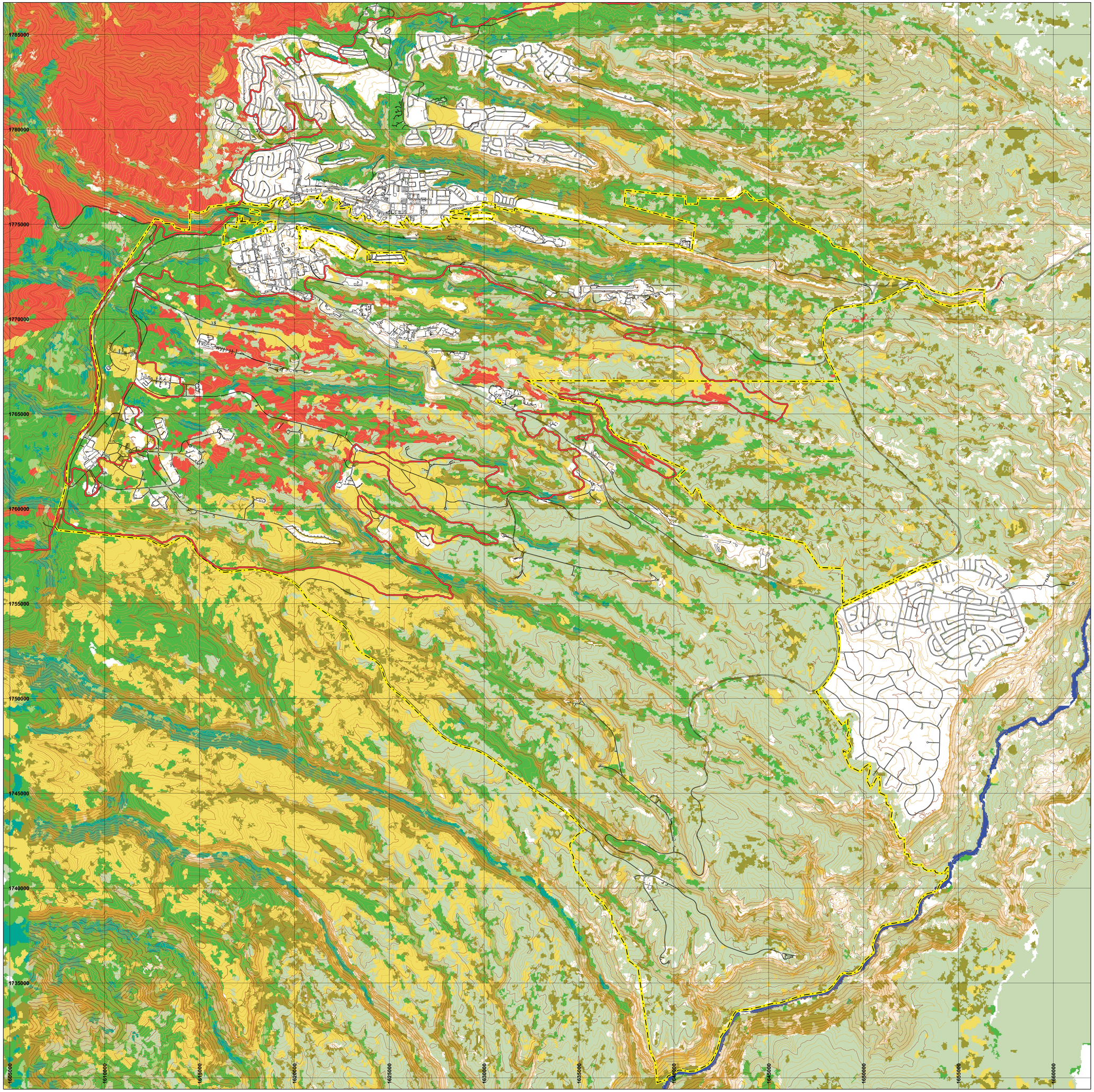
Name (printed): _____

Name (signature): _____

Organization: _____

Phone number: _____

Plate 1. Vegetation Land Cover Map of the LANL Area



Legend

- Cerro Grande extent
- LANL boundary
- Paved road
- 20 ft Contour
- 100 ft Contour

Description

- Aspen-Riparian-Wetland
- Cerro Grande Fire high burn severity
- Grass species
- Mixed conifer-Spruce-Fir
- Open Water
- Pinon-Juniper
- Ponderosa Pine
- Shrub species
- Urban-Sparse-Bare rock

DATA SOURCES
Title; Owner; ID; Intended Scale; Publication Date.

1991 Hypsography (100ft); Los Alamos National Laboratory, Remediation Services Project; NA; Unknown; 1991.

1991 Hypsography (20ft); Los Alamos National Laboratory, Remediation Services Project; NA; Unknown; 1991.

Boundary of Department of Energy Property In and Around the Los Alamos National Laboratory; Site and Project Planning Group; NA; Unknown; February 1, 2003.

Cerro Grande Extent, Los Alamos National Laboratory, Earth and Environmental Sciences GISLab; NA; Unknown; April 5, 2002.

Roads, surfaced; Los Alamos County; ER2004-0071; Unknown; 2004; Provided by the Los Alamos National Laboratory Remediation Services Project.

Vegetation Taxonomy; Los Alamos National Laboratory, Environmental Stewardship Ecology Group; NA; Unknown; June 19, 2003.

LA-UR-04-8246

Cartography by: Doug Walther
Date: December 21, 2004
GISLab IM#: 201262
Request #: 13572, 13580, 13690

GISLab
Los Alamos National Laboratory

State Plane Coordinate System, New Mexico Central Zone, 1983 North American Datum. Units Feet.

Grid displays New Mexico State Plane coordinates in feet.

1:24,000

0 2,000 4,000 8,000 Feet

0 600 1,200 1,800 Meters

N

Los Alamos

Work performed on behalf of the RRES Remediation Services Project, Los Alamos National Laboratory, Los Alamos, NM, 87545.