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Screening Level Ecological Risk Assessment Approach for the Environmental Restoration Project at Los Alamos National Laboratory

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Acknowledgments

The approach for screening level ecological risk assessments described in this document has largely been shaped by guidance from the New Mexico Environment Department (NMED). The approach (and the authors) have benefited greatly from discussions with Barbara Toth from NMED's Hazardous and Radioactive Materials Bureau and Ralph Ford-Schmid from NMED's DOE Oversight Bureau. The authors are also indebted to Roger Ferenbaugh, Wayne Hansen, Orrin Myers, Michael Ebinger, David Breshears and David Bradbury, whose previous work provided considerable material for this document. The document has also benefited from careful review from members of the Environmental Restoration Project's ecological risk assessment team. The authors are particularly grateful to Joe Mose of DOE's Los Alamos Area Office for his guidance and support during the development of this document.

Executive Summary

This document describes the approach used by the Los Alamos National Laboratory (the Laboratory) Environmental Restoration (ER) Project for screening level assessments of potential, adverse impacts to ecological resources resulting from legacy wastes (wastes resulting from past operations at the Laboratory). This approach follows the New Mexico Environment Department's Hazardous and Radioactive Materials Bureau (NMED/HRMB) guidance dated March 4, 1998 (NMED 1998), the "Ecological Risk Assessment Guidance for Superfund" released in 1997 (EPA 1997), and the Environmental Protection Agency's (EPA) "Proposed Guidelines for Ecological Risk Assessment" (EPA 1996).

The purpose of this document is twofold: (1) to provide a basis for reaching consensus with regulators, managers, and other interested parties as to the best approach for conducting screening level ecological risk investigations at the Laboratory, and (2) to provide guidance to ER ecological risk assessors that will promote consistency in ecological screening investigations and the reporting of investigation results. It is anticipated that the ecological risk assessment approach described in this document will continue to improve, especially as baseline assessment methods are developed and experience is gained through field application of the screening methods.

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

Section 1 provides a brief introduction to the document. Section 2 provides an overview of the ER screening assessment process (including a process flow diagram). This section explicitly links the ER screening steps to the NMED Risk Based Decision Tree (NMED 1998), which is provided in Appendix A.

Section 3 describes the Laboratory-wide information that is needed for the screening-level ecological risk problem formulation, including the environmental setting, contaminant fate and transport, exposure pathways, and food webs. This laboratory-wide information provides the basis for the specification of screening level ecological receptors (Section 3.5) and assessment endpoints (Section 3.6).

Section 4, the longest and most complex section, describes in detail the two phases of the screening assessment: the scoping evaluation (Section 4.1) and the screening evaluation (Section 4.2). The scoping evaluation includes (1) the data assessment step, which identifies the list of contaminants of potential concern (COPCs) at the potential release site (PRS), (2) the problem formulation step for the specific PRS under investigation, and (3) the bioaccumulation evaluation step, which evaluates the level of concern for persistent bioaccumulation and/or biomagnification from contaminants at the PRS. The basis for the site-specific problem formulation is found in the scoping checklist. The scoping checklist 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 (EESCM). The scoping checklist also provides the basis for evaluating the adequacy of the data for ecological risk screening. The scoping checklist is provided in Appendix B.

The screening evaluation includes the calculation of hazard quotients (HQs) and hazard indices (HIs) for all COPCs and all appropriate screening receptors. The HQ can be thought of as the ratio of the calculated exposure dose to the receptor (based on contaminant levels at the PRS) to a dose that has been determined to be acceptable (based on toxicity studies for the receptor). An HI is a sum of HQs, across contaminants with like effects, for a given screening receptor. An HQ or HI greater than 1 is considered an indicator of potential adverse impacts, and the chemical constituents resulting in an HQ or HI greater than 1 are identified as contaminants of potential ecological concern (COPECs). HQ calculations require toxicity, bioconcentration and bioaccumulation information for all chemicals for all receptors. This information is

not provided in this document. NMED requires that the Laboratory document this information in detail to ensure that the best available information is used to develop HQs. The Laboratory is now in the process of developing toxicity and bioaccumlation/bioconcentration factor databases to meet these requirements. These databases will be provided in a companion document.

Section 4.3 describes the uncertainty analysis that follows the COPEC identification. This section describes the key sources of uncertainty in the screening assessment. The uncertainty analysis can result in adding chemical constituents to or removing them from the list of COPECs.

The results of the screening assessment are interpreted in the context of a risk management decision. This step is described in Section 4.4. Possible decisions include a recommendation of the appropriate corrective action, in terms of ecological concerns. Possible recommendations include ecological NFA, voluntary corrective action (VCA), voluntary corrective measure (VCM), and corrective measures study/corrective measure implementation (CMS/CMI), any of which will be incorporated into an integrated risk management decision to include human health risk evaluations, ground and surface water issues, and other applicable regulations. If the data are not adequate to support a recommendation, further investigation will be conducted to support an aggregate or baseline risk assessment.

List of Acronyms and Abbreviations

AA	administrative authority
ARAR	applicable or relevant and appropriate regulations
BAF	bioaccumulation factor
BCF	bioconcentration factor
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
COPC	contaminant of potential concern
COPEC	contaminant of potential ecological concern
DOE	Department of Energy
EESCM	ecological exposure site conceptual model
EPA	Environmental Protection Agency
EQL	estimated quantitation limit
ER	Environmental Restoration
ESL	ecotoxicity screening level
ESRV	ecotoxicity screening reference value
FIMAD	Facility for Information Management, Analysis, and Display
GIS	geographical information system
HI	hazard index
HQ	hazard quotient
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
Laboratory	Los Alamos National Laboratory
LOAEL	lowest-observed adverse effect level
MDL	method detection limit
NFA	no further action
NMED/HRMB	New Mexico Environment Department's Hazardous and Radioactive Materials Board
NOAEL	no observed adverse effect level
OU	operable unit
PQL	practical quantitation limit
PRS	potential release site
RCRA	Resource Conservation and Recovery Act
RFI	RCRA facility investigation
SCM	site conceptual model
SMDP	scientific/management decision point
T&E	threatened and endangered
TA	technical area
UCL	upper confidence level
VCA	voluntary corrective action
VCM	voluntary corrective measure
WQCC	Water Quality Control Commission

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Screening Level Ecological Risk Assessment Approach

for the Environmental Restoration Project at

Los Alamos National Laboratory

1.0 Introduction

This document describes the approach used by the Los Alamos National Laboratory (the Laboratory) Environmental Restoration (ER) Project for screening level assessments of potential, adverse impacts to ecological resources resulting from legacy wastes (wastes resulting from past operations at the Laboratory). This approach follows the New Mexico Environment Department's Hazardous and Radioactive Materials Bureau (NMED/HRMB) guidance dated March 4, 1998 (NMED 1998) and the Ecological Risk Assessment Guidance for Superfund dated June 5, 1997 (EPA 1997). The NMED guidance includes a "Risk-Based Decision Tree," which is referred to often in this document and is provided in Appendix A.

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

2.0 Ecological Screening Process

The ecological screening process flow diagram is shown in Figure 2.1. The screening process is composed of three parts, the scoping evaluation, the screening evaluation, and the risk management decision, which is based on an interpretation of the screening results. The first step of the scoping evaluation is to determine if the potential release site (PRS) is a candidate for an administrative no further action (NFA) decision based on the following NMED criteria:

- NFA criterion 1 (site does not exist)
- NFA criterion 2 (site never used for solid waste or hazardous wastes)
- NFA criterion 3 (documentation of no release through an evaluation of process knowledge)

The ER Project personnel provide the justification for administrative NFA recommendations. Given one of the above criteria, environmental sample information is usually not required, and ecological evaluations are unnecessary.

During the data assessment (documented in the Resource Conservation and Recovery Act [RCRA] Facilities Investigation [RFI] report), contaminants of potential concern (COPCs) are identified by comparing the maximum constituent concentrations to levels approved by the administrative authority (AA), including any of the following:

- background for inorganic constituents, fallout for radionuclide concentrations, or method detection limits (MDLs), practical quantitation limits (PQLs), or estimated quantitation limits (EQLs) for organic constituents (Box 2, criterion 3 of the NMED/HRMB Risk-Based Decision Tree, Appendix A), and
- standards or other approved values (Box 2, criterion 4 of the NMED/HRMB Risk-Based Decision Tree, Appendix A).

AA-approved standards exist for surface water in the form of Water Quality Control Commission (WQCC) wildlife and livestock watering standards. There are no AA-approved soil or sediment standards at this time. If there are no COPCs (that is, none of the maximum constituent values exceed AA-approved levels), then the PRS may be recommended for NFA (Boxes 3 and 4 of the NMED/HRMB Risk-Based Decision Tree, Appendix A). The ER Project personnel provide the justification for these recommendations in the RFI report and further ecological evaluations are unnecessary.

Any PRSs that are not proposed for NFA by this point must undergo further ecological scoping, including a site visit by a member of the ecological risk assessment task team and completion of the scoping checklist (described in detail in Section 4.1 and presented in Appendix B). The ecological exposure site conceptual model (EESCM) is developed during scoping, and fate and transport issues are assessed (Boxes 5 and 10 of the NMED/HRMB Risk-Based Decision Tree, Appendix A). The aggregation issue is also addressed during scoping (i.e., should other PRSs be combined with this PRS in an aggregate assessment?). After the scoping evaluation, if the ecological risk assessment team determines that the PRS or PRS aggregate poses no threat to the environment because there are no ecological receptors and/or there are no pathways to receptors, a recommendation for ecological NFA is made. The justification for this recommendation is documented in the Ecological subsection of the Screening Assessments section of the RFI report. This recommendation is then evaluated along with potential human health impacts and surface water, groundwater, and other regulatory requirements, to make an integrated site recommendation.

During scoping and data assessment, a decision is made about the adequacy of the data and the EESCM for the screening evaluation (Figure 2.1). At a minimum, the ecological screening evaluation must be performed for all relevant media (e.g., soil, water, or air) that have a significant ecological exposure pathway as defined in the EESCM. Before screening calculations can be performed, PRS- or aggregate-specific data must be deemed adequate for characterizing the nature, rate, and extent of contamination in order to justify use of the sample maximums as reasonable estimates for the highest concentrations expected at the PRS or aggregate. If data do not exist for the PRS or aggregate, a recommendation must be made to

collect site-specific data. If existing data may not represent the highest contaminant levels, the benefits of collecting additional data should be evaluated against the bias in the current sample maximum values.

In the final step of the scoping evaluation the PRS or aggregate is evaluated for bioaccumulation potential (Boxes 6 and 7 of the NMED/HRMB Risk-Based Decision Tree, Appendix A). The first step of the bioaccumulation evaluation is to assess the presence of "persistent bioaccumulators and biomagnifiers," which requires operational definitions of relevant terms.

There are three terms describing similar processes for biological transfer of chemical constituents that are important for exposure assessment: bioaccumulation, bioconcentration, and biomagnification. Because these terms are sometimes confused, definitions (as used in this document) are provided below. The most broadly applicable term, bioaccumulation, is defined by Maughan (1993) as occurring "when contaminants are passed between organisms through trophic as well as nontrophic means."

The bioaccumulation factor (BAF) is defined as the ratio of the concentration of a chemical constituent in the tissue of an organism (or the organism as a whole) to the concentration of the constituent in its food or environmental media. It should be emphasized that bioaccumulation is a very broadly applicable term as it implies both nontrophic (absorption) and trophic (ingestion) pathways to the receptor. Transfer of chemical constituents by trophic pathways alone is always distinguished as bioaccumulation. Bioconcentration and biomagnification can be considered special cases of bioaccumulation and are useful terms for clarifying transport pathway processes in the biotic environment. Maughan (1993) defines bioconcentration of contaminants as occurring "when organisms intake and retain contaminants through nontrophic means."

The bioconcentration factor (BCF) is defined as the ratio of the concentration of a chemical constituent in specific tissues of an organism (or the organism as a whole) to the concentration of the constituent in abiotic environmental media. Nontrophic means include absorption of chemical constituents vis-à-vis environmental media; e.g. uptake by plants from absorption of interstitial water, inhalation and dermal pathways in animals, and active or diffuse transfer across permeable tissues (such as the gills of aquatic organisms).

Biomagnification is defined by Maughan (1993) as occurring "when each successive trophic level has increased contaminant concentrations, relative to their food source."

The BCF is most commonly calculated as the steady-state or equilibrium-state ratio of the concentration of a potential toxicant in water to the concentration of the constituent in an organism's fresh tissue. The BCF, as used in this document, applies to the uptake of chemical constituents by plants, soil-dwelling invertebrates, and aquatic organisms through nontrophic means. The BAF will therefore apply when the transfer of a chemical constituent implies trophic only or trophic and nontrophic mechanisms of intake.

Although the EPA has no guidance defining critical values for bioaccumulation estimators, NMED/HRMB specifies bioaccumulators as contaminants with a bioconcentration/bioaccumulation factor (BCF/BAF) greater than or equal to 40, or an organic constituent with the logarithm of the octanol/water partitioning coefficient (log K_{ow}) greater than or equal to 4 (Box 6 of the NMED/HRMB Risk-Based Decision Tree, Appendix A). The interpretation of bioaccumulators in this context is appropriately those chemical constituents that have the potential to be "persistent bioaccumulators and biomagnifiers" (Ralph Ford-Schmid, State of NM DOE Oversight Bureau, personal communication). This convention is adopted in this document. Persistent bioaccumulators are those chemical constituents that cannot be sufficiently metabolized or excreted such that they accumulate to concentrations within the organism to cause toxicologically observable effects. The current list of NMED potentially persistent bioaccumulators and potential biomagnifiers, is provided in Table 2-1. It should be noted that this list is not exhaustive and there may be other chemicals at a site that need to be evaluated for bioaccumulation concerns (e.g., pesticides not on the list). It should also be noted that the chemicals on this list are only potentially persistent bioaccumulators and biomagnifiers. If they occur at a site, then further evaluation is needed to determine if they will in fact be persistent and/or biomagnify given the environmental conditions specific to the site

under investigation (e.g. some of these chemicals present bioaccumulation concerns in aquatic environments only).

The bioaccumulation evaluation includes determining if the potentially persistent bioaccumulators and biomagnifiers can build up to a level of concern in the environment directly at the PRS or aggregate, or offsite (in an aquatic environment) through a transport mechanism (Box 7 of the NMED/HRMB Risk-Based Decision Tree, Appendix A). If, as a result of this evaluation, persistent bioaccumulation and biomagnification are of concern, then the screening assessment proceeds immediately to a risk management decision or scientific management decision point (SMDP) as described in the Superfund guidance for ecological risk assessment (EPA 1997) and Box 8 of the NMED/HRMB Risk-Based Decision Tree (Appendix A).

Table 2-1. List of Potentially Persistent Bioaccumulators and Biomagnifiers

Volatile and Semivolatile Organics	**PCBs/Pesticides**
Bis(2-ethylhexyl)phthalate	All Aroclors
Butyl benzyl phthalate	beta-BHC
Dibenzofuran	BHC-mixed isomers
Dichlorobenzene[1,4-]	Chlordane
Di-n-butyl phthalate	Chlorecone (Kepone)
Di-n-octyl phthalate	DDT and metabolites
Trichlorobenzene[1,2,4-]	Dieldrin
Acenaphthene	Endosulfan
Anthracene	Endrin
Benzo(a)anthracene	Heptaclor
Benzo(a)pyrene	Lindane
Benzo(b)fluoranthene	Methoxyclor
Benzo(g,h,i)perylene	Toxaphene
Benzo(k)fluoranthene	
Chrysene	**Inorganics**
Dibenzo(a,h)anthracene	Aluminum
Fluoranthene	Cadmium
Fluorene	Copper
Indeno(1,2,3-cd)pyrene	Lead
Phenanthrene	Mercury
Pyrene	Nickel
Pentachloronitrobenzene	Selenium
Pentachlorophenol	
Xylene (mixed isomers)	**Radionuclides**
	Americium-241
Dioxins/Furans	Cesium-137
2,3,7,8-tetrachloro-dibenzo(p)dioxin	Plutonium-238,239,240
2,3,7,8-tetrachloro-dibenzo(p)furan	Radium-226,-228
	Strontium-90
	Thorium-228,-230,-232
	Uranium-234,-235,-238

The first consideration in the risk management decision will be to identify interim actions to reduce or eliminate the transport of persistent bioaccumulators and biomagnifiers off site (to aquatic environments). The risk management decision will be made to minimize ecological injury, and will consider the impact of cleanup actions on the environment. A screening step is not formally part of this risk management decision, but decision-makers may need information on the relative toxicity of contaminated sites to make a decision to remediate a PRS or aggregate. The risk management decision will consider corrective actions, including cleanup to approved site background levels, cleanup to detection levels for manmade organic constituents,

or cleanup to risk-based concentrations (Box 8 of the NMED/HRMB Risk-Based Decision Tree, Appendix A). In some cases the data may not be adequate to support the risk management decision and further investigation will be conducted. In the case of cleanup to risk-based concentrations, it may be necessary to conduct further investigation to support a risk assessment to develop the cleanup levels. Because loss of habitat is a major ecological concern, cleanup decisions may need to include comparative risk evaluations of habitat loss and disruption versus potential risks from contamination. If the evaluation shows that persistent bioaccumulation or biomagnification are not of concern, the PRS or aggregate enters the screening evaluation (Box 11 of the NMED/HRMB Risk-Based Decision Tree, Appendix A).

In the screening evaluation, a hazard quotient (HQ) is calculated for each COPC for each screening receptor. The selection of appropriate screening receptors is an important step in ecological risk screening (see Section 3.5). Currently, eight terrestrial receptors have been identified for screening: a "generic" plant, an earthworm (Family *Megadrili*), the deer mouse *(Peromyscus maniculatus)*, the vagrant shrew *(Sorex vagrans)*, the desert cottontail *(Sylvilagus audubonii)*, the American kestrel *(Falco sparverius)*, the American robin *(Turdus migratorius)*, and the red fox *(Vulpes vulpes)*. In addition four aquatic receptors have been selected for screening, algae, daphnids *(Crustacea)*, snails *(Gastropoda)*, and a generic bony fish. The HQ can be thought of as the ratio of the calculated exposure dose to the receptor (based on COPC levels at the PRS) to a dose that has been determined to be acceptable (based on toxicity studies for the receptor). An HQ greater than 1 is considered an indicator of potential adverse impacts. Details on HQ calculations are provided in Section 4.2 of this document. Hazard quotients for nonradionuclide COPCs are summed separately from HQs for radionuclide COPCs to determine the respective hazard indices (HIs) for each receptor. If the HIs are all less than 1, there are no contaminants of potential ecological concern (COPECs). If any of the HIs are greater than or equal to 1, COPECs have been identified (Box 12 of the NMED/HRMB Risk-Based Decision Tree, Appendix A).

The HQ and HI calculations are followed by an uncertainty analysis that focuses on key sources of uncertainty in the screening assessment and can result in the addition or deletion of COPECs (Box 11.f of the NMED/HRMB Risk-Based Decision Tree, Appendix A). If adequate toxicity information is not available to calculate HQs for all receptors for the COPC, the COPC is retained as a COPEC and enters the uncertainty analysis. The main components of the uncertainty analysis are described in Section 4.3 of this document.

Following the uncertainty analysis, the results of the screening assessment are interpreted in the context of a risk management decision or SMDP (Boxes 13 and 14 of the NMED/HRMB Risk-Based Decision Tree, Appendix A). The details of this step are described in Section 4.4 of this document. If the data are adequate, a recommendation of the appropriate corrective action, in terms of ecological concerns, can be made. Possible recommendations include ecological NFA, voluntary corrective action (VCA), voluntary corrective measure (VCM), and corrective measures study/ corrective measures implementation (CMS/CMI) any of which will be incorporated into an integrated SMDP to include human health risk evaluations. If the data are not adequate to support a recommendation, further investigation will be conducted to support an aggregate or baseline risk assessment (Box 15 of the NMED/HRMB Risk-Based Decision Tree, Appendix A).

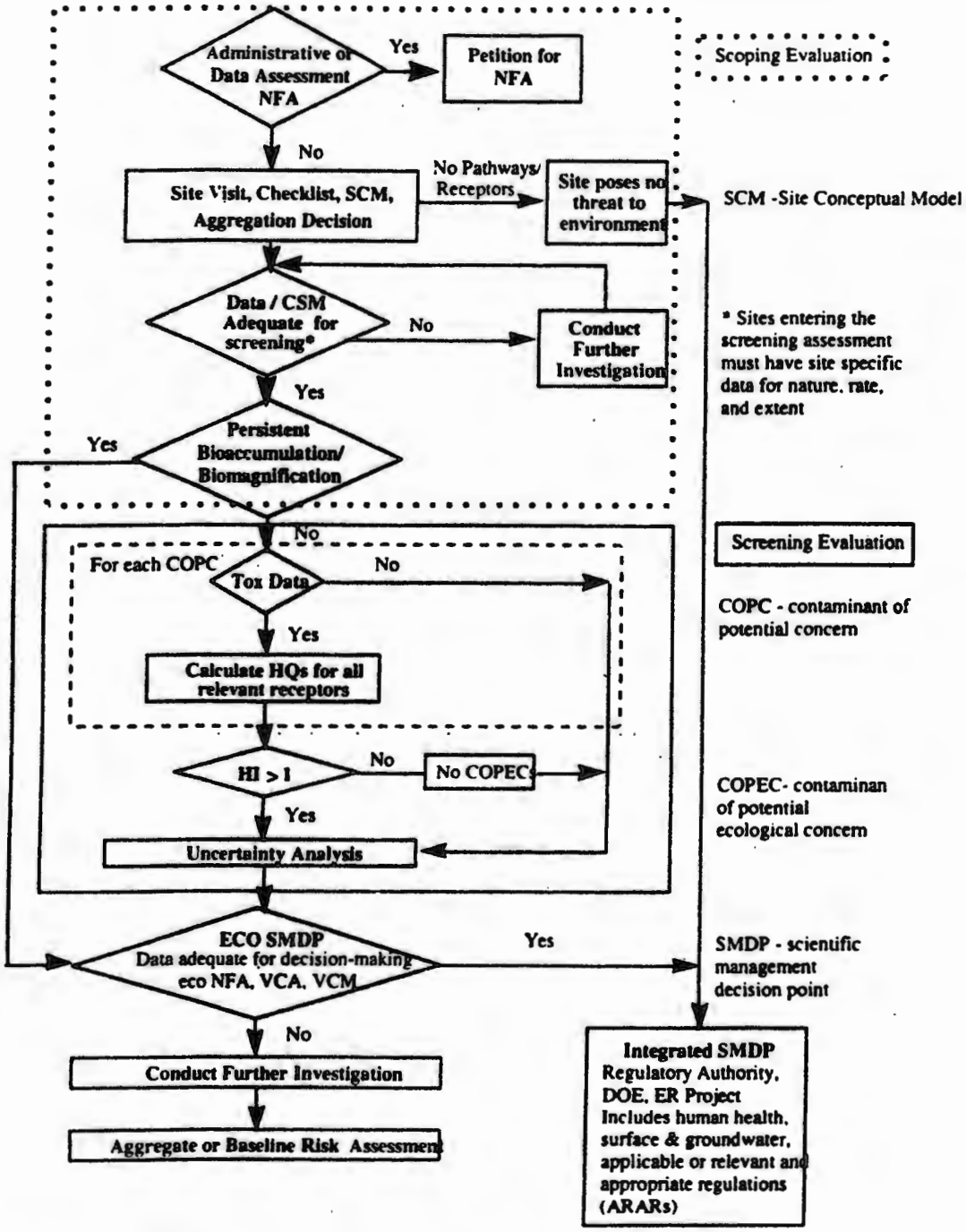

Figure 2.1. Process flow for ecological screening assessment.

3.0 Generic Problem Formulation for Ecological Screening Assessments

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

1. Environmental (physical and biological) setting
2. Contaminant fate and transport
3. Screening receptor categories
4. Exposure pathways

Problem formulation at Los Alamos, therefore, requires understanding of the physical and biological setting of the Laboratory. The physical setting greatly influences the potential contaminant transport pathways, which also influence the potential exposure pathways for ecological receptors. The biological setting is important for receptor selection, since 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. Thus, understanding the feeding relationships among animals and plants can be used to develop rational groups of ecological receptors. Receptor groupings, based on feeding relationships, are an efficient and effective way to represent all ecological resources (biota) of concern. In the following sections, the physical setting will be summarized first and followed by descriptions of the salient biotic features.

3.1 Environmental Setting

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

3.1.1 Geographic Setting

The Laboratory and residential and commercial areas of Los Alamos and White Rock are located in Los Alamos County, in north-central New Mexico, approximately 60 miles north-northeast of Albuquerque and 25 miles northwest of Santa Fe (Figure 3.1). The surrounding land is largely undeveloped, with large tracts of land north, west, and south of the Laboratory held by the Santa Fe National Forest, Bureau of Land Management, Bandelier National Monument, General Services Administration, and Los Alamos County. The Pueblo of San Ildefonso borders the Laboratory to the east.

The Laboratory is divided into technical areas (TAs) that are used for building sites, experimental areas, waste disposal locations, roads, and utility rights-of-way (see Figure 3.2). However, these uses account for only a small part of the total land area. Most land provides buffer areas for security and safety and is held in reserve for future use. Thus, the majority of the Laboratory is undeveloped land that supports diverse and abundant ecological resources.

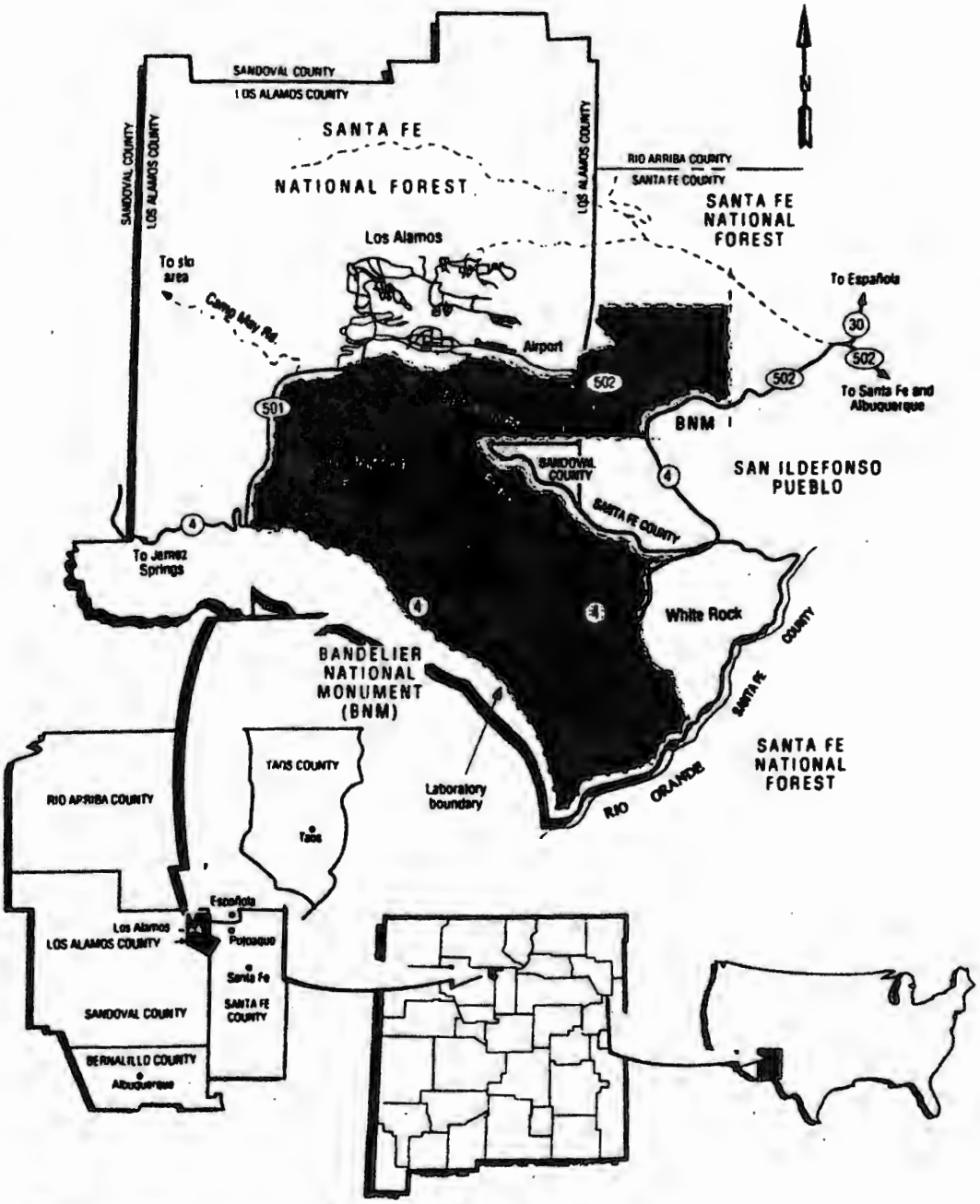

Figure 3.1. Regional location of Los Alamos National Laboratory.

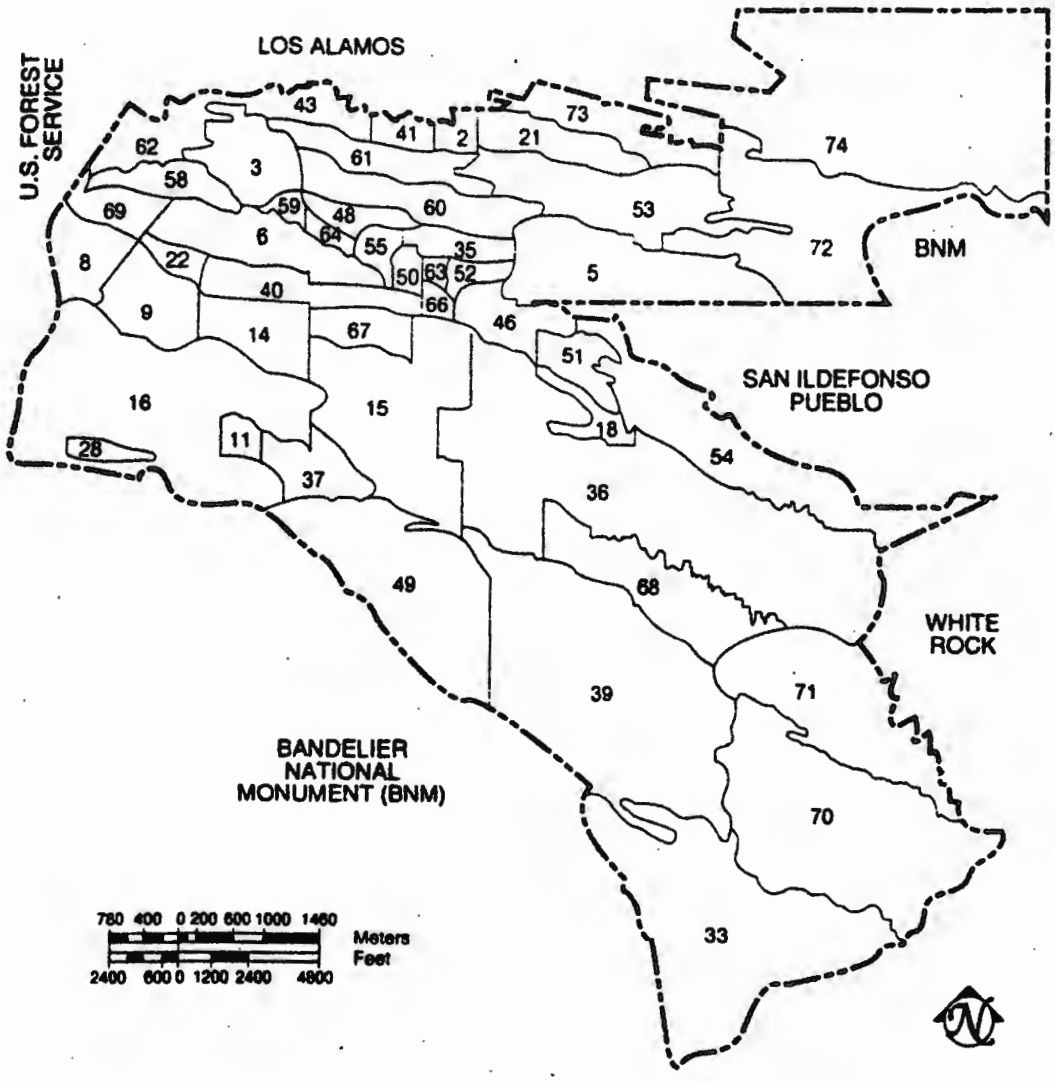

Figure 3-2. Technical areas of Los Alamos National Laboratory in relation to surrounding landholdings.

3.1.2 Climate

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. High-intensity thunderstorms in the summer can cause erosion of unstabilized sediment or soil. The form, frequency, intensity, and evaporation potential of precipitation can strongly influence surface water runoff and infiltration of contaminants (Section 3.2). The speed, frequency, direction, and persistency of wind can influence the airborne transport of contaminants. High winds, which are common in the spring, can result in atmospheric transport of contaminants (see Section 3.2).

3.1.3 Geology

The geology associated with the Laboratory is described in more detail in Chapter 2 of the Installation Work Plan (IWP) (LANL 1996). The geology and hydrology information provided in this section forms the basis for the discussion of hydrologic transport.

The Laboratory extends over the east-sloping, dissected tableland of the Pajarito Plateau, and is bounded on the west by the eastern Jemez Mountains and on the east by White Rock Canyon of the Rio Grande. The geology of the Pajarito Plateau primarily reflects ancient volcanism in the Jemez Mountains and surrounding areas. The Rio Grande rift lies to the east of the plateau, forming a series of north-south trending fault troughs from southern Colorado to southern New Mexico. Most of the finger-like mesas in the Los Alamos area (Figure 3.3) are formed in Bandelier Tuff, which includes ash fall, ash fall pumice, and rhyolite tuff. The tuff is more than 1000 ft thick in the western part of the plateau and thins to about 260 ft eastward above the Rio Grande. It was deposited as a result of major eruptions in the Jemez Mountains' volcanic center about 1.2 to 1.6 million years ago. Deep canyons are incised into the Bandelier Tuff and expose it to depths of up to several hundred feet below the upper elevation of the plateau. Some of the deeper canyons expose older lava deposits and sedimentary rocks.

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

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

3.1.4 Hydrology

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

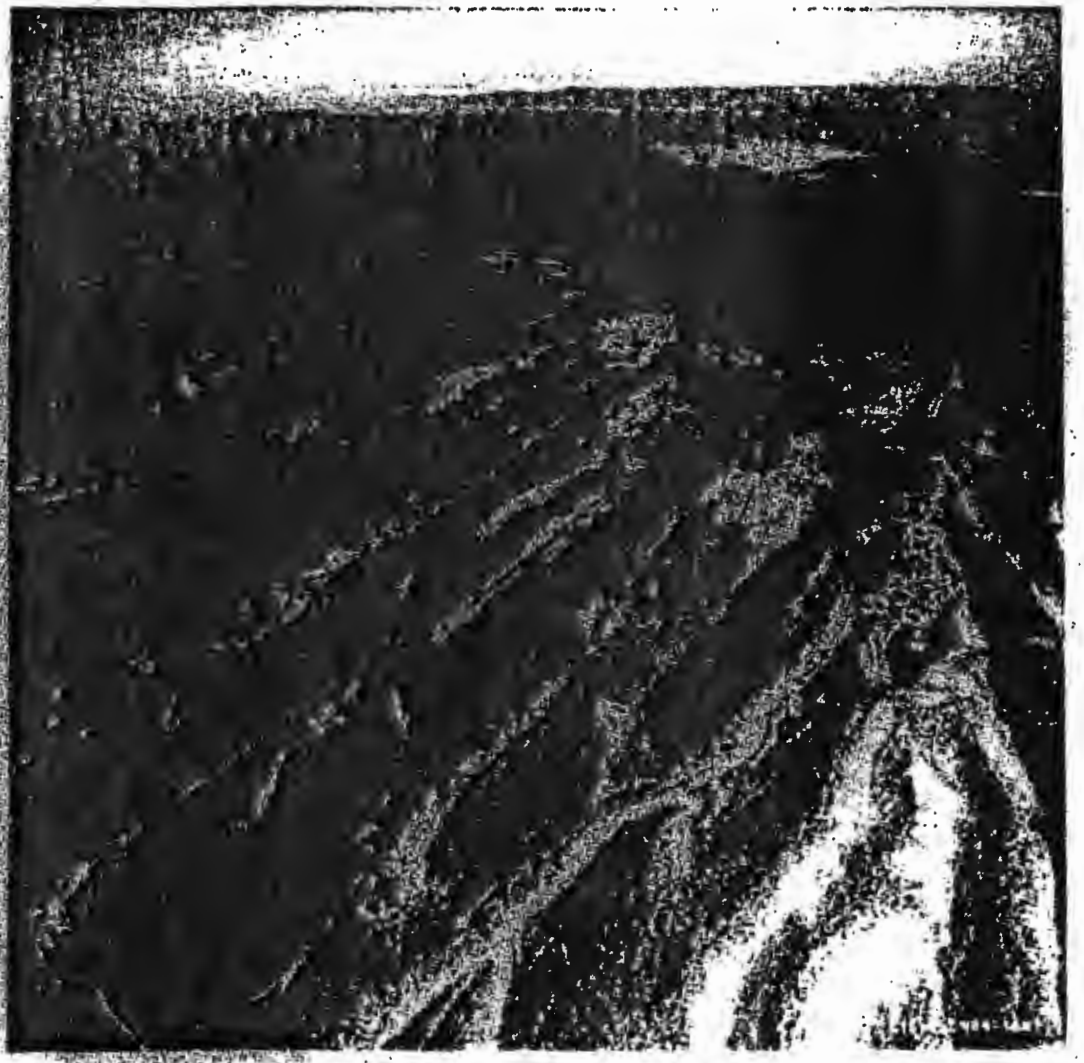

Figure 3.3. Topography of the Los Alamos area.

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

Groundwater in the Los Alamos area occurs in three forms: (1) water in shallow alluvium in canyons, (2) perched water (a body of groundwater above a less permeable layer that is separated from the underlying regional aquifer by an unsaturated zone), and (3) the regional aquifer of the Los Alamos area.

3.1.5 Biology

Biota found on or near the Laboratory property include approximately 500 plant species, 29 mammal species, 200 bird species, 19 reptile species, 8 amphibian species, and many hundreds of insect species. Roughly twenty species are designated as either threatened and endangered species or "species of special concern" by the federal and/or state government.

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

The Laboratory has recently developed a vegetation land cover map (Figure 3.4) for the purpose of locating habitat that is suitable, or potentially suitable, for T&E species (Koch et al. 1997). The land cover map identifies areas by the dominant overstory vegetation. The map was developed using the Iterative Self-Organizing Data Analysis Technique to interpret a 1992 Landsat thematic mapper image into thirty classes. The thirty classes were then aggregated into ten land cover types through field surveys, aerial photo interpretation, and the incorporation of topographic information. The resulting cover types include major vegetation zones and physiognomic types that are important to the distribution and abundance of several T&E species (Koch et al. 1997). The areal extent of each cover type on Laboratory property is provided in Table 3-1.

The land cover types can be subdivided into types that correspond to the major elevation and climatic gradient of the region and those that correspond to edaphic, topographic, or moisture criteria (Koch et al. 1997). The elevation and climatic gradients in the LANL region most strongly influence four vegetative cover types defined by their dominant tree species and by their structural characteristics (shown in Figure 3.4): juniper savannas, piñon-juniper woodlands, ponderosa pine forests, and mixed conifer forests. In contrast, aspen forests, grasslands, open water, and unvegetated lands are not primarily influenced by elevation and climatic gradients. Instead, they are most strongly influenced by topographic features, soils and geologic conditions, and moisture levels. Steep terrain or clouds cause the shadowed areas (identified as unclassified on the map shown in Figure 3.4).

Table 3-1. Areal extent of land cover types at the Los Alamos National Laboratory[a]

Cover Type	Area (mi^2)	Proportion Area (%)
Mixed Conifer	1.3	3
Aspen	0.1	0.1
Ponderosa Pine	12.6	25
Piñon-Juniper	20.0	40
Juniper Savanna	1.6	3
Grassland	2.9	6
Water	0.04	0.1
Unvegetated	2.9	6
Developed	8.6	17
Unclassified (Shadows)	0.2	0.4
Total	**50.2**	**100**

[a] Modified from Koch et al. 1997 (an estimated 7 mi^2 of developed land associated with the Los Alamos town area was added).

Vegetation Cover Types

Juniper savannas. One-seed juniper (*Juniperus monosperma*) is the dominant overstory species in the juniper savanna. Canopy coverage for this species typically ranges between ten and thirty percent. Piñon (*Pinus edulis*) may also be widely scattered. Landscapes along the Rio Grande from Frijoles Canyon (elevation 1634 m, 5360 ft) to Otowi Bridge (elevation 1681 m, 5513 ft) are primarily vegetated by the juniper savanna cover type. Juniper savanna communities also extend approximately to an elevation of 1768 m (5800 ft) in the bottoms of adjacent canyons.

Piñon-juniper woodlands. The dominant tree species in piñon-juniper woodlands are one-seed juniper or piñon. Although piñon-juniper woodlands can extend to elevations as low as 1650 m (5500 ft) on protected topographic positions, they are the dominant, upland community type between 1740 and 2100 m (5800 and 7000 ft) in elevation (Koch et al. 1997). They also can be found as high as 2160 m (7200 ft) on south-facing slopes.

Ponderosa pine forests. Ponderosa pine (*Pinus ponderosa*) is the dominant tree species in the ponderosa pine cover type. One-seed juniper and piñon may also be present, particularly at lower elevations. At higher elevations, Douglas fir (*Pseudotsuga menziesii*) and Rocky Mountain juniper (*Juniperus scopulorum*) can be found in ponderosa pine forests. Ponderosa pine forests extend to elevations as low as 1860 m (6200 ft) in some of the protected canyons in the Laboratory region. At these lower extremities ponderosa pine forests blend with piñon-juniper woodlands. On the mesas and the lower slopes of the Sierra de Los Valles, ponderosa pine forests extend to 2340 m (7800 ft) in elevation. They may also be found at higher elevations, up to 2610 m (8700 ft), on steep, south-facing slopes.

Mixed conifer forests. Mixed conifer forests begin above 2070 m (6900 ft) in elevation, blended with ponderosa pine communities, but also extend to lower elevations on north aspects of the canyons. These communities continue to the highest elevations of the Sierra de los Valles, 3149 m (10 496 ft). Douglas fir and white fir (*Abies concolor*) are the typical overstory dominants in mixed conifer forests. At elevations above 2700 m (9000 ft), Engelmann spruce (*Picea engelmannii*) becomes more important. Ponderosa pine and aspen (*Populus tremuloides*) are also typically present. Limber pine (*Pinus flexilis*) can also be found in mixed conifer forests, especially on rocky ridgelines.

Aspen forests. Aspen (*Populus tremuloides*) communities are common at mid-elevations in the mountains, from approximately 2700 m to 3030 m (8900 ft to 9950 ft). Below 2820 m (9250 ft), aspen stands occupy north and northeast aspects, whereas above this elevation they are mostly found on southeast- to southwest-facing positions. At higher elevations and on southerly aspects, aspen typically exceeds forty-five percent 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 thirty percent of the overstory coverage. Depending on the fire history of the specific stand, other tree species, such as ponderosa pine and limber pine, may be blended with aspen.

Grassland. Grasslands are dominated by grasses, narrow-leaf plants, or species that dominate disturbed areas (colonizing species). Forbes and other non-shrubby species may be dominant components of these communities. Shrubs and trees are absent or rare. The grassland cover type consists of a wide range of communities, including areas undergoing post-fire succession, abandoned homestead areas, montane meadows, and subalpine grasslands.

Open water. This cover type includes all land that is at least periodically flooded or is open water. In the wettest of these sites, the vegetative cover is limited to plant species that require or prefer permanent or seasonally mesic conditions. In general, these cover types are marshes, lakes, rivers, and streams.

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

3.1.6 Wetlands

Definitions of wetlands adopted in this document follow the U.S. Fish and Wildlife Service "Classification of Wetlands and Deep Water Habitats of the United States" (Cowardin et al. 1979). Riparian/wetland ecosystems are directly associated with wetlands adjacent to rivers, stream banks, or canyon floors (e.g., marshes, bogs, and riverbank areas). Wetlands can be important in contaminant pathways since they are of central importance to both terrestrial and aquatic biota. Additionally, many of the organisms occupying wetlands are more susceptible to persistent bioaccumulators and biomagnifiers because of their means of respiration. 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. The few riparian areas or wetlands that occur at the Laboratory are too small to be resolved at the scale used in Figure 3.4. Larger wetland areas on the Laboratory include upper Sandia Canyon, lower Pajarito Canyon, and Mortandad Canyon. Naturally emergent wetlands (spring-fed wetlands and seeps) are found mostly in canyon bottoms. Anthropogenically influenced emergent wetlands may be found where canyon bottoms have been dredged or are associated with outfalls (Foxx 1996).

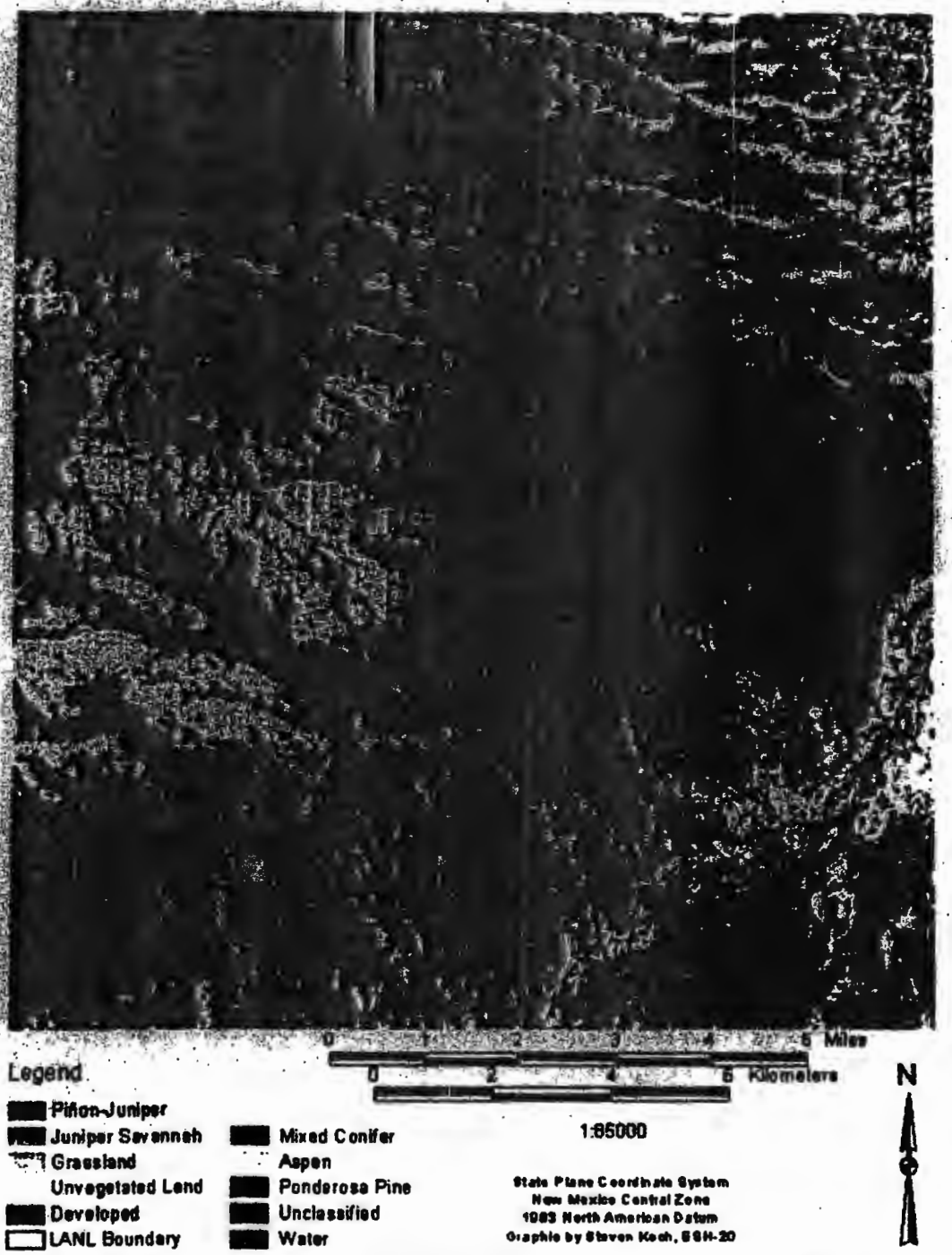

Figure 3.4. Land cover map for Los Alamos National Laboratory and vicinity (from Koch et al. 1997).

3.2 Contaminant Fate and Transport

The geomorphology of the Pajarito Plateau, with its alternating mesas and canyons, determines the primary contaminant transport pathways for sources of legacy environmental contamination. Figure 3.5 is a schematic showing the key transport pathways:

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

These pathways are discussed briefly below, and pathways applicable to a particular PRS or PRS aggregate will be discussed in the site-specific RFI report.

3.2.1 Hydrologic Transport

3.2.1.1 Surface Water and Sediment Transport

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

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

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

3.2.1.2 Groundwater Transport

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

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

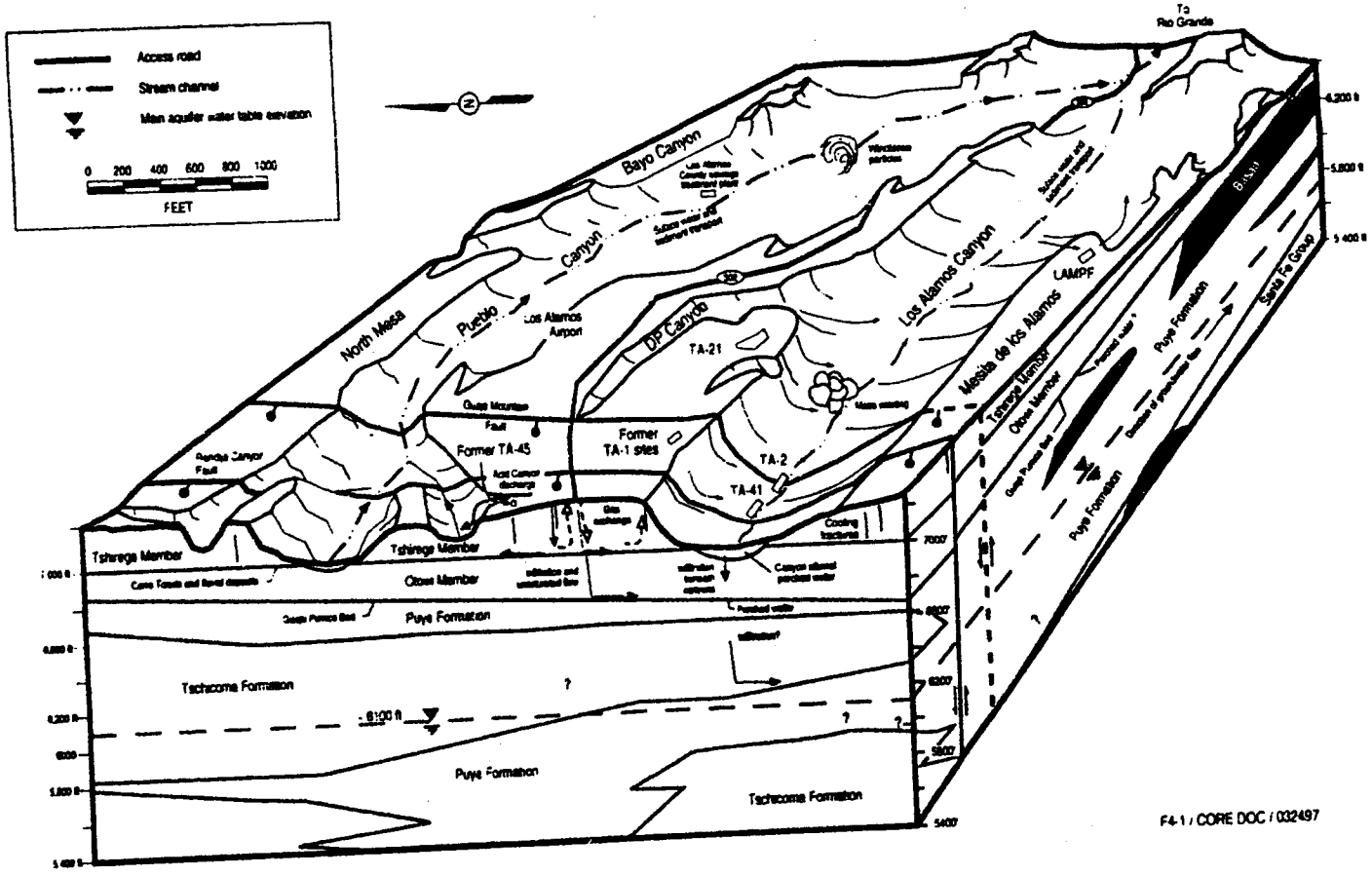

Figure 3-5. Schematic illustration of the conceptual model for contaminant transport, source LANL 1997

3.2.2 Physical Transport

Physical transport of surface or subsurface materials is most dramatically possible through a mechanism termed "mass wasting." Mass wasting is the process in which blocks of rock break off the cliffs and are deposited violently into the canyons. Mass wasting is an episodic phenomenon and could be an important mechanism of contaminant transport for mesa-top sites located near canyon walls. Exposure to ecological receptors would result if subsurface contamination became surficial contamination through mass wasting into the canyons. The transport pathways would then be similar to media subject to surface water transport. A much slower physical transport mechanism is surficial erosion through wind or water (Sections 3.2.1.1 and 3.2.3).

3.2.3 Atmospheric Transport

Atmospheric transport may occur through transport of windblown particles or vaporization of volatile chemicals. Transport of soil or fine sediment particles by wind can be a means of dispersing contaminants. Wind resuspension and transport of contaminant-laden soil or sediment is not believed to be a significant transport pathway.

3.3 Exposure Pathways

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

- rain splash of contaminated soil onto plants
- root uptake of water-soluble contaminants
- incidental ingestion of soil
- dermal contact with soil
- inhalation of soil
- food web transport (consumption of contaminated plants and animals)
- direct exposure to soil containing gamma-emitting radioactive contaminants

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

- ingestion of surface water
- foliar uptake of surface water
- incidental ingestion of sediments
- dermal contact with surface water or sediments
- inhalation of fine sediment materials during dry periods
- food web transport (consumption of contaminated plants and animals)
- direct exposure to sediments containing gamma-emitting radioactive contaminants

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

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

- respiration by animals or plants of contaminants present as vapors
- inhalation of particulates
- deposition of particulates on foliage
- deposition of particulates on animals, and subsequent ingestion during grooming

3.4 Functional Food Web

The food web diagram is important for evaluating dietary exposure pathways and for specifying ecologically relevant groups of organisms for exposure assessment. The food web structure captures functionally relevant biotic assimilation and associative relationships and is key for receptor selection.

A food web diagram shows pathways of food consumption in a biotic system by means of boxes and connecting arrows. Boxes in a food web diagram represent biota, explicitly defined as functional assemblages or as taxonomic groups, while arrows define the direction of energy flow between biota (e.g., from prey to predators). In developing a food web diagram, ecological receptors can be viewed from a taxonomic or functional perspective. The taxonomic perspective uses phylogenetic classification to organize all species present at the Laboratory into groups (e.g., class, family, or species associations). A taxonomic classification, for example, places rodents (class Mammalia), birds (class Aves) and ants (class Insecta) into different taxonomic groups and is insensitive to potentially similar feeding habits among these taxa.

For the purposes of this ecological risk assessment methodology, biological receptors are classified into functional groups that recognize similarity of feeding roles instead of a taxonomic classification. A "feeding guild" is a collection of species that share a common food consumption roles. For example, animals that eat seeds (granivores) are one feeding guild. A food web based on feeding guilds allows identification of critical ecological functions performed by members of the guilds. This feeding guild approach is more useful than a taxonomic approach because it recognizes potentially common exposure pathways by means of food web transport.

Figure 3.6 represents the functional food web for the Laboratory. The food web includes three basic trophic positions: producers (vascular and non-vascular plants), consumers (herbivores, omnivores, carnivores, and parasites), and decomposers. Therefore, a minimum of three receptors must be selected to represent these primary trophic associations. Within these basic trophic levels, several feeding guilds have been identified. For example, one group of consumers is herbivores, consisting of six feeding guilds: seed-eaters (granivores), fruit-eaters (frugivores), foliage or leaf-eaters (folivores), nectar and pollen feeders (nectarivores/pollen eaters), fungi eaters (fungivores), and browser/grazers.

3.5 Screening Receptors

As described in Section 3.1, Laboratory property supports numerous habitats with a variety of vegetation and wildlife, and any particular PRS may support a variety of plant and animal species. As a consequence, the selection of a set of receptors that includes representatives of every class of biota for every trophic level would result in an unwieldy number of receptors for use in ecological screening. Therefore, the rationale behind receptor selection is to select an appropriate set of receptors that satisfy the following criteria (based on Fordham and Reagan 1991):

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

Terrestrial Receptors

Table 3-2 summarizes the factors that led to the selection of the eight terrestrial screening receptors. A generic plant was selected primarily because producers are the major food base that directly and indirectly supports the entire food web. The use of a generic plant is also indicative of the broad-base taxonomic concern for plants in general, rather than any particular species. Additionally, plants form much of the physical habitat structure used by animal species. The generic plant is also used to represent several plant species of special concern present at the Laboratory.

The earthworm (Family *Megadrili*) was selected because it represents the important 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 desert cottontail *(Sylvilagus audubonii)* was selected because it is a strict herbivore (browser/grazer), and can be used as a sensitive receptor 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 to represent the importance of rodents as a food source for higher consumers (carnivores and omnivores), which makes it important to many food webs. The vagrant shrew *(Sorex vagrans)* was selected largely because of its high exposure to contaminants from grubbing for invertebrates in soil and because of its high-level intake of soil-dwelling invertebrates (including earthworms). The red fox *(Vulpes vulpes)* was selected because it represents a mammal with relatively high contaminant biomagnification potential due to its largely carnivorous feeding habits.

The American robin *(Turdus migratorius)* was selected because it is representative of birds that forage for ground-dwelling invertebrates as well as fruits, with relatively high potential exposure to contaminants from its diet. The American kestrel *(Falco sparverius)* was selected because it serves well as a conservative representative of several T&E bird species at the Laboratory, especially the peregrine falcon *(Flaco peregrinus)* and the Mexican spotted owl *(Strix occidentalis mexicanus)*. Furthermore, as an intermediate carnivore, it represents an organism with high susceptibility to contaminant biomagnification via terrestrial pathways.

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

Aquatic Receptors

Four aquatic receptors were selected for screening. Algae was 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, generic bony fish were selected to represent intermediate carnivores. There is no direct representative for the Jemez Mountain Salamander, an endangered species with both aquatic and terrestrial life stages. Juvenile salamanders are associated with water, while adults inhabit terrestrial environments. Adult Jemez Mountain Salamanders are invertebrate consumers, and can be considered functionally similar to shrews. We assume that juvenile salamanders or other amphibians are represented by the aquatic herbivore and omnivore receptors described above.

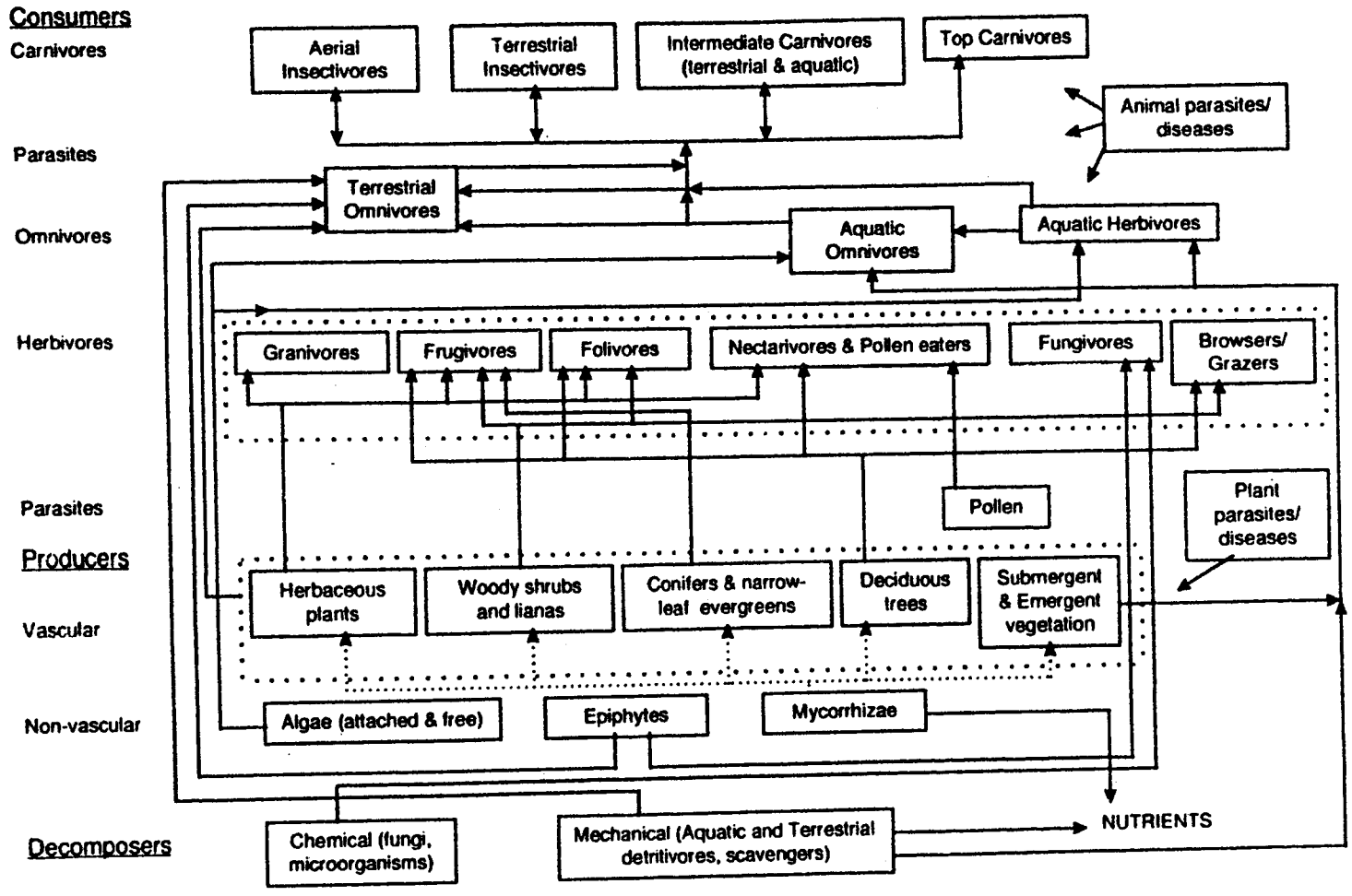

Figure 3-6. Functional food web diagram

Table 3-2. List of receptor species selected for screening at the Laboratory

Receptor Species	Receptor Category	Selection Factors
Generic plant	Terrestrial autotroph (producer)	Food source for many animals Provides habitat structure and functional base for terrestrial animals Represents culturally important plants Representative of T&E plant species Toxicity data is available Representative of all terrestrial angiosperm and gymnosperm plant species.
Earthworm	Soil-dwelling invertebrate	Represents decomposer group, which are important for nutrient cycling Large body of toxicity data Direct exposure to contaminated soil and detritus Represents a food source Representative of all soil-dwelling invertebrates
Desert cottontail	Mammalian herbivore	Food source for carnivores Ubiquitous and abundant Exposure data and toxicity data available Surrogate for economically important browsers (deer and elk)
Deer mouse	Mammalian omnivore	Food source for carnivores Ubiquitous and abundant Exposure data and toxicity data available Surrogate for T&E (Meadow Jumping Mouse)
Vagrant shrew	Mammalian insectivore	Food source for carnivores High fraction of soil in diet relative to rabbit and deer mouse Diet is 100% invertebrates and thereby maximizes this exposure pathway Surrogate for T&E (Jemez Mountain Salamander)
American robin	Avian omnivore	Food source for some carnivores Ubiquitous and abundant Exposure data available High fraction of soil in diet
American kestrel	Intermediate Carnivore/ Top Carnivore	Surrogate for peregrine falcon and Mexican spotted owl by assuming 100% flesh diet Ubiquitous Exposure data available Addresses potential biomagnification in avian food chain Conservative choice for this category, given the food intake to body weight ratio (see Section 4.2)
Red fox	Top carnivore	Exposure data available Addresses potential biomagnification in mammalian food chain Conservative choice for this category, given the food intake to body weight ratio (see Section 4.2)
Algae	Aquatic autotroph (producer)	Food source for animals Provides structure (substrate) for animals Ubiquitous and abundant Exposure and toxicity data available
Daphnids	Aquatic omnivore/herbivore	Food source for carnivores High exposure to contaminated water and sediment Ubiquitous and abundant Exposure and toxicity data available *Daphnia* and *Cerodaphnia* are typically the most sensitive aquatic organisms for a variety of contaminants
Aquatic snails	Aquatic herbivore (grazer)	Food source for some carnivores (e.g. fish) High exposure to contaminated sediment Ubiquitous and abundant Exposure and toxicity data available
Fish	Intermediate carnivore	Representative of potential waterborne contaminant effects in the Rio Grande High potential exposure to contaminants; potentially sensitive to persistent bioaccumulators and biomagnifiers.

3.6 Assessment Endpoints

Superfund guidance states that for the screening-level assessment, assessment endpoints are any adverse effects on ecological receptors, where receptors are populations and communities, habitats, and sensitive environments (EPA 1997). Following the Superfund guidance, the Laboratory's assessment endpoints are adverse effects on receptor populations, and adverse effects on these populations can be inferred from endpoints related to impaired reproduction, growth, and survival (EPA 1997). These endpoints will be considered in the identification and evaluation of appropriate toxicity information and in the development of ecotoxicity screening reference values (ESRVs).

4.0 Site-Specific Screening Level Ecological Risk Assessment

This section describes the three steps of the screening-level ecological risk assessment: (1) the scoping evaluation (or problem formulation phase described in Section 4.1), (2) the screening evaluation (or the screening-level risk and uncertainty analysis phase described in Sections 4.2 and 4.3), and (3) risk interpretation (or screening-level risk characterization described in Section 4.4).

4.1 Scoping Evaluation

The goals of the scoping evaluation are to identify those sites that need a screening evaluation, assess the need for an aggregate assessment, identify COPCs, determine data adequacy for screening, develop the EESCM, and evaluate bioaccumulation concerns. The scoping evaluation is equivalent to the site-specific problem formulation step.

4.1.1 Administrative NFA

The first step of the scoping evaluation is to determine if the PRS is a candidate for an administrative NFA based on the following NMED criteria:

- NFA criterion 1 (site does not exist),
- NFA criterion 2 (site never used for solid waste or hazardous wastes)
- NFA criterion 3 (documentation of no release through an evaluation of process knowledge)

The ER Project personnel provide the justification for these NFA recommendations. Environmental sample information is not required, and further ecological evaluations are unnecessary. If the site is not an administrative NFA, an RFI is conducted and data are collected to determine if the site poses a potential threat to human health or the environment. The site visit and scoping checklist described in Section 4.1.3 can be used to guide the data collection process.

4.1.2 Data Assessment

After the RFI (or equivalent investigation), the data are assessed (documented in the RFI report) to determine if there are COPCs at the site. The COPCs are identified by comparing the maximum constituent concentrations to AA-approved levels, including:

- background for inorganic constituents, fallout for radionuclide concentrations, or MDLs, PQLs, or EQLs for organic constituents (Box 2, criterion 3 of the NMED/HRMB Risk-Based Decision Tree, Appendix A); and/or,
- standards or other approved values (Box 2, criterion 4 of the NMED/HRMB Risk-Based Decision Tree, Appendix A).

AA-approved standards exist for surface water in the form of WQCC wildlife and livestock watering standards. There are no AA-approved soil or sediment standards at this time. If there are no COPCs (none of the maximum constituent values exceed AA-approved levels), then the PRS may be recommended for NFA (Boxes 3 and 4 of the NMED/HRMB Risk-Based Decision Tree, Appendix A). The ER project personnel provide the justification for these recommendations in the RFI report and further ecological evaluations are unnecessary.

Those PRSs at which COPCs are present require further ecological scoping, including completion of the scoping checklist, which requires a site visit by a member of the ecological risk assessment task team.

4.1.3 Scoping Checklist

The purpose of the scoping checklist is to provide information to

1. confirm that ecological receptors can be affected by a release;
2. determine if the PRS should be combined with other PRSs for screening and establish the functional/operational boundaries of the assessment;
3. determine if adequate quality and quantity of data exist for the screening evaluation, primarily as related to nature, rate, and extent of contamination;
4. prepare for HQ/HI analysis by determining whether screening should encompass terrestrial and/or aquatic receptors; and
5. gather information to develop the EESCM (e.g., what are the dominant/important transport pathways, exposure routes, and receptors).

Completion of the 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 (1) and collect field notes to help complete the development of the site conceptual screening model. The site visit can be used to document the presence or lack of receptors and off-site migration pathways. Notes are also made regarding the applicability of existing data for determining the nature, rate and extent of contamination. Specific attention is paid to the likelihood that the sample maximum represents the highest contaminant concentrations.
3. Completing the EESCM diagram to identify the complete and incomplete exposure pathways.

4.1.3.1 Checklist Step 1: Assemble Existing Information

In order to prepare for the site visit, the following information should be obtained: (1) the most current biological assessment information for the PRS (typically the Biological and Floodplain Assessment for applicable operable unit [OU] and/or TA); (2) AP 4.5 Parts A,B; (3) RFI work plan or report, as applicable, that provides contamination source, sample locations, analytical suites, and sample results; and (4) Facility for Information Management, Analysis, and Display (FIMAD) geographical information system (GIS) maps that show (if applicable) neighboring PRSs, sample locations, vegetation types, watershed name, and wetlands.

In most cases a meeting will be needed before the site visit to discuss the existing information for the PRS through a structured review of PRS history and status. The results of the meeting (or equivalent) will be documented in Part A of the Scoping Checklist (Appendix B). The information required for Part A of the Scoping Checklist includes: (1) site identification; (2) nature of PRS releases (solid, liquid, gaseous, or other); (3) a list of the primary impacted media (soil, water/sediment, subsurface [greater than 3 ft depth], or other); (4) specification of the applicable FIMAD vegetation classes (water, bare ground, spruce/fir/aspen/mixed conifer, ponderosa pine, piñon juniper/juniper woodland, grassland/shrubland, and developed. [Note that the FIMAD vegetation classes do not match 1:1 the cover types listed in Table 3-1 and described in Section 3.1.5]); (5) identification of T&E habitat, if present (list species if applicable); (6) a list and description of neighboring/contiguous/upgradient PRSs (discuss whether it is necessary to aggregate PRSs for screening); (7) AP 4.5 Part B information (runoff score and the terminal point of surface water transport); and (8) documentation of other scoping meeting notes (as appropriate).

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

4.1.3.2 Checklist Step 2: Site Visit

The main objective of the site visit is to affirm whether or not ecological receptors can interact with site releases. A secondary objective is to evaluate whether site data provide information to determine the nature, rate, and extent of contamination. The site visit should be arranged at an appropriate time of year (ideally spring or summer) to best evaluate biological resources at the site. If the site visit is planned for another time of year, any uncertainties introduced in the initial biological assessment by such timing must be noted.

The following resources are typically needed for the site visit: (1) maps showing sample locations and results, (2) a camera, (3) a measuring device to roughly locate relevant biological features (measuring tape and/or rangefinder), and (4) pin flags or other markers to specify locations for surveying.

Part B of the checklist is to be completed during the site visit, and includes: (1) site identification, (2) date of site visit, (3) personnel conducting visit, (4) receptor information (primarily aimed at determining if ecological receptors are present at the site), (5) contaminant transport information (emphasizing surface water transport, but also noting if there are other modes of transport), and (6) ecological effect information (notes on physical disturbance and obvious ecological effects [such as dead vegetation or lack of fossorial faunal activity]).

If there are no receptors and no offsite transport pathways, the remainder of the checklist (Part C) should not be completed. The checklist will be stopped at this point and any additional explanation/justification will be provided for proposing that the site poses no threat to the environment.

If there are receptors and pathways, then subsequent questions involving data adequacy will be addressed. Specifically, do existing data provide information on the nature, rate, and extent of contamination? Also, do existing data for the PRS address potential pathways of site contamination and receptor exposure? Completion of Part B includes additional field notes on the site setting and potential ecological receptors.

4.1.3.3 Checklist Step 3: Ecological Pathways Conceptual Exposure Model

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

4.1.3.4 Bioaccumulator/Transport Evaluation

If potentially persistent bioaccumulators or biomagnifiers are identified in Part C of the Scoping Checklist, then an evaluation is needed to determine if the site has fate and transport mechanisms and source terms such that persistent bioaccumulation and/or biomagnification are of concern. If so, further screening characterization is not necessary since the NMED guidance suggests that the PRS should proceed directly to a risk management decision to evaluate corrective actions (NMED 1998).

4.2 Screening Evaluation/Initial Identification of COPECs

This section describes the methods for calculating an HQ and an HI, which are used to identify COPECs for potentially affected receptors. This step is equivalent to the screening-level risk analysis phase.

4.2.1 Hazard Quotient and Hazard Index Calculations

This section presents the methods used to calculate an HQ and an HI for screening assessments of nonradiological and radiological substances. The HQ calculation adopted for aquatic and terrestrial screening receptors is a ratio of a dose exposure (presumed dose of a contaminant to a receptor) to an ecotoxicity screening reference value (ESRV). For ecological risk screening, the ESRV is the no observed adverse effect level (NOAEL)[1]. The U.S. EPA defines the NOAEL as the "highest level of a stressor evaluated in a toxicity test or biological field survey that causes no statistically significant difference in effect compared with controls or a reference site." (EPA 1997). Effects on organisms may be measured as reproductive, or measures of morbidity and mortality. The HQ calculation takes the form of Equation 4.1, below (EPA 1997).

$$HQ = \frac{dose\ exposure}{NOAEL} = \frac{function\left(receptor\ ,site\ media\ concentration\right)}{NOAEL} \qquad \text{Equation 4.1}$$

The numerator of Equation 4.1 is a variable, dependent on site-specific and receptor-specific information. The units of the NOAEL or ESRV are milligrams of a contaminant per kilogram of receptor body weight per day for any wildlife screening receptor, with the exception of plants and invertebrates, for which the units are milligrams of contaminant per unit mass of media (e.g., kilograms of soil). The denominator of Equation 4.1 is regarded as a constant value for a particular receptor and is expressed in the same units as the numerator; the HQ is, therefore, unitless. The wildlife receptor dose is dependent on the intake (consumption) of the contaminant from dietary and nondietary sources (e.g., soil). In all cases, the wildlife contaminant intake is assumed to be proportional to the contaminated media concentration. This fact allows for an alternative calculation of the wildlife HQ, which is discussed below.

The HQ can be calculated from the ratio of an observed media concentration to the media-specific and receptor-specific concentration limit, referred to in this document as the ecological screening level (ESL). The term ESL is also used by NMED (Box 11 criterion 1.e of the NMED/HRMB Risk-Based Decision Tree, Appendix A). This method of calculation is advantageous because ESL values may be calculated for any given receptor, provided receptor-specific information (e.g. body weight, rates of media consumption, etc.) and toxicity information (e.g., NOAEL or LOAEL) are available. The ESL is derived from a back-calculation of Equation 4.1, where the HQ is set equal to one. Thus, the ESL for a given contaminant is the contaminant concentration in a particular medium that confers calculation of an HQ of 1 for a given receptor. This latter relationship is clearly delineated in Equation 4.2, below.

$$HQ = \frac{Site\ Media\ Concentration}{ESL} \qquad \text{Equation 4.2}$$

In cases where multiple media are contaminated at a PRS, e.g., soil and water, the appropriate adjustments must be made to account for exposure to multiple media for the same receptor. The information needed to make the back-calculations to derive receptor-specific and single media-specific ESLs is provided in the following sections.

The HQ may assume any value from 0 to infinity. Since the HQ is a ratio that may exceed 1, by definition the HQ cannot be a probability and cannot be equated to risk. However, the HQ is an index that can be viewed as an indicator of risk (Bartell 1996, EPA 1997). Recall that the NMED guidance requires that an HQ equal to 1 be used as an indicator of risk for a particular chemical or radionuclide. If the HQ is greater than 1, the COPC is identified as a COPEC.

[1] NMED guidance (Box 11 criterion 1.c of the NMED/HRMB Risk-Based Decision Tree, Appendix A) states that "in the absence of a literature NOAEL, the NOAEL can be estimated by applying an uncertainty/safety factor of 10 for the lowest available lowest-observed adverse effect level (LOAEL) or of 100 for the lowest available acute toxicity value (LD50 or LC50) or effective concentration (EC50).

Hazard indices are calculated as the sum of all HQs at a given site for a given receptor, with common toxicity endpoints (i.e., for HI calculations, radiological effects are summed separately from nonradiological effects). The HI can be thought of as a summary index that implies there may be risk to a particular organism from a combination of environmental contaminants with common toxicity endpoints. The HI is specific to the type of exposure to which wildlife may be susceptible; for example, distinction is made between terrestrial and aquatic receptors.

For screening-level assessment, the calculation of HQs and HIs are used directly to indicate whether the chemical constituents at a site pose a potential harm to the biota. As stated above, individual constituents measured at, or in association with a source term, and scoring an HQ greater than or equal to 1 for target organisms, are to be carried forward from a screening assessment level to subsequent levels of consideration in assessing ecological risk. These constituents are consequently labeled COPECs. In addition, those chemicals that contribute more than 0.1 to an HI that exceeds 1 are considered COPECs.

4.2.2 ESRVs for Nonradiological and Radiological Contaminants

This methodology adopts a NOAEL (or an appropriate estimate) as an ESRV for screening-level ecological risk assessment. ESRVs are cut-points for considering toxic dosages for chemical constituents that may confer harm to a given ecological receptor. ESRVs must be experimentally derived and based upon determination of the toxicological kinetics for specific organisms under experimental conditions of uptake. Terrestrial and aquatic ESRVs for nonradiological constituents will be based on investigation of primary literature, experimental resources, and other NMED-approved resources. Chemical-specific toxicological information and the determination of ESRVs will be deferred to supporting documentation at a later date rather than presented in this methodology. The nature of the information to be reported is provided in Table 4-1.

Table 4-1. Categories of information to be supplied to support ESRVs for screening receptors

chemical
chemical form
test organism
NOAELs (mg/kg/day) in literature
endpoint exposure length
exposure route
dosage
study notes
calculation
test species identification and body weight
test species water consumption rate
test species food consumption rate
reference (NOAEL)
NOAEL chosen for use
reason for choosing NOAEL

ESRVs for radiological constituents are 0.1 rad per day for all terrestrial and aquatic receptors (IAEA 1992).

4.2.3 ESL Calculations for Nonradiological Constituents

4.2.3.1 Terrestrial Receptors

The ESLs for terrestrial receptors are determined differently for plants, invertebrates (earthworms), and vertebrates. For plants, earthworms and other soil-dwelling invertebrates, dose is measured as the concentration of a chemical constituent in soil; therefore ESL values directly determine the critical dose at which $HQ=1$ is conferred. Dose to terrestrial vertebrates, however, is dependent on the transfer of a chemical constituent from a given medium (such as soil or foodstuff) to the organism through direct and indirect means (i.e., via ingestion, inhalation, and dermal exposure pathways). Ingestion is typically considered the major pathway for terrestrial organisms; consequently, it serves here as the sole model for terrestrial dose exposure calculation (EPA 1993). For vertebrate receptors, therefore, ESL values must be based on the dietary regimen of the receptor, including consumption of plants, invertebrates, vertebrate flesh, and drinking water, with some incidental soil ingestion.

Dose models for plant and invertebrate receptors are presented in the next two sections. The terrestrial vertebrate dose exposure model is presented following the discussion on plant and invertebrate bioconcentration and constituent transfers. The mathematical model for dose exposure to a terrestrial vertebrate receptor is based on the *Wildlife Exposure Factors Handbook* (EPA 1993), and is presented below.

Plant Bioconcentration and Constituent Transfer

The receptor for the plant model is considered generic for the purpose of the screening assessment. Plant metabolic assimilation (uptake) of inorganic and organic substances is characterized by soil-to-plant bioconcentration factors, also know as BCFs, as defined in Section 2.0 of this document. Bioconcentration of chemical constituents into plant tissues is simply the product of a BCF and the concentration of a constituent in soil, often representing the total measure of the constituent in all mineralogical and elemental forms in a given medium (e.g., soil for terrestrial plants), regardless of bioavailability. The simple model of plant bioconcentration of inorganic and organic substances is given below.

$$C_p = C_{soil} \cdot BCF_p \qquad \text{Equation 4.3}$$

where
C_p is the concentration of a constituent in plant tissue,
C_{soil} is the concentration of the constituent in soil, and
BCF_p is the bioconcentration factor for plants.

For inorganic constituents, BCFs are calculated as the ratio of the concentration of a constituent in the tissue of an organism (either homogenous or tissue-specific) to the concentration in the specific media. For plants, the media for calculating the BCF for inorganics is soil that has been dosed with known quantities of a given inorganic constituent. Studies providing the metabolic assimilation (uptake) and transfer of inorganic constituents from soil to plant are taxon-specific; however, the dose exposure model used applies generically across taxa. The BCFs for inorganic substances are taken from Baes et al. (1984).

For organic chemicals, calculation of the BCF_p is dependent on a regression relationship developed by Travis and Arms (1988) and presented in Equation 4.4 below. The variable *Kow* (Equation 4.4) is the octanol-water partition coefficient. The octanol-water partition coefficient *(Kow)* is a ratio of the solubility of a chemical constituent in octanol (an eight carbon alcohol) to its solubility in water;

$$\log BCF_p = 1.588 - 0.578 \cdot \log Kow \qquad \text{Equation 4.4}$$

The *Kow* is a measure of an organic chemical's miscibility in octanol versus water (a ratio of the two). Thus, this ratio can be thought of as a chemical's relative hydrophobicity or (conversely) the affinity of one organic compound for another. Equation 4.4 is a standard regression relationship derived empirically from regressing an experimentally measured *BCF* of an organic constituent on the *Kow* for that constituent. The higher an organic chemical's *Kow* value, the greater its affinity for organic materials in soil and the less available it is for plant uptake (thus, the negative slope value for Equation 4.4). Values for *Kow* will be taken from Mackay et al. (1992), and other NMED-approved sources.

It is important to recognize that partition coefficients, such as the *Kow*, are in practice based on simple diffusion-equilibrium models and experimentation. For the plant BCF (Equation 4.4) this becomes important, as mentioned above, because the uptake of the organic chemical constituent is determined solely from the interstitial water fraction of the soil. Conceptually speaking, therefore, any of the organic constituent that is adsorbed to inorganic and organic soil particulates is unavailable for plant uptake because it is not in the water fraction (interstitial water) of soil. This recognition makes it more difficult to estimate plant uptake because it is likely that the overall concentration of a constituent in soil is not representative of that which is available to plants.

Soil-Dwelling Invertebrate Bioconcentration and Constituent Transfer

Calculation of bioconcentration in soil-dwelling invertebrates is similar to that for plants. Models are formulated appropriately for organisms that live out their lives intimately associated with soil, obtaining at least some of their nutrients (including water and gases) through their integument. This grouping of organisms might include earthworms, terrestrial gastropods, nematodes, and some soil-dwelling insects, arachnids, and crustaceans. For the majority of invertebrates, which are largely herbivores and carnivores with little or no intimate contact with soil, this invertebrate dose exposure model is inappropriate.

The model of invertebrate uptake of inorganic and organic substances is presented in Equation 4.5, below.

$$C_{inv} = C_{soil} \cdot BCF_{inv} \qquad \text{Equation 4.5}$$

For inorganic substances, invertebrate BCFs are derived from the ratio of the concentration of a substance found in an invertebrate (usually an earthworm) to the concentration in soil. Invertebrate BCFs for inorganic substances are found in various literature sources and will be adopted by the Laboratory based on investigation of primary literature, experimental resources, and other NMED-approved resources. The default value of $BCF_{inv} = 1.0$ will be used when no other information is available for a given constituent. A BCF_{inv} of 1 means the concentration of an inorganic constituent in soil is equal to the concentration within a soil-dwelling invertebrate.

The organic constituent BCF model for soil-dwelling invertebrates (BCF_{inv}) was adopted from Connell and Markwell's (1990) interpretation of earthworm bioconcentration studies, and is presented in Equation 4.6 below.

$$BCF_{inv} = \frac{L \cdot Kow^{y}}{c \cdot f_{oc}} \qquad \text{Equation 4.6}$$

In Equation 4.6, L is the lipid fraction of the organism, c is a proportionality constant set equal to 0.66 (following Connell and Markwell 1990), f_{oc} is the fraction of organic matter in soil, *Kow* is the octanol-water partition coefficient (described above), and y is a nonlinearity constant set equal to 0.05 (Lord et al. 1980).

Equation 4.6 is based on a diffusion-equilibrium process for passive "soil water-to-soil organic matter" and "soil water-to-earthworm" diffusion of an organic substance. It is important to note that the model does not infer active metabolic processes that may influence the uptake of organic constituents by earthworms from the organic fraction of the soil (e.g. by means of ingestion of organic matter). Given these model tenets, the concentration of the organic constituent in soil water can be described by the following regression relationships:

$$C_w = \frac{C_{om}}{c \cdot f_{oc} \cdot Kow^a} = \frac{C_{worm}}{L \cdot Kow^b}, \qquad \text{Equation 4.7}$$

where C_w is the concentration of the constituent in soil water when in equilibrium with the concentration of the same in the organic fraction of soil (C_{om}) and (passively) the worm (C_{worm}). Other variables and constants of Equation 4.7 are identical to that of Equation 4.6, with the exception of a and b. These latter variables (treated as the single constant $y=b-a$ in Equation 4.6) can be thought of as the relative affinity that an organic constituent has for soil organic matter and worm lipids, respectively. Worm lipids are generally considered more affinitive of organic compounds than are soil-borne organic constituents. By solving Equation 4.7 for C_{worm}, one basically obtains Equation 4.6, where $y=b-a$, with one exception: for Equation 4.6, C_{soil} is substituted for C_{om}. Clearly this latter substitution makes Equation 4.6 conservative from the standpoint that there is likely far more of an organic constituent in soil than might be available to an earthworm. However, Equation 4.6 does not include the direct ingestion of contaminated organic matter, which introduces a negative bias into the calculation of earthworm contaminant body burden.

Terrestrial Consumer Dose Exposure and Constituent Transfer

The general vertebrate dose exposure model is used to calculate the dose exposure of inorganic and organic constituents in the environment to terrestrial vertebrate herbivores, omnivores, and carnivores. The model is reliant on the simple concepts that consumers' diets are rather simply comprised of known or assumed dietary proportions, and that contaminants are passed to the organism through dietary media, incidental soil ingestion and contaminated water ingestion (where appropriate).

The dose exposure model used for vertebrates is adopted from the *Wildlife Exposure Factors Handbook* (EPA 1993, Chapter 4), and is provided below in Equation 4.8:

$$D_x = C_{soil} \cdot I_{soil} \cdot F_{soil} + C_{water} \cdot I_{water} \cdot F_{water} + C_{soil} \cdot I_{tot} \sum_{i=1}^{m} BAF_i \cdot F_i \cdot \kappa_i \qquad \text{Equation 4.8}$$

Where:

- D_x is the estimated daily dose from chemical constituent x (mg/kg/day),
- C_{water} is the concentration of chemical constituent x in water (mg/L)
- I_{water} is the normalized daily water ingestion rate (g of water / [g of body weight • day])
- F_{water} is the fraction water ingested from a contaminated area
- C_{soil} is the concentration of chemical constituent x in soil (mg/kg dry weight)
- I_{soil} is the normalized daily soil ingestion rate (g of soil / [g of body weight • day])
- F_{soil} is the fraction of incidental soil ingested from a contaminated area
- I_{tot} is the normalized total daily dietary ingestion rate (g of food [dry weight]/[g of body weight • day])
- BAF_i is the bioaccumulation factor for chemical constituent x in soil to diet item i
- F_i is the fraction diet constituted by item i, derived from a contaminated area
- κ_i is the proportion of the organism's diet composed of item i
- i is the dietary item (choices include: plants, soil invertebrates, and flesh)
- m is the number of diet items

This model provides an estimate of the dose associated with a concentration of an inorganic or organic chemical toxicant in soil, given an organism's normalized daily ingestion rate. In this model, incidental ingestion of soil and ingestion of contaminated water are considered. Soil ingestion is calculated from a fraction of the dietary intake that is soil (see EPA 1993, Chapter 4).

The above model requires that all measures of ingestion are in dry weight. Because EPA (1993) presents normalized food ingestion rates on a wet weight basis, these dietary constituents must undergo wet-to-dry weight conversions. Metrics required for these conversions and other elements of the model (with the exception of bioaccumulation

factors) are provided for terrestrial vertebrate receptors in Table 4-2, below. Note that the information provided in Table 4-2 is for the screening receptors adopted by the Laboratory.

Table 4-2. Measures required for the elements of Equation 4.10 (except bioaccumulation factors), the vertebrate dose exposure model

Species	Parameter	Value	Units	Reference (page)	Notes
American kestrel	body weight	103	g	EPA (1993) p 2-112	smallest male was 103 g
	food intake[a]	0.31	g/g/day	EPA (1993) p 2-112	higher of 2 values (assumed weight was 119 g)
	food moisture content	0.68	proportional	EPA (1993) p 4-13	diet includes insects, birds, mammals, other (see p 2-113) [value assumes mammals, birds]
	water intake	0.12	g/g/day	EPA (1993) p 2-112	higher of 2 values
	inhalation rate	0.089	m^3/day	EPA (1993) p 2-113	higher of 2 values
	fraction soil in diet	0.02	unitless	none	default value
	soil invertebrate diet[b]	0.5 (0)	unitless	EPA (1993) p 2-113	rounded EPA value to 50%
	flesh diet[b]	0.5 (1)	unitless	EPA (1993) p 2-113	rounded EPA value to 50%
American robin	body weight	77	g	EPA (1993) p 2-197	smallest weight was 77 g
	food intake[a]	1.52	g/g/day	EPA (1993) p 2-197	higher of 2 values, weight was 55 g
	food moisture content	0.69	proportional	EPA (1993) p 4-13	diet includes: invert, plants (fruits), assumed grasshoppers
	water intake	0.14	g/g/day	EPA (1993) p 2-197	estimated
	inhalation rate	n/a	m^3/day	n/a	n/a
	fraction soil in diet	0.1	unitless	EPA (1993) p 4-20	used Woodcock value
	soil invertebrate diet	1	unitless	none	assumed strict insectivore diet
deer mouse	body weight	20	g	EPA (1993) p 2-295	for females
	food intake[a]	0.22	g/g/day	EPA (1993) p 2-296	diet of lab chow, 8-10% H2O
	food moisture content	0.1	proportional		see note on line above
	water intake	0.19	g/g/day	EPA (1993) p 2-296	adult male or female
	inhalation rate	0.025	m^3/day	EPA (1993) p 2-296	higher of 2 values, estimated
	fraction soil in diet	0.02	unitless	EPA (1993) p 4-20	for white-footed mouse
	plant diet	0.5	unitless	EPA (1993) p 2-297	rounded EPA value to 50%
	soil invertebrate diet	0.5	unitless	EPA (1993) p 2-297	rounded EPA value to 50%

Table 4-2 (continued). Measures required for the elements of Equation 4.10 (except bioaccumulation factors), the vertebrate dose exposure model

Species	Parameter	Value	Units	Reference (page)	Notes
eastern cottontail for desert cottontail	body weight	800	g	EPA (1993) p 2-355	Lower 95th percentile of mean weight of males. Chosen based on reported body weight of smaller desert cottontail
	food intake[a]	0.24	g/g/day	Nagy(1987)	Estimated as 95% upper CI using Nagy(1987)
	food moisture content	0.85	proportional	EPA (1993) p 4-14	Assume dicotyledonous leaves
	water intake	0.097	g/g/day	EPA (1993) p 2-356	estimated
	inhalation rate	0.63	m^3/day	EPA (1993) p 2-356	estimated
	fraction soil in diet	0.024	unitless	EPA (1993) p 4-20	for meadow vole
	plant diet	1	unitless	EPA (1993) p 2-356	strict herbivore diet
short-tailed shrew for vagrant shrew	body weight	15	g	EPA (1993) p 2-213	smallest weight was 15 g
	food intake[a]	0.62	g/g/day	EPA (1993) p 2-213	higher of 3 intakes, weight was 21 g
	food moisture content	0.84	proportional	EPA (1993) p 4-13	assume earthworms
	water intake	0.223	g/g/day	EPA (1993) p 2-213	one value reported
	inhalation rate	0.026	m^3/day	EPA (1993) p 2-213	one value reported
	fraction soil in diet	0.1	unitless	EPA (1993) p 4-20	used woodcock
	soil invertebrate diet	1	unitless	EPA (1993) p 2-213	strict insectivore diet
red fox for gray fox	body weight	3 940	g	EPA (1993) p 2-224	lowest of 4 values
	food intake[a]	0.14	g/g/day	EPA (1993) p 2-224	female after whelping
	food moisture content	0.68	proportional	EPA (1993) p 4-13	mostly mammals, some birds [assume mammals]
	water intake	0.086	g/g/day	EPA (1993) p 2-224	higher of 2 values, estimated
	inhalation rate	2	m^3/day	EPA (1993) p 2-224	higher of 2 values, estimated
	fraction soil in diet	0.03	unitless	EPA (1993) p 4-20	for red fox
	flesh diet	1	unitless	EPA (1993) p 2-224	rounded diet to 100% flesh

[a] Normalized ingestion rates are presented in units of g of food (wet weight)/[g of body weight • day]
[b] There are two variants on the American kestrel, one more realistically models its actual diet (half insect and half flesh), and the strict flesh-eater is used to mimic the diet of the Mexican spotted owl or peregrine falcon.
n/a = not available

For the screening assessment, the fraction of the organism's diet constituted by item *i*, derived from a contaminated area is simply set to 1 for the most conservative calculation (this assumption is further considered in the uncertainty analysis). Likewise the fraction of soil ingested form the contaminated site is also set to 1 in the screening assessment. Where contaminated water is available for wildlife, animals are assumed to drink from the most contaminated water source.

For herbivores and strict flesh-eating carnivores, the fraction of the relevant diet item is equal to 100%. For omnivores, the diet is evenly divided between plant and animal (either soil-dwelling invertebrate, vertebrate, or both) portions, and for carnivores whose diet is partially invertebrate and partially vertebrate, the diet is evenly divided between invertebrate and vertebrate portions.

The bioaccumulation factors in the model (Equation 4.8) represent the ratio between the concentration of a contaminant in a diet item and the concentration in soil. For plants as a diet item, this value is provided by the BCF_p used in Equation 4.3. For soil-dwelling invertebrates as a diet item, this value is provided by the BCF_{inv} used in Equation 4.5. For the flesh diet item, the bioaccumulation factor is typically represented as a product of the bioconcentration from soil to food for prey item (BCF_p and/or BCF_{inv}) and bioaccumulation into prey muscle tissue (BAF_{fm}). The BAF_{fm} (bioaccumulation from food-to-muscle) is defined as "a chemical's concentration in an organism or tissue divided by its concentration in food (for terrestrial organisms)," (Travis and Arms 1988). BAF_{fm} values for inorganic substances are derived from Baes et al. (1984), and other NMED-approved resources.

For organic chemicals, BAF_{fm} values will be based on a regression relationship of log*Kow* values, as developed by Travis and Arms (1988). The equation presented below, is based upon conversion of Travis and Arms' (1988) Equation(1) from "biotransfer factor" to "bioconcentration factor."

$$logBAF_{fm} = -6.832 + 1.033 \cdot logKow, \qquad \text{Equation 4.9}$$

where:

BAF_{fm} is the food-to-muscle bioaccumulation factor,
Kow is the octanol-water partition coefficient, and

Parenthetically, Travis and Arms (1988) incorrectly identify the *BAF* in Equation 4.9 as the *BCF*. Because trophic transfer is explicit in Equation 4.9, *BAF* is the correct term. Equation 4.9 was developed on the basis of the concentration of an organic constituent found in beef muscle in *wet weight* units. Thus, the food-to-muscle bioaccumulation factors must be scaled to dry weight units before they are used. This scaling requires receptor-specific knowledge of average moisture contents (see Table 4-2). Thus, the receptor-specific form of the food-to-muscle bioaccumulation factor is presented in Equation 4.10.

$$BAF_{dw} = \frac{10^{-6.832+1.033 \cdot logKow}}{1 - MC_{food}} \qquad \text{Equation 4.10}$$

where:

BAF_{dw} is the food-to-muscle bioaccumulation factor in dry weight units,
Kow is the octanol-water partition coefficient, and
MC_{food} is the moisture content of the food.

ESL Calculation for Terrestrial Vertebrate Receptors

The ESL refers to an organism's exposure-response threshold for a given chemical constituent. As mentioned above, the ESL is considered the concentration of a substance in a particular medium that confers calculation of an HQ of 1 for a given organism. The ESL, therefore, is useful in the direct calculation of HQs and HIs for the screening assessment analyses. The ESL for a chemical constituent's concentration in soil (mg/kg) is simply calculated by setting the HQ equal to 1 and solving for the soil concentration (C_{soil}) of an organism's bioaccumulation or dose exposure model (as appropriate). These models are Equation 4.3 for plants, Equation 4.5 for invertebrates, and Equation 4.8 for terrestrial consumers. For plants and invertebrates, the ESL simply corresponds to the ESRV (NOAEL). The following equation shows the calculation of the ESL for terrestrial consumers, under the assumption that there is no contaminated drinking water associated with the PRS.

$$ESL = NOAEL \Big/ \left[I_{soil} \cdot F_{soil} + I_{tot} \sum_{i=1}^{m} BAF_i \cdot F_i \cdot \kappa_i \right] \qquad \text{Equation 4.11}$$

Equation 4.11 implies that, the HQ can also be calculated as a quotient of the observed concentration of a chemical constituent in soil to the ESL. Therefore, a soil-borne chemical constituent with a concentration greater than that of the ESL may be considered a COPEC.

4.2.3.2 Aquatic Receptors

For aquatic receptors, ESL values will be determined by investigation of primary literature, experimental resources, and other NMED-approved resources (including U.S. EPA ambient water quality criteria). Aquatic ESL value selections will be deferred to the ESRV/ESL document to be provided at a later date.

4.2.4 ESL Calculations for Radiological Constituents

4.2.4.1 Dose limits (ESRVs)

The International Atomic Energy Agency (IAEA) concluded that doses protective of human health were protective of ecological resources, with certain exceptions (1992). The report from a Department of Energy (DOE) workshop convened to revisit this conclusion, states:

> Participants further agreed with the IAEA that protecting humans generally protects biota except when (1) human access is restricted but access by biota is not restricted, (2) unique exposure pathways exist, (3) rare or endangered species are present, or (4) other stresses are significant. To deal with these exceptions, site-specific exposures should be considered in developing secondary standards. The participants concluded that existing exposure models are sufficient in principle for developing secondary standards. However, transfer coefficients must be developed for some important species and exposure routes that have not been adequately studied, and improved (radiological dose) models for reference biota are needed to eliminate unnecessary conservatism and provide a practical approach to implementation of the standards. (ORNL 1995)

For the four special situations described above, IAEA (1992) recommends a dose limit of 0.1 rad per day. However, this limit is to be applied with judgment about the applicability of the limits to the situation being analyzed, particularly if threatened or endangered species are involved. These limits are consistent with the the results of reviews by NCRP (1991) and Eisler (1994).

4.2.4.2 Estimating Radiological Dose

The dose to biota is the sum of the dose from internally deposited radionuclides and the external dose from the same radionuclides in soils. The following discussion is divided into internal dose and external dose estimation methods. The methods presented provide an overestimate of the dose and are for screening purposes only. Obtaining a better estimate of the dose to an organism will require much more sophisticated models or measurements of the external radiation dose or the concentration of the radionuclides of interest in the biota of interest. The equations and parameters used in this model are similar to those published by Amiro (1997) and Baker and Soldat (1992).

Internal Dose to Biota

The dose to biota from radioactive materials ingested or inhaled and deposited internally is dependent on several factors. The primary factors are the type of radiation, the biochemistry of the radionuclide, the organ in which the radionuclide may deposit preferentially, and the complexity of the food chain of the organism of interest. Each of these factors influences the dose absorbed by the animal or plant. Preparing parameters for screening of environmental concentrations in food sources and food chains requires overestimating parameters enough to minimize the possibility of screening out concentrations that may lead to an effect. Such overestimation, however, must not be so large as to make the screening useless and misleading.

The following discussion is divided into separate stages of analysis. The first stage deals with the energy deposited in tissue by the different types of radiation. The second deals with the transport of the radioactive material through the

environment to a receptor where the biochemistry and food chain are considered. The information is combined to estimate the absorbed dose for the receptor using an equilibrium model with corrections for radioactive decay and biological retention.

Energy Deposition by Radiation Types

The energy deposited in tissue is dependent on radiation type. For alpha particles, the discrete energy of the helium nucleus is absorbed by the tissues. For beta particles, the average energy deposited is calculated from a distribution of energies, which is dependent on the maximum energy of a particle. The assumption for both alpha and beta particles is that all the energy is deposited in the tissue. In the case of beta particles, this assumption can lead to an overestimate for high energy particles that have a range in tissue greater than the radius of the organ or organism. In the case of gamma and X-rays the energy absorbed is a function of the radius of the organ or organism and the energy of the photon, which is emitted at a discrete energy.

The radionuclides uranium, plutonium, americium, thorium, and radium have radioactive progeny. The amount of progeny formed is dependent on the half-life of the decay product. Equations have been derived to estimate the amount of progeny at any time and its contribution to the total energy absorption, ΣE, in tissue (ICRP 1959). For screening, the summation of energies for the decay chains of uranium, plutonium, americium, thorium, and radium isotopes will be used. This approach results in an overestimate. The energy absorption is dominated by the large number of alpha particle emitters in the chain. The lifetime of many of the biota of interest is short compared to the time for buildup of the progeny. For example, the dose from thorium and its progeny to organisms that live only one year is overestimated using this approach because the decay of thorium-232 to radium-228 has a half-life of 5.75 years.

Estimation of the energy deposited by beta particles starts with the estimation of the average energy of the distribution of electrons emitted during decay. A listing of decay parameters and average energy per disintegration is presented in International Commission on Radiological Protection (ICRP) Report No. 38 (ICRP 1983). The values that will be used for the calculations are listed in Table 4-3 as the MeV per disintegration.

Table 4-3. Average beta particle energies for major radionuclides

Radionuclide	Beta Maximum (MeV)	Fraction of Disintegrations	Average MeV per Disintegration
Cesium-137 decays to Barium-137m (electron emissions)	0.5116 1.1732 0.00367 0.0264 0.624 0.656 0.660	0.946 0.054 0.0761 0.008 0.0808 0.0146 0.0048	0.164 0.0229
Protactinium-234m	2.28	0.983	0.811
Protactinium-234	22 betas		0.224
Plutonium-241	0.021	~1.00	0.00524
Strontium-90	0.546	1.00	0.196
Thorium-234	0.076 0.095 0.096 0.1886	0.027 0.062 0.186 0.725	0.000526 0.00154 0.00464 0.0366
Tritium	0.018591	1.00	0.00568
Yttrium-90	2.284	1.00	0.935

The beta-emitting radionuclides of concern at the Laboratory are cesium-137, strontium-90, and tritium (Hydrogen-3). The decay product of strontium-90 is yttrium-90, which emits a higher-energy beta particle than strontium-90. Uranium is an alpha emitter, but its progeny include thorium-234, protactinium-234m, and protactinium-234, which are beta emitters. For radionuclides with multiple beta decay levels in a radionuclide, the energy per disintegration is calculated as the sum of the MeV per disintegration for that radionuclide. For these radionuclides (e.g., protactinium-234), the total average energy per decay (or disintegration) is listed rather than the total decay scheme.

Alpha particle emission is in discrete energies rather than over a distribution of energies as for beta particle emission. The amount of energy deposited in tissue is assumed to be total and the energy is deposited in a small volume (Table 4-4). As in the case of the beta emitters, the radioactive elements of uranium, plutonium, americium, thorium, and radium have decay products that are radioactive. Inclusion of the energy from the decay of progeny is taken into consideration for each chain in the calculation of the dose factor.

Gamma ray and x-ray emissions from radionuclides contribute to the dose from both internal deposition and from external radiation. The amount of energy deposited in the biota from internally deposited radionuclides is a function of the effective radius of the animal or plant and the energy of the photons emitted. While complex geometric models can be developed to represent the energy absorbed in an organism, the assumption of a sphere of a density of 1 g/cm is conservative, as it overestimates the actual energy absorption (ICRP 1959).

At the Laboratory, the gamma ray and x-ray emitters most commonly encountered are barium-137m formed by the decay of cesium-137, and the gamma rays and x-rays from the decay series of uranium, plutonium, americium, thorium, and radium. Table 4-5 lists the gamma and x-rays from the major radionuclides (Schleien, 1992).

Calculating Internal Dose Rate (rad/day) for Terrestrial Animals

The second step in calculating internal dose is to convert the energy deposited for radionuclides (Table 4-4), and Table 4-5) to dose resulting from food chain intake. The conversion of the units MeV/disintegration to g-rad/pCi-day is necessary because units for radioactivity in the food chain are measured in pCi/g. The total radioactivity intake by the organism per gram of body weight of a given material is in units of grams of dry food/grams of fresh body weight in one day. The amount of radioactivity reaching tissues is estimated from the amount of element that passes through the digestive system to the blood. The terrestrial animals equations are based on an equilibrium model, where the activity concentration reaches steady-state in a time dependent upon the rate of radiological decay and metabolic elimination of the element from the organism's body.

Table 4-4. Alpha particle energies for major radionuclides[a]

Radioisotope	Energy(MeV)	Fraction of Decay	Activity Abundance of Isotope [b]
Americium-241	5.486 5.443 5.388	0.84 0.13 0.016	
Plutonium-238	5.499 5.456	0.709 0.29	0.016 [c]
Plutonium-239/240			0.56 [c]
Plutonium-239	5.156 5.143 5.105	0.731 0.15 0.118	0.81 [c]
Plutonium-240	5.168 5.124	0.73 0.27	0.19 [c]
Plutonium-241	4.85 4.90	0.000003 0.00002	0.42
Radium-226	4.60 4.78	0.0555 0.944	
Thorium-232	3.83 3.95 4.01	0.002 0.23 0.768	
Uranium-234	4.72 4.77	0.274 0.723	0.497 [d]
Uranium-235	4.2-4.32 4.366 4.398 4.5-4.6	0.103 0.176 0.56 0.113	0.0225 [d]
Uranium-238	4.15 4.20	0.229 0.768	0.481 [d]

a) From Schleien 1992.

b) The activity abundance of americium-241 is dependent on the formation by decay of plutonium-241. If not measured, the activity can be estimated using the plutonium-241 content at a known time.

c) The activity abundance of the plutonium isotopes is based on a measured ratio for Pueblo and Los Alamos Canyons. (Ferenbaugh, et. al, 1994). The weapons grade makeup of plutonium-239/240 is 94 % plutonium-239 and 6 % plutonium-240 by weight (Wenzel and Gallegos, 1982). In this mixture of the two isotopes, the plutonium-239 is 0.81 of the activity and the plutonium-240 is 0.19 of the activity. The radiochemical analytical methods detect both isotopes, but cannot distinguish between the two. Results are reported as single number, usually indicated as plutonium-239/240 or plutonium-239+240.

d) The activity abundance of the uranium isotopes is based on the natural abundance. For depleted uranium the activity abundance is 0.084 as uranium-234, 0.0146 for uranium-235, and 0.904 for uranium-238.

Table 4-5 Gamma ray and x-ray emissions from the major radionuclides at Los Alamos

Radionuclide	Photon Energy (MeV)	Fraction of Disintegrations
Americium-241	0.0263	0.024
	0.0595	0.357
	0.099	0.0002
	0.103	0.0002
Barium-137m	0.00447	0.0104
	0.03182	0.0207
	0.03219	0.0322
	0.0364	0.0139
	0.66165	0.8998
Plutonium-238	0.0136	0.1157
	0.0553	0.000473
Plutonium-239	0.0136	0.0441
	0.1129	0.000476
Plutonium-240	0.0136	0.1101
	0.0543	0.000525
Radium-226	0.186	0.0328
Thorium-232	0.059	0.0019
	0.126	0.0004
Uranium-234	0.053	0.0012
	0.121	0.0004
Uranium-235	0.1438	0.105
	0.163	0.047
	0.1857	0.54
	0.205	0.047
Uranium-238	0.0496	0.0007

Calculation of the internal dose factor (g-rad/pCi-day) was performed as follows:

$$\mathrm{g \cdot rad}/\mathrm{pCi \cdot day} = \Sigma E\left[\frac{\mathrm{MeV}}{\mathrm{disintegration}}\right] \times 1.6 \cdot 10^{-6}\,\frac{\mathrm{ergs}}{\mathrm{MeV}} \times 1\,\frac{\mathrm{rad}}{100\,\mathrm{ergs/g}} \times 1\,\frac{\mathrm{disintegration}}{27.03\,\mathrm{pCi \cdot s}} \times 8.64 \cdot 10^{4}\,\frac{\mathrm{s}}{\mathrm{day}}$$

$$\mathrm{g \cdot rad}/\mathrm{pCi \cdot day} = \Sigma E\left[\frac{\mathrm{MeV}}{\mathrm{disintegration}}\right] \times \left(5.11 \cdot 10^{-5}\,\frac{\mathrm{disintegrations \cdot g \cdot rad}}{\mathrm{MeV \cdot pCi \cdot day}}\right) \qquad \text{Equation 4.12}$$

Table 4-6 is the summation of the energy deposition in tissues (ΣE, Equation 4.12) for the radionuclides encountered at the Laboratory and the absorbed dose factor in g-rad/pCi-day. Table 4-7 is a list of the fractions of the radionuclide reaching the blood, which is assumed in the screening to equal the fraction reaching a target organism's tissue.

Table 4-6. Summation of energy deposition in tissues

Radionuclide	ΣMeV Deposited	Internal Dose Factor (g-rad/pCi-day)
Americium-241	5.7	2.9x10E-4
Cesium-137 and Barium-137m	0.59	3.0x10E-5
Plutonium-238	5.7	2.9x10E-4
Plutonium-239	5.3	2.7x10E-4
Plutonium-240	5.3	2.7x10E-4
Plutonium-241	0.23	1.2x10E-5
Radium-226	11	5.6x10E-4
Strontium-90 and Yttium-90	1.131	5.8x10E-5
Thorium-232	6.2	3.2x10E-4
Tritium	0.00568	2.9x10E-7
Uranium-234	4.9	2.5x10E-4
Uranium-235	4.6	2.4x10E-4
Uranium-238	4.3	2.2x10E-4

Table 4-7. Fractions of radionuclides in tissue from ingestion

Radionuclide	Fraction Reaching Blood	Reference and Notes
Americium	2x10E-3	ICRP 1986, americium incorporated in tumbleweed
Cesium	1	In equilibrium with sodium and potassium in tissues
Plutonium	1x10E-3	ICRP 1986, plutonium in soluble form
Radium	0.3	ICRP 1959
Strontium	0.3	ICRP 1959
Thorium	1x10E-3	ICRP 1986, thorium nitrate
Tritium	1	In equilibrium with tissue water
Uranium	1 1x10E-2	Birds, Kennedy and Strenge 1992 Mammals, Kennedy and Strenge 1992

Equation 4.13 is used to estimate the internal absorbed dose for animals.

$$D\left(\text{rad}/\text{day}\right) = [(\text{g}\cdot\text{rad})/(\text{pCi}\cdot\text{day})]\times[\text{pCi}/(\text{g of food})]\times(\text{fraction reaching blood})\times$$
$$[(\text{g of food})/(\text{g of animal body weight}\cdot\text{day})]\times(\text{retention time of radionuclide [day]})$$

Equation 4.13

The form of the dietary intake term in units of g/(g of animal body weight-day) is the same as in Equation 4.8. For the fraction of energy deposition that is due to alpha particle absorption *(Fa)* by tissue, the relative biological effectiveness is about 20 times that of beta or photon (gamma and x-ray) emissions (NCRP 1989). Thus, the total internal dose is given by.

$$D\left(\text{rad}/\text{day}\right)=\left(Fa\cdot\frac{\text{rad}}{\text{day}}\right)\cdot 20+\left(1-Fa\cdot\frac{\text{rad}}{\text{day}}\right)$$ Equation 4.13

The retention time (R) in days is calculated as (Baker and Soldat 1992):

$$R=\frac{1-e^{-\lambda Tc}}{\lambda}$$ Equation 4.14

Where:

- $\lambda = \lambda r + \lambda b$
- $\lambda r = \ln 2/Tr$, where Tr is the radiological half-life of the radionuclide
- $\lambda b = \ln 2/Tb$, where Tb is the biological half-life of the radionuclide
- T_c = exposure duration, 365 days

The half-lives of radionuclides of interest are presented in Table 4-8.

Table 4-8. Radiological (Tr) and biological (Tb) half-lives in days[a]

Radionuclide	Tr	Tb
Americium-241	1.6×10^{5}	2.0×10^{4}
Cesium-137/ Barium-137m	1.1×10^{4}	115
Plutonium-238	3.2×10^{4}	6.5×10^{4}
Plutonium-239/240	8.8×10^{6}	6.5×10^{4}
Radium-226	5.8×10^{5}	8.1×10^{3}
Strontium-90/ Yttrium-90	1.1×10^{4}	1.4×10^{4}
Thorium-232	5.2×10^{12}	5.7×10^{4}
Tritium	4.5×10^{3}	10
Uranium-234	8.9×10^{7}	100
Uranium-235	2.6×10^{11}	100
Uranium-238	1.6×10^{12}	100

[a] Baker and Soldat 1992

The use of the fraction reaching the blood from food to calculate dose to animals includes an assumption that the tissue concentration for the organs is the same. In reality, the fraction of the radionuclide reaching tissues is dependent on the metabolism of the element. For tritium, the concentration in the blood and tissues is nearly in equilibrium, whereas for actinides only a small fraction of the concentration in the blood is absorbed into tissues. Hence, for the actinides this assumption will overestimate the dose to organs such as the reproductive organs and other soft tissues.

Calculating Internal Dose for Plants, Invertebrates, and Aquatic Animals

Internal dose for plants is calculated as:

$$D\left(\text{rad}/\text{day}\right)=\left[(\text{g}\cdot\text{rad})/(\text{pCi}\cdot\text{day})\right]\times\left[(\text{pCi})/(\text{g-soil})\right]\times \text{TF}_p$$ Equation 4.15

where TF_p is the plant to soil concentration factors of the element of interest (Table 4-9) or the ratio for the pCi/g of wet weight plant tissue to the dry weight soil concentration in pCi/g measured in mature plants. The product of pCi/g

of soil and TF_p provides a concentration in tissue material. For the purposes of radiological screening, TF_p is extracted from the default values for the RESRAD computer code to retain consistency with human health dose assessment (Wang et al. 1993).

Table 4-9. Soil to plant concentration factors (TF_p) for radionuclides

Radionuclide	TF_p
Americium	0.001
Cesium	0.04
Plutonium	0.001
Radium	0.04
Strontium	0.3
Thorium	0.001
Tritium	4.8
Uranium	0.0025

For calculating doses to soil invertebrates and aquatic organisms, similar concentration factors are used for the ratio of organism to soil or water concentrations, respectively. For invertebrates, the default factor is 1. For aquatic organisms, values are taken from Baker and Soldat (1992). The aquatic values are presented in Table 4-10, below.

Table 4-10. Radionuclide concentration factors for aquatic organisms

Radionuclide	Fish	Crustacean	Mollusc	Plant
Americium	100	100	100	3,000
Cesium	2000	100	100	500
Plutonium	250	100	100	890
Radium	50	1000	1000	30,000
Strontium	50	100	100	3,000
Thorium	100	100	100	3000
Tritium	1	1	1	1
Uranium	50	100	100	900

External Dose to Biota

In addition to the absorbed dose from radionuclides deposited internally, the organism receives a dose from radioactive contaminants in the soil. External exposure from radionuclides in soil is from gamma-rays, x-rays, beta particles, and electrons. External radiation exposure from alpha particles is considered negligible because the first cell layer stops the alpha particles. The amount of exposure is strongly dependent on the location of the receptor in relation to the soil. Animals and plants on the surface of the soil will receive less dose than those in the soil. Estimation of the dose is a complex calculation; however, such calculations have been conducted to estimate the dose rate at 1 m above the soil surface for radionuclides at several depths in soil (Eckerman and Ryman, 1993). Dose estimators for immersion in water have also been calculated, for radionuclides. While these results use data for the human body, the skin dose estimates will be used here for estimation of dose to biota. The skin dose estimator is the largest estimator compared to other organ doses and does not account for self-shielding of the internal organs. For plant cells and animals of small radius this dose estimator will account for dose from penetrating and weakly penetrating radiation (beta particles, electron emissions, and low energy x-rays). The dose to larger biota such as fox, coyote, deer, elk, and raccoon will be overestimated for every radionuclide considered, because of absorption of weak radiation by fur and hide.

External Dose To Terrestrial Animals Or Plants Living In Or Burrowing In Soil

The dose to an animal or plant part is dependent on where the living habits place the organism in relation to the area of soil containing radionuclides. In the case of burrowing animals (such as earthworms) or plant roots, the organism is submerged in a radiation field that is dependent on the radionuclide distribution in the soil. The depth of the radionuclides in soil is variable and may immerse the organism in only a small thickness of soil or in an infinitely contaminated media. For the purpose of screening, the dose estimators used will be for infinitely contaminated media. The radionuclide dose estimation coefficients of Eckerman and Ryman (1993) will be used for immersion in an infinite water source. Use of water rather than soil can be corrected for the density of soil; the dose estimation coefficients are reduced by a factor of 62.5% assuming an average soil density of 1.6 g/cubic centimeter. The radiation adsorption coefficient of soil is even higher because of the presence of elements such as iron, so this approach provides an overestimate of the dose. The dose coefficients for each radionuclide and its progeny are listed in Table 4-11.

Table 4-11 External dose coefficients for biota living in soil and on the soil surface

Radionuclide	External Dose Factor (g-rad/ pCi-day) to organisms	
	living in Soil	living on Soil < 0.5 m
Americium-241	5.96E-07	2.98E-07
Cesium-137/Barium-137m	1.71E-05	8.56E-06
Plutonium-238	1.91E-08	9.55E-09
Plutonium-239+240	1.04E-08	5.18E-09
Plutonium-241	5.00E-11	2.50E-11
Radium-226+progeny	6.06E-05	3.03E-05
Strontium-90/Yttrium-90	1.53E-05	7.64E-06
Thorium-232+progeny (also background)	8.12E-05	4.06E-05
Uranium-234	1.91E-08	9.55E-09
Uranium-235 +Protactinum-231	4.82E-06	2.41E-06
Uranium-238+Thorium-234, Protactinium-234m, Protactiuium-234	6.24E-05	3.12E-05
Chemically separated natural Uranium	3.01E-05	1.51E-05
Depleted Uranium	5.65E-05	2.82E-05
Primordial Uranium + progeny (background)	1.23E-04	6.15E-05

For radionuclides that form radioactive progeny, the dose coefficients need to be added to account for the total dose in the decay chain. For radionuclides added to the environment at the Laboratory, the oldest additions would be fifty years ago. In decay chains (such as cesium-137 to barium-137m) equilibrium can be assumed because of the short half-life of the progeny. For the actinide decay chains, equilibrium does not exist for long decay chains, where the half-life of the progeny is long compared to the period the initial element was deposited in the environment. An example of such a decay chain is uranium-238. The values in Eckerman and Ryman (1993) are listed in [Sv/(Bq-s-cubic meter)]. A combined conversion factor of 2.00 x 10E11 was used to correct for the density difference between soil and water and to convert to units of g-rad/pCi-day.

External Dose Terrestrial Animals and Plants Living On or Above Soil

For animals and plants that live on or above the soil but are less than 0.5 m tall, the dose estimator used will be one half the immersion dose coefficient for water, corrected for the density of soil (Table 4-12). This approach assumes that the biota is exposed to the radiation from a hemisphere of infinitely contaminated soil with no distance between the biota and the soil surface. The results provided will overestimate the dose. Use of the inverse square relationship for large disc shapes or hemisphere distributions of contaminants to estimate the dose for distances less than 0.5 m to the surface underestimates the dose more and more as the distance to the surface decreases (Schleien 1992).

For plants and animals that are above the soil surface at 0.5 m or greater, the external dose coefficients for soils contaminated to an infinite depth are used to estimate the dose based on the soil concentration. The external dose coefficients are calculated for 1 m above the soil surface. The inverse square relationship between radiation dose and distance is used to provide dose estimates at 0.5 m and 2 m (Equation 4.15). The correction factor 5.12×10^{11} is used to convert from units of [Sv/(Bq-s-cubic meter)] to [(g-rad)/(pCi-day)].

The external dose to an organism is estimated by multiplying the dose coefficient from either Table 4-11 or Table 4-12 depending on the living habits of the biota and the soil concentration in pCi/g.

Table 4-12. External dose coefficients for biota living 0.5, 1, and 2 meters above soil contaminated at an infinite depth

Radionuclide	External Dose Factor (g-rad/pCi-day) for soil biota living above soil by: 0.5m	1m	2m
Americium-241	6.36E-07	1.59E-07	3.97E-08
Cesium-137 + Barium-137m	4.64E-05	1.16E-05	2.90E-06
Plutonium-238	1.04E-08	2.60E-09	6.51E-10
Plutonium-239+240	7.48E-09	1.87E-09	4.68E-10
Plutonium-241	7.60E-11	1.90E-11	4.84E-12
Radium-226 + progeny	1.54E-04	3.84E-05	9.60E-06
Strontium-90 + Yttrium-90	2.06E-05	5.15E-06	1.29E-06
Thorium-232 + progeny (also background)	2.15E-04	5.38E-05	1.34E-05
Uranium-234	1.23E-08	3.07E-09	7.67E-10
Uranium-235 + Protactinium-231	1.15E-05	2.87E-06	7.17E-07
Uranium-238 + Thorium-234, Protactinium-234m, Protactinium-234	1.64E-04	4.11E-05	1.03E-05
Chemically separated natural Uranium	7.92E-05	1.98E-05	4.95E-06
Depleted Uranium	1.65E-04	4.12E-05	1.03E-05
Primordial Uranium + progeny (background)	2.33E-04	5.46E-05	1.45E-05

Calculating External Doses to Aquatic Organisms

For calculating doses to organisms immersed in water, the immersion coefficients in Table 4-11 are used after converting them back to account for the lower attenuation by water (dividing the coefficients by 62.5%). External exposure to contaminated sediments is calculated directly using the coefficients in Table 4-11 for organisms in or on sediment.

4.2.4.3 Calculating Ecological Screening Levels for Radionuclides

The ESLs (environmental levels that lead to a calculated dose equal to the dose limit [HQ = 1]) are obtained by back-calculating the media concentration from the dose limit value through the dose calculations given above.

4.2.5 ESRV/ESL Summary

Using the information and equations presented in the preceding sections, ESLs are back-calculated from the ESRVs (this is straightforward when there is no significant water ingestion pathway, but the appropriate adjustment must be made when there is significant contaminant ingestion from drinking water). This approach allows comparison of the site-specific media concentrations of contaminants to ESLs (e.g. evaluating HQs and HIs) to determine if the site presents a potential threat to the environment. The alternative is to use the information and equations above to

calculate the site-specific doses and compare these to the ESRVs. These two approaches are equivalent. The Laboratory has chosen to develop the ESLs, as these values are more useful to the field investigators. The ESRVs and the relationship of these values to ESLs are summarized in Table 4-13.

Table 4-13. Summary of ESRV/ESL relationship

Screening receptor type	Nonradiological ESRV/ESL	Radiological ESRV/ESL
Terrestrial plants	ESRV (NOAEL) in units of soil concentration, ESL is equal to (ESRV) NOAEL	NOAEL is in units of rad/day, ESL requires calculation
Terrestrial invertebrates	ESRV (NOAEL) in units of soil concentration, ESL is equal to ESRV (NOAEL)	NOAEL is in units of rad/day, ESL requires calculation
Terrestrial wildlife	ESRV (NOAEL) in units of mg/kg/day, ESL requires calculation	NOAEL is in units of rad/day, ESL requires calculation
Aquatic receptors	Ambient water quality and sediment standards will be proposed as ESLs, no calculation required	NOAEL is in units of rad/day, ESL requires calculation

4.3 Screening Evaluation/Uncertainty Analysis

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

- likelihood of screening receptors (or receptors in respective feeding guild) being present at the PRS
- likelihood that the screening pathways are complete
- likelihood that significant pathways not included in the ecological screening assessment are complete (e.g., the inhalation pathway)
- qualification of the analytical data
- possible bias or uncertainty introduced in the sample collection process
- artificially elevated quantitation limits
- likelihood that the maximum value is truly the maximum for the site
- likelihood that the maximum value represents a reasonable exposure concentration (if the data are adequate, HQs and HIs calculated for the maximum value may be contrasted with those calculated for the 95th upper confidence level [UCL] for the mean)
- uncertainty in contaminant background concentrations
- environmental fate and transport of contaminants (including uncertainties associated with the assessment of persistent bioaccumulation and/or magnification)
- possibility of cumulative effects
- additivity of effects assumed by the HI calculation
- chemical form likely to be present in the environment
- constituent toxicity values
- possibility of contaminant interactions
- assumed values of intake parameters
- multiple exposure pathway assumptions
- metabolic fate of COPEC
- ecological factors that affect receptor exposure
- size of the contaminated area relative to the receptor home range
- distribution of analytical results—nature and extent

It is important to identify the type of effect uncertainty introduces into risk characterization. Do the uncertainties lead to a significant bias in risk estimates, or do uncertainties lead to a less precise estimate of risk? What data could be

collected to cost-effectively reduce uncertainty? What part of the uncertainty is linked to variation in the dynamical nature of contaminant releases and natural variation in biological populations?

4.4 Interpretation

At the completion of the screening evaluation, the risk assessor communicates the results to the risk manager, with an emphasis on the uncertainty analysis. The purpose of the communication is to provide the risk manager with sufficient information to support a risk management decision with respect to ecological concerns. It 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 characterize risk.

There are four possible decisions based on ecological evaluations at this point:

1. There is adequate information to conclude that the ecological risks are negligible and NFA for ecological risk is appropriate.
2. There are sufficient lines of evidence to document potential or actual adverse ecological effects. Thus, remediation to approved risk-based levels or background may be needed (e.g., cleanup or stabilization). Note that risk-based remediation levels are <u>not</u> equal to ecological risk screening values.
3. Ecological risks are not negligible, but there is not sufficient information to suggest that adverse ecological effects are occurring. Thus additional ecological risk assessment is needed to properly evaluate the potential for adverse ecological impacts.
4. There is not adequate information to make a risk management decision. Data needs must be identified to effectively collect additional data.

If decisions 1 or 2 are reached, the recommendation is then evaluated along with potential human health impacts, surface water, groundwater, and other regulatory requirements to make an integrated site recommendation.

5.0 References

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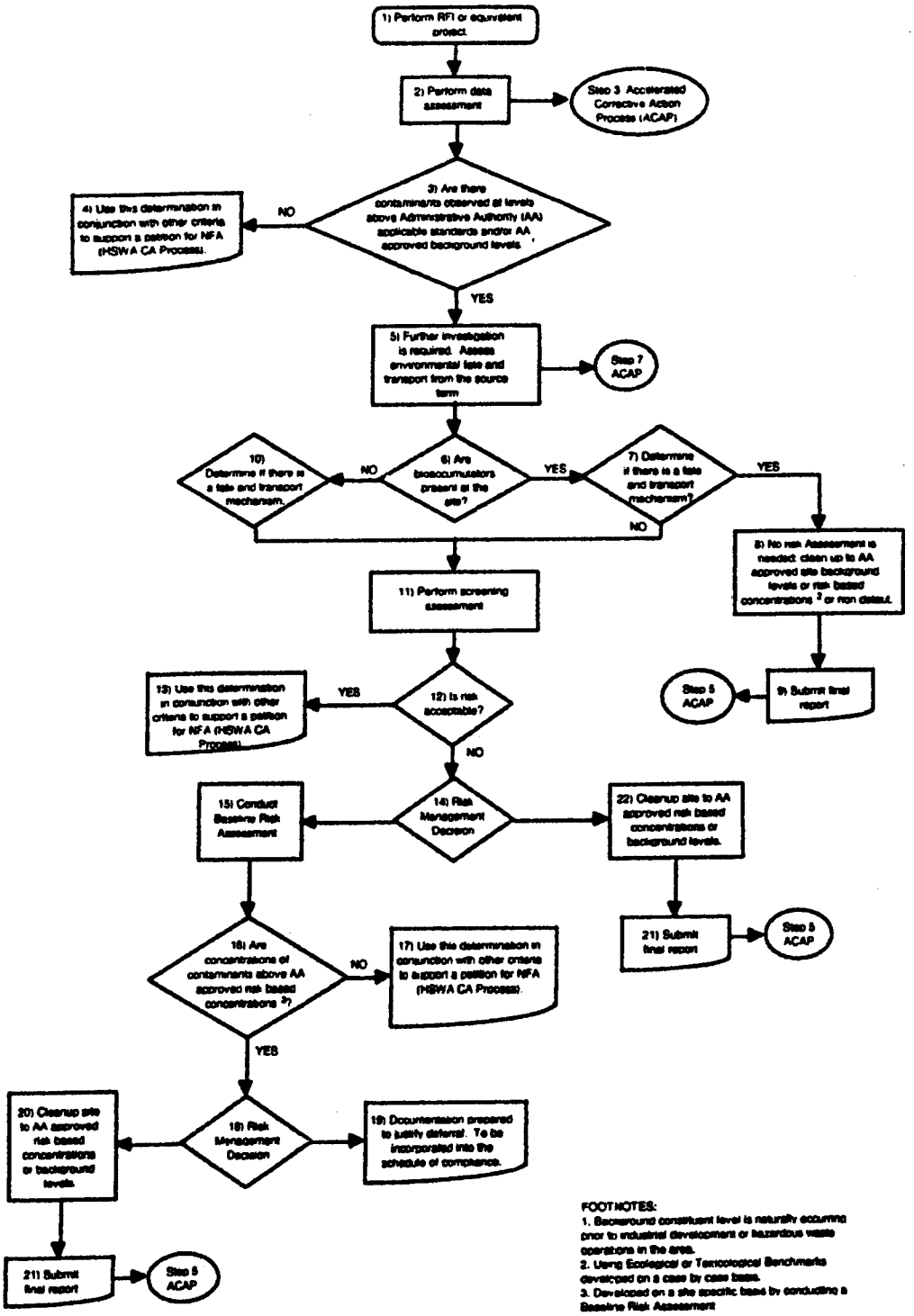
Appendix A

NMED Risk-Based Decision Tree

Risk Based Decision Tree



FOOTNOTES:
1. Background constituent level is naturally occurring prior to industrial development or hazardous waste operations in the area.
2. Using Ecological or Toxicological Benchmarks developed on a case by case basis.
3. Developed on a site specific base by conducting a Baseline Risk Assessment

NMED Risk-Based Decision Tree Description (March 4, 1998)

All or portions of this Risk-based Decision Tree may not be applicable to all facilities. Please contact the RPMP Facility Manager if applicability is questionable.

Box 1: Perform RCRA Facility Investigation (RFI) or equivalent project.

Box 2: Perform Data Assessment. (This step corresponds to Step 3 in the Accelerated Corrective Action Process [ACAP]).

Criteria:

1. Compare results to data quality objectives (DQOs);
2. Determine the nature, rate, and extent (vertical and horizontal) of contamination;
3. Compare the maximum constituent concentrations to the Administrative Authority (AA)-approved:
   a. Background for inorganic constituent concentrations,
   b. Fallout for radionuclide concentrations, or
   c. MDLs, PQLs, or EQLs for organic constituent concentrations; and
4. Compare the maximum constituent concentrations to AA applicable standards or other approved values.

Box 3: Are there contaminants above Criterion 3 and 4?

If **NO**, move to Box 4
If **YES**, move to Box 5

Box 4: Use this determination in conjunction with other criteria to support a petition for NFA (HSWA Corrective Action Process).

Box 5: Assess Environmental Fate & Transport from the Source Term. (This step corresponds to Step 7 of the ACAP.)

Consider the following:

1. Determine if bioaccumulation in plant and/or animal tissue is of concern. The constituent is considered a bioaccumulator, if:
   a. For inorganics (including radionuclides), the bioconcentration factor (BCF) exceeds 40, or
   b. For organics, the logarithm of the octanol-water partition coefficient (log K_{ow}) exceeds 4.

2. Other important environmental fate processes to be evaluated include, but are not limited to the following:
   a. Soil/sediment sorption/desorption potential;
   b. Leaching to underlying ground water and discharging into surface water and/or other habitats;
   c. Vertical migration in unsaturated zone;
   d. Erosion of contaminated soils as a potential contaminant transport pathway;
   e. Other movement of contaminant within various components of the ecosystem (e.g., plant uptake, soil or aquatic invertebrate uptake); and
   f. Chemical and biological transformation and degradation processes in abiotic media.

Box 6: Are bioaccumulators present at the site?

The constituent is considered a bioaccumulator, if:

1. for inorganics (including radionuclides), the bioconcentration factor (BCF) exceeds 40, or

2. for organics, the logarithm of the octanol-water partition coefficient (log K_{ow}) exceeds 4.

If YES, move to Box 7.
If NO, move to Box 10.

Box 7 : Determine if there is a fate and transport mechanism?

If bioaccumulators are present at the site, evaluate the following environmental fate and transport processes:

1. Soil/sediment sorption/desorption potential;
2. Leaching to underlying ground water and discharging into surface water and/or other habitats;
3. Vertical migration in unsaturated zone;
4. Erosion of contaminated soils as a potential contaminant transport pathway;
5. Other movement of contaminant within various components of the ecosystem (e.g., plant uptake, soil or aquatic invertebrate uptake); and
6. Chemical and biological transformation and degradation processes in abiotic media.

If, as a result of this evaluation the environmental transport is of concern, move to Box 8.

If, as a result of this evaluation the environmental transport is not of concern, move to Box 11.

Box 8: No risk assessment needed: clean up the site to AA-approved site background levels or risk-based concentrations or non-detect.

Criteria:

1. Background constituent level is the naturally occurring concentration of inorganic chemicals (including naturally occurring radionuclides) present in the area upgradient or upwind from the site prior to industrial or hazardous waste operations in the area. Fallout concentrations of man-made radionuclides derived from sources unrelated to the facility activities are considered baseline levels. A facility shall have it's background inorganic constituent concentrations (including naturally occurring radionuclides) and baseline fallout concentrations of man-made radionuclides approved by the AA prior to their use.

2. Risk-based concentrations are represented by ecological or toxicological benchmarks/criteria developed on a case by case basis, addressing the results of the fate and transport evaluation to protect human health and the environment.

3. The concept of "non detect" applies to man-made organic constituents that shall be cleaned up to levels of their PQLs, EQLs, or an analytical method detection limit, if cleanup to "non detect" is the elected remedy for the site.

Box 9: Submit final report. (This step corresponds to Step 5 of the ACAP.)

Box 10: Determine if there is a fate and transport mechanism.

If BIOACCUMULATORS are NOT present at the site, at a minimum, evaluate the following environmental fate and transport processes. The results of this evaluation shall be used to adequately focus a screening assessment (see Box 11).

1. Soil/sediment sorption/desorption potential;
2. Leaching to underlying ground water and discharging into surface water and/or other habitats;
3. Vertical migration in unsaturated zone;
4. Erosion of contaminated soils as a potential contaminant transport pathway;
5. Other movement of contaminant within various components of the ecosystem (e.g., plant uptake, soil or aquatic invertebrate uptake); and

6. Chemical and biological transformation and degradation processes in abiotic media.

Box 11: Perform Screening Assessment.

1. Perform Ecological Screening Assessment:
   a. Develop site conceptual model and relevant food webs, and select receptors representing all feeding guilds and trophic levels;
   b. In the absence of site-specific data, estimate potential exposure of these receptors to site contaminants using the following conservative/protective assumptions and exposure parameter values:
      i. Use the highest measured contaminant concentrations at a site to represent the exposure point concentration to biota;
      ii. Use the highest (conservative) literature transfer coefficients to address constituents bioconcentration/bioaccumulation and biomagnification potential and food chain transfer;
      iii. Assume the receptor resides 100% of time in the contaminated area;
      iv. Assume the constituents bioavailability to be 100%;
      v. Assume the most sensitive life stage of the receptor for the exposure assessment;
      vi. Use minimum body weight and maximum ingestion rate;
      vii. Assume that 100% of diet consists of the most contaminated dietary component; however, if evaluating potential exposure of an omnivore receptor, it acceptable to assume that diet consists of e.g., about 50% of plant material and about 50% of invertebrates (with soil ingestion rate estimate at less than 1%);

      In the subsequent phases of the ACAP (e.g., ecological baseline risk assessment) following collection of additional information/data, these conservative assumptions can be examined and adjusted (relaxed) to better reflect site and receptor-specific conditions.

   c. Select a current literature no-observed-adverse-effect level (NOAEL) to represent the ecotoxicity screening reference value (ESRV) (i.e., exposure dose). NOAELs shall be derived for each ecologically significant exposure pathway/route and they shall:
      i. Utilize the most sensitive species (select most sensitive assessment endpoints);
      ii. Be derived from chronic mortality, reproduction, and growth studies; and
      iii. Utilize the lowest NOAEL.

      In the absence of a literature NOAEL, the NOAEL can be estimated by applying an uncertainty/safety factor of 10 for the lowest available lowest-observed-adverse-effect level (LOAEL) or of 100 for the lowest available acute toxicity value (LD50 or LC50) or effective concentration (EC50). If toxicity values are not available for the habitat of interest (e.g., terrestrial or aquatic), toxicity values derived from other habitat studies should not be used, and the constituent should be retained for further evaluation in the ecological (baseline) risk assessment. In any case, the original study (i.e., primary literature from which the ESRV is derived) shall be examined and referenced.

   d. Calculate hazard quotients (HQs) and hazard indices (HIs) for exposure to multiple contaminants of receptors of concern.

   e. And/or estimate abiotic media (e.g., soil, sediment, or water) ecological screening levels (ESLs) from calculated HQs (for receptor's exposure to a single contaminant) or HIs (for receptor's exposure to multiple contaminants) assuming HQ=1 or HI=1, respectively;

   f. Perform an uncertainty analysis; at a minimum, analysis should focus on the following key sources of uncertainty associated with a screening assessment:
      i. Definition of a site physical setting (e.g., exposure assumptions such as the likelihood of exposure pathways and land uses actually occurring, and receptors selected for evaluation);
      ii. environmental monitoring data (e.g., media-contaminant distribution, using laboratory or otherwise qualified data, lack of quantitation, high detection limits);
      iii. Environmental fate and transport models;

iv. Constituent toxicity values (or their lack) and interactions;
v. Intake parameters and their assumed values; and
vi. Multiple pathway exposure assumptions.

g. Combine the results of Steps (d) or (e) and (f) above.

In the subsequent phases of the Corrective Action process (e.g., ecological baseline risk assessment) and following collection of additional information/data, these conservative assumptions can be examined and adjusted (relaxed) to better reflect site and receptor-specific conditions.

2. Perform Human Health Screening Assessment:

a. Follow the process presented in the RCRA Permits Management Program (RPMP) position paper entitled "*Human Health Risk-Based Screening Action Levels and Screening-Level Assessment*".

Note, that although food-chain transfer of contaminants has been excluded from consideration in calculation of human health screening action levels (HHSALs) it may be important under certain exposure scenarios (e.g., agricultural) or for certain exposure pathways (e.g., human consumption of home-grown produce under residential exposure scenario). Therefore, when these exposure scenarios or pathways are of potential concern at a site, a contaminant food-chain transfer shall also be evaluated and the results shall be incorporated into the revised HHSAL.

b. Perform an uncertainty analysis; at a minimum, analysis should focus on the following key sources of uncertainty associated with a screening assessment:
i. Definition of a site physical setting (e.g., exposure assumptions such as the likelihood of exposure pathways and land uses actually occurring, and receptors selected for evaluation);
ii. Environmental monitoring data (e.g., media-contaminant distribution, using laboratory or otherwise qualified data, lack of quantitation, high detection limits);
iii. Environmental fate and transport models;
iv. Constituent toxicity values (or their lack) and interactions;
v. Intake parameters and their assumed values; and
vi. Multiple pathway exposure assumptions.

c. Combine the results of Steps (1) or (2) and (3) above.

In the subsequent phases of the Corrective Action process (e.g., human health baseline risk assessment) and following collection of additional information/data, these conservative assumptions can be examined and adjusted (relaxed) to better reflect site-specific conditions.

Box 12: Is risk acceptable?

Use both ecological and human health screening assessment determinations.

1. Ecological

Ecological risk is considered acceptable, if:

a. HQ<1 (for receptor's exposure to a single contaminant) or HI<1 (for receptor's exposure to multiple contaminants); and/or
b. The maximum constituent media concentrations are below their respective media ecological screening level (ESL)s.

2. Human Health

Human health risk is considered acceptable, if:

a. For noncarcinogens, HQ<1 (for exposure to a single contaminant) or HI<1 (for exposure to multiple contaminants), and for carcinogens, excess lifetime risk of developing cancer by an individual is less than 10^{-6} for Class A and B carcinogens and less than 10^{-5} for Class C carcinogens; and/or

b. The maximum constituent media concentrations are below their respective human health screening action levels (HHSALs).

If answer to both 1 and 2 is **YES**, move to Box 13.

If answer to either 1 and 2 is **NO**[1], move to Box 14.

Box 13: Use this determination in conjunction with other criteria to support a petition for NFA (HSWA Corrective Action Process).

Box 14: Risk Management Decision

A risk management decision (RMD) must be made at this point. It should be determined whether it would be less costly to clean up the site to generic preliminary cleanup levels (PCLs) based on risk-based concentrations (HHSALs and/or ESLs, whichever is more stringent) or to collect more site-specific data and conduct baseline risk assessment (i.e, ecological and/or human health baseline risk assessments [EBRA and/or HHBRA]). As a result of these EBRA and HHBRA, site-specific risk-based cleanup levels (CLs) could be established. Consideration should be given to fact that even after considerable expense conducting an EBRA or HHBRA, the site may still need to be cleaned up to PCLs.

Box 15: Conduct Baseline Risk Assessment.

Both ecological and human health baseline risk assessments should be performed, if warranted. Additional information and site-specific data shall be collected to address the critical data needs (gaps) identified during the ecological and human health screening assessments that will support baseline risk assessments. The following steps shall be considered for site-specific baseline risk assessments:

1. Collect additional information and/or site-specific data;
2. Select Contaminants of Potential Concern (COPCs);
3. Evaluate receptors exposure;
4. Evaluate contaminants toxicity, including potential interactions;
5. Estimate and characterize risk (including quantification of risk and uncertainty analysis);
6. Provide risk interpretation and recommendations; and
7. Calculate revised ESLs (RESLs) and/or HHSALs (RHHSALs) and obtain AA approval.

Box 16: Are concentrations of contaminants above AA approved risk-based concentrations?

Compare site-specific RESLs and RHHSALs to the site media constituent concentrations.

If site-specific RESLs and/or RHHSALs exceed the site media constituent concentrations, move to Box 17.

If site-specific RESLs and/or RHHSALs are below the site media constituent concentrations, move to Box 18.

Box 17: Use this determination in conjunction with other criteria to support a petition for NFA (HSWA Corrective Action Process).

[1]This determination does not automatically require corrective action (e.g., cleanup) but may require more analysis (e.g., a baseline risk assessment should be conducted).

Box 18: Risk Management Decision

A risk management decision must be made at this point. A decision must be made to defer further action at this time (Box 19) or to cleanup the site to AA approved site-specific risk-based cleanup levels (CLs)(based on RESLs and/or RHHSALs, whichever is more stringent)(Box 20).

Box 19: Documentation prepared to justify deferral. To be incorporated into the schedule of compliance.

Prepare documentation to justify deferral. If approved by AA, deferral will be incorporated into the schedule of compliance.

Box 20: Cleanup site to AA-approved risk-based concentrations or background levels.

Cleanup the site to AA approved site-specific risk-based cleanup levels (CLs) or background levels or "non detects" (as defined in Box 8, Steps 1 and 3).

Box 21: Submit Final Report. (This step corresponds to Step 5 of the ACAP.)

Requirements:

1. Verification sampling and analysis is conducted to determine COPCs concentrations have been reduced to RCLs or background levels or "non-detects" (as defined in Box 8, Steps 1 and 3).

2. This determination should be used in conjunction with other criteria to support petition for NFA (HSWA CA Process).

Box 22: Cleanup site to AA-approved risk-based concentrations or background levels.

1. Calculate generic preliminary risk-based cleanup levels (PCLs) based on ESLs (RESLs) and/or HHSALs (RHHSALs) and obtain AA approval.

2. Cleanup the site to AA approved PCLs or background levels or "non-detects" (as defined in Box 8, Steps 1 and 3).

Box 23: Submit Final Report. (This step corresponds to Step 5 of the ACAP.)

Requirements:

1. Verification sampling and analysis is conducted to determine COPCs concentrations have been reduced to PCLs or background levels or "non detects" (as defined in Box 8, Steps 1 and 3).

2. This determination should be used in conjunction with other criteria to support petition for NFA (HSWA CA Process).

Appendix B

Ecological Scoping Checklist, Parts A, B, and C

Ecological Scoping Checklist: Part A
Scoping Meeting Documentation

Site ID	
Nature of PRS releases (indicate all that apply)	Solid Liquid Gaseous Other, explain
List of Primary Impacted Media (indicate all that apply)	Surface soil Surface water/sediment Subsurface Groundwater Other, explain
FIMAD vegetation class (indicate all that apply)	Water Bare Ground/Unvegetated Spruce/fir/aspen/mixed conifer Ponderosa pine Piñon juniper/juniper savannah Grassland/shrubland Developed
Is T&E Habitat Present? list species if applicable	
Provide list and description of Neighboring/ Contiguous/ Upgradient PRSs (consider need to aggregate PRS for screening)	
AP 4.5 Part B Information Run-off score (out of 46) Terminal point of surface water transport	
Other Scoping Meeting Notes	

Ecological Scoping Checklist: Part B
Site Visit Documentation

Site ID	
Date of Site Visit	
Site Visit Conducted by	

Receptor Information:

Estimate cover	% vegetated % wetland % structures/asphalt, etc.
Field notes on the FIMAD vegetation class	
Field notes on T&E Habitat, if applicable	
Are ecological receptors present at the PRS? (yes/no/uncertain) Provide explanation	

Contaminant Transport Information:

Surface water transport Field notes on the terminal point of surface water transport (if applicable)	
Are there any off-site transport pathways? (yes/no/uncertain) Provide explanation	

Ecological Effects Information:

Physical Disturbance (provide list of major types of disturbances)	
Are there obvious ecological effects? (yes/no/uncertain) Provide explanation	

Ecological Scoping Checklist: Part B
Site Visit Documentation (cont.)

No Receptor/ No Pathways:

If there are no receptors and no offsite transport pathways the remainder of the checklist should not be completed. Stop here and provide any additional explanation/justification for proposing an ecological No Further Action recommendation (if needed).

Data Adequacy:

Do existing data provide information on the nature, rate and extent of contamination? (yes/no/uncertain) Provide explanation (consider if the maximum value was captured by existing sample data)	
Do existing data for the PRS address potential pathways of site contamination? (yes/no/uncertain) Provide explanation (consider if other sites could be impacting this PRS)	

Additional Field Notes:

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

Ecological Scoping Checklist: Part C
Ecological Pathways Conceptual Exposure Model

Provide answers to Questions A to Q and use this information to complete the Ecological Pathways Conceptual Exposure Model

Question A:

Could soil contaminants reach receptors via vapors?

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

Answer (yes/no/uncertain)

Provide explanation:

Question B:

Could the soil contaminants identified above 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 (yes/no/uncertain)

Provide explanation:

Question C:

Can contaminated soil be transported to aquatic ecological communities (use AP 4.5 run-off score and terminal point of surface water runoff to help answer this question)?

- If the AP 4.5 run-off score* equal to zero, this suggests that erosion at PRS is not a transport pathway. (* note that the runoff score is not the entire erosion potential score, rather it is a subtotal of this score with a maximum value of 46 points)
- If erosion is a transport pathway, evaluate the terminal point to see if aquatic receptors could be affected.

Answer (yes/no/uncertain)

Provide explanation:

Ecological Scoping Checklist: Part C
Ecological Pathways Conceptual Exposure Model (cont.)

Question D:

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

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

Answer (yes/no/uncertain)

Provide explanation:

Question E:

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

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

Answer (yes/no/uncertain)

Provide explanation:

Ecological Scoping Checklist: Part C
Ecological Pathways Conceptual Exposure Model (cont.)

Question F:

Could airborne contaminants interact with receptors through respiration of vapors?

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

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

Provide explanation:

Question G:

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

- Contaminants must be present as particulates in the air or as dust for this pathway to be viable.
- Exposure via inhalation of fugitive dust is particularly applicable to ground-dwelling species that would be exposed to dust disturbed by their foraging or burrowing activities or by wind movement.

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

Provide explanation:

Question H:

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

- Contaminants in bulk soil may partition into soil solution, making them available to roots.
- Exposure of terrestrial plants to contaminants present in particulates deposited on leaf and stem surfaces by rain striking contaminated soils (i.e., rain splash).

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

Provide explanation:

Ecological Scoping Checklist: Part C
Ecological Pathways Conceptual Exposure Model (cont.)

Question I:

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

- The chemicals may bioaccumulate in animals (see list of bioaccumulating chemicals presented in Table 1).
- Animals may ingest contaminated prey.

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

Provide explanation:

Question J:

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

- Incidental ingestion of contaminated soil could occur while animals grub for food resident in the soil, feed on plant matter covered with contaminated soil or while grooming themselves clean of soil.

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

Provide explanation:

Question K:

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

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

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

Provide explanation:

Ecological Scoping Checklist: Part C
Ecological Pathways Conceptual Exposure Model (cont.)

Question L:

Could contaminants interact with plants or animals through external irradiation?

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

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

Provide explanation:

Question M:

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

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

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

Provide explanation:

Question N:

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

- The chemicals may bioaccumulate in animals (see list of bioaccumulating chemicals presented in Table 1)
- Animals may ingest contaminated prey.

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

Provide explanation:

Ecological Scoping Checklist: Part C
Ecological Pathways Conceptual Exposure Model (cont.)

Question O:

Could contaminants interact with receptors via incidental ingestion of water and sediment?

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

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

Provide explanation:

Question P:

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

- If sediments are present in an area that is only periodically inundated with water, terrestrial species may be dermally exposed during dry periods.
- Terrestrial organisms may be dermally exposed to water-borne contaminants as a result of wading or swimming in contaminated waters.
- Aquatic receptors may be directly exposed to sediments or may be exposed 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 pathway (0=no pathway, 1=unlikely pathway, 2=minor pathway, 3=major pathway)

Provide explanation:

Ecological Scoping Checklist: Part C
Ecological Pathways Conceptual Exposure Model (cont.)

Question Q:

Could contaminants interact with plants or animals through external irradiation?

- External irradiation effects are most relevant for gamma emitting radionuclides.
- Burial of contamination severely attenuates radiological exposure.
- The water column acts to absorb radiation, thus external irradiation is typically more important for sediment dwelling organisms.

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

Provide explanation:

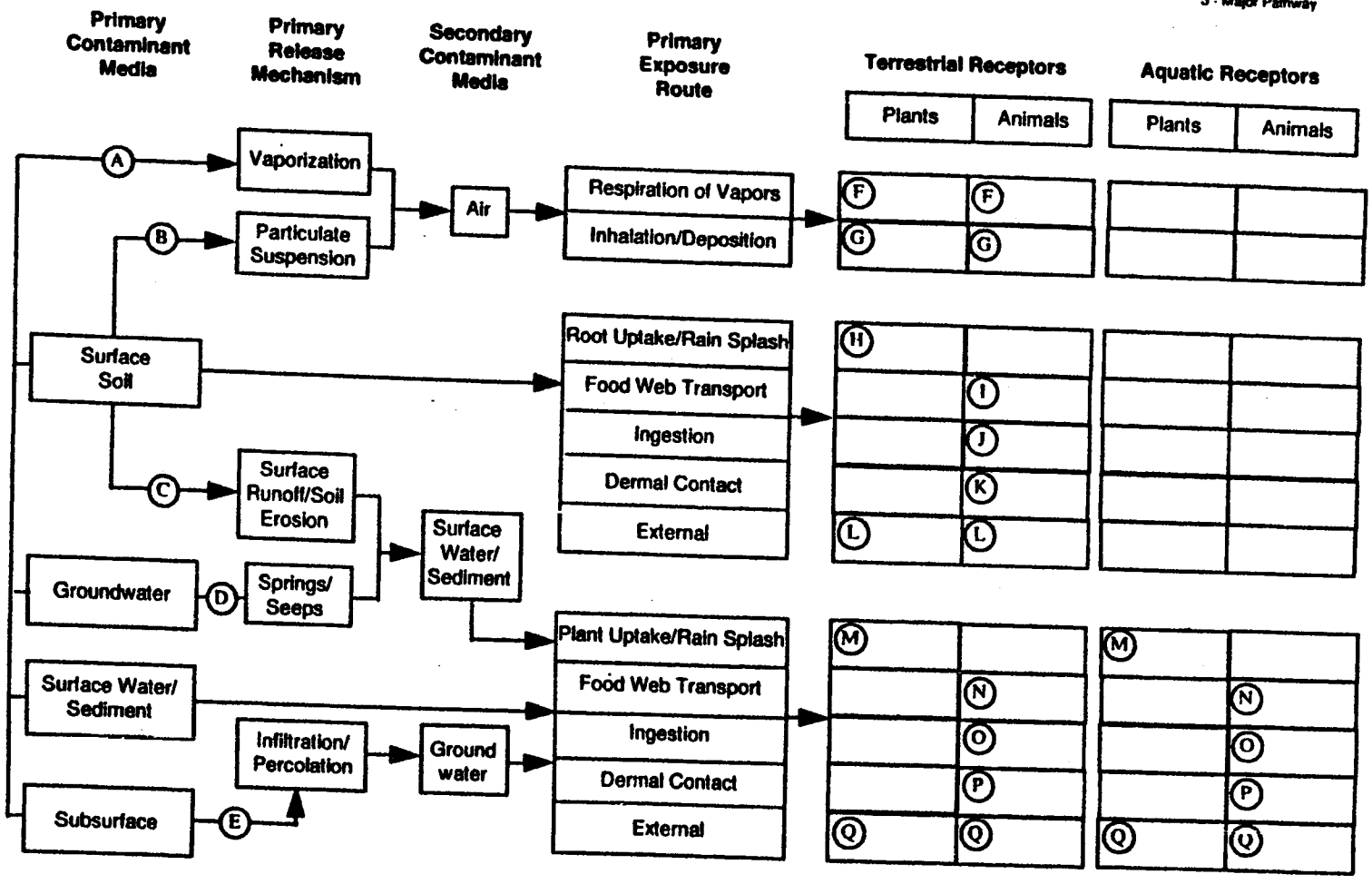
Table 1

List of Bioaccumulating Chemicals

Volatile and Semivolatile Organics	**PCBs/Pesticides**
Bis(2-ethylhexyl)phthalate	All Aroclors
Butyl benzyl phthalate	beta-BHC
Dibenzofuran	BHC-mixed isomers
Dichlorobenzene[1,4-]	Chlordane
Di-n-butyl phthalate	Chlorecone (Kepone)
Di-n-octyl phthalate	DDT and metabolites
Trichlorobenzene[1,2,4-]	Dieldrin
Acenaphthene	Endosulfan
Anthracene	Endrin
Benzo(a)anthracene	Heptaclor
Benzo(a)pyrene	Lindane
Benzo(b)fluoranthene	Methoxyclor
Benzo(g,h,i)perylene	Toxaphene
Benzo(k)fluoranthene	
Chrysene	**Inorganics**
Dibenzo(a,h)anthracene	Aluminum
Fluoranthene	Cadmium
Fluorene	Copper
Indeno(1,2,3-cd)pyrene	Lead
Phenanthrene	Mercury
Pyrene	Nickel
Pentachloronitrobenzene	Selenium
Pentachlorophenol	
Xylene (mixed isomers)	**Radionuclides**
	Americium-241
Dixoins/Furans	Cesium-137
2,3,7,8-tetrachloro-dibenzo(p)dioxin	Plutonium-238,239,240
2,3,7,8-tetrachloro-dibenzo(p)furan	Radium-226,228
	Strontium-90
	Thorium-228,230,232
	Uranium-234,235,238

Ecological Scoping Checklist:
Ecological Pathways Conceptual Exposure Model

KEY
0 - No Pathway
1 - Unlikely Pathway
2 - Minor Pathway
3 - Major Pathway



Signatures and certifications:

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

Name (printed): _____

Name (signature): _____

Organization: _____

Phone number: _____

Date completed: _____

Verification by (provide name, organization and phone number)

Name (printed): _____

Name (signature): _____

Organization: _____

Phone number: _____

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