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Date: FEB 13 2018
Refer To: ADEM-18-0016
LAUR: 18-20670

John Kieling, Bureau Chief
Hazardous Waste Bureau
New Mexico Environment Department
2905 Rodeo Park Drive East, Building 1
Santa Fe, NM 87505-6303

Subject: Development of Ecological Preliminary Remediation Goals for Los Alamos National Laboratory, Revision 1.1

Dear Mr. Kieling:

Enclosed please find two hard copies with electronic files of the Development of Ecological Preliminary Remediation Goals for Los Alamos National Laboratory, Revision 1.1. This document was revised to incorporate comments received from the New Mexico Environment Department (NMED) on December 12, 2017. Pursuant to Section XXIII.E of the 2016 Compliance Order on Consent, the U.S. Department of Energy and Los Alamos National Security, LLC conducted informal discussions with NMED and submitted proposed changes on January 22, 2018. Revision 1.1 of this document includes the changes discussed and reviewed by NMED. The changes include updates to several reference callouts and a table note that references revision changes.

If you have any questions, please contact Kent Rich at (505) 665-4272 (krich@lanl.gov) or Arturo Duran at (505) 665-7772 (arturo.duran@em.doe.gov).

Sincerely,

Bruce Robinson, Program Director
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Sincerely,

David S. Rhodes, Director
Office of Quality and Regulatory Compliance
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BR/DR/KR:sm

Enclosures: Two hard copies with electronic files – Development of Ecological Preliminary Remediation Goals for Los Alamos National Laboratory, Revision 1.1 (EP2018-0017)

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Development of Ecological Preliminary Remediation Goals for Los Alamos National Laboratory, Revision 1.1

Prepared by the Associate Directorate for Environmental Management

Los Alamos National Laboratory, operated by Los Alamos National Security, LLC, for the U.S. Department of Energy (DOE) under Contract No. DE-AC52-06NA253 and under DOE Office of Environmental Management Contract No. DE-EM0003528, has prepared this document. The public may copy and use this document without charge, provided that this notice and any statement of authorship are reproduced on all copies.

EXECUTIVE SUMMARY

Remediation of contaminated sites or media requires information on concentrations of chemicals in the environment that are protective of ecological receptors. These concentrations can be considered ecological preliminary remediation goals (EcoPRGs) and differ from ecological screening levels (ESLs). Ecological exposure models used to calculate ESLs for ecological risk-screening assessments were modified to derive soil EcoPRGs for representative assessment endpoint receptors. The modifications of the ESLs include the use of site-specific studies (bioassays, bioaccumulation) for plants, soil invertebrates, and wildlife and application of area use factors (individual or population) that are the fraction of a terrestrial animal's individual home range or assessment population area potentially affected by a contaminated site. Wildlife assessment population boundaries are based on a receptor's dispersal distance. Assuming that wildlife receptors are unlikely to disperse beyond some distance from their natal site, dispersal distance can be thought of as the radius of the assessment population's boundaries. This general relationship is useful for estimating assessment population areas for terrestrial animals and accounts for wildlife without direct measurements of dispersal. Sediment EcoPRGs are recommended to be calculated on a site-specific basis. This document presents an approach based on the triad for developing sediment EcoPRGs as well as an example application of this approach.

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Acronyms and Abbreviations

ADEM	Associate Directorate for Environmental Programs
AE	assessment endpoint
AOC	area of concern
ASTM	American Society for Testing and Materials
AUF	area use factor
BAF	bioaccumulation factor
CMI	corrective measures implementation
COPC	chemical of potential concern
COPEC	chemical of potential ecological concern
DDT	dichlorodiphenyltrichloroethane
DDX	DDT plus metabolites
EC	effect concentration
EcoPRG	ecological preliminary remediation goal
Eco-SCV	ecological soil cleanup value
Eco-SSL	ecological soil screening level
EPA	Environmental Protection Agency (U.S.)
ESL	ecological screening level
HARP	High-Angle Remediation Project
HE	high explosives
HI	hazard index
HMX	octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine
HQ	hazard quotient
HR	home range
Laboratory	Los Alamos National Laboratory
LANL	Los Alamos National Laboratory
L-ESL	LOAEL-based ESL
LOAEL	lowest observed adverse effect level
LOEC	lowest observed effect concentration
NEBA	Net Environmental Benefits Analysis
NOAEL	no observed adverse effect level
NOEC	no observed effect concentration
PAH	polycyclic aromatic hydrocarbon
PAUF	population area use factor
PCB	polychlorinated biphenyl
PETN	pentaerythritol tetranitrate
PNEC	predicted no effect concentration
PRG	preliminary remediation goal

RDX	hexahydro-1,3,5-trinitro-1,3,5-triazine
SCI	Stream Condition Index
SCIR	Sandia Canyon Investigation Report
SERF	Sanitary Effluent Reclamation Facility
SSD	species sensitivity distribution
SLERA	screening-level ecological risk assessment
SWMU	solid waste management unit
T&E	threatened and endangered
TA	technical area
TAL	target analyte list
tBLM	terrestrial biotic ligand model
TF	transfer factor
TRV	toxicity reference value
UCL	upper confidence limit
WOE	weight of evidence

1.0 INTRODUCTION

Screening-level ecological risk assessments typically make a series of protective assumptions regarding exposure and toxicity of chemicals. The protective assumptions used in screening are generally not characteristic of realistic wildlife population exposure or reflective of population toxicant susceptibility. U.S. Environmental Protection Agency (EPA) guidance recommends evaluating ecological effects at the population rather than at the individual level (EPA 1999, 070086), except when evaluating threatened and endangered (T&E) species. If the EPA risk management goal of maintaining healthy populations of ecological receptors is considered (EPA 1999, 070086), ecological screening thresholds resulting from such protective assumptions are inappropriate for determining cleanup goals (EPA 2003, 076077).

Efroymson et al. (1997, 070825) and Suter et al. (2000, 073480) note that ecological cleanup goals generally correspond to chemical concentrations expected to cause minimal effects on populations and communities. Population-based ecological preliminary remediation goals (EcoPRGs) are appropriate unless remediation activities are needed to address potential for adverse effects on T&E species. Methods to assess T&E species should be as protective as possible because small numbers of individuals may represent a significant portion of the population.

This document supplements the "Screening-Level Ecological Risk Assessment Methods, Revision 5" (hereafter, SLERA Rev. 5) (LANL 2017, 602649). As referenced in SLERA Rev. 5, both New Mexico State and federal regulatory guidance are available for conducting ecological risk assessments. As discussed in section 4.4 of SLERA Rev. 5, cleanup may be warranted if an ecological risk assessment indicates unacceptable risks from site contamination. In such situations, corrective actions would benefit from information on concentrations of chemicals in environmental media that are likely to be protective of ecological receptors (Efroymson et al. 1997, 070825; Suter et al. 2000, 073480; Greenberg et al. 2014, 259132). These concentrations can be considered EcoPRGs. In contrast, ecological screening levels (ESLs) calculated using the methods presented in SLERA Rev. 5 are not cleanup values, just as screening values developed by EPA are not cleanup values because of the conservative and overly protective assumptions used (Wentsel and Fairbrother 2014, 259136). An approach for developing soil EcoPRGs was developed in 2004 with the information available at that time (Ryti et al. 2004, 600901); however, more information has since become available from Los Alamos National Laboratory (the Laboratory or LANL) site-specific studies as well as emerging guidance from the United States and Europe on methods for estimating more realistic adverse effect levels for a variety of ecological receptors, contaminants, media, and exposure pathways. A variety of such approaches has been reviewed to recommend methods to calculate inorganic chemical ecological soil cleanup values (Wentsel and Fairbrother 2014, 259136).

Highlights of EcoPRG Approach

- Impact assessments for plants and soil invertebrates use literature toxicity studies and site-specific bioassays.
- Ecological cleanup goals are based on area-use factors for wildlife populations.
- EcoPRGs are protective of T&E receptors and receptor populations.
- The approach is designed to avoid unnecessary habitat destruction associated with overly protective remediation goals.
- EcoPRGs are not developed for water because water-quality standards and criteria are available for this medium.

The reason for developing EcoPRGs is to provide risk managers with a tool to determine if remediation would mitigate potential adverse effects on the environment without using overly conservative thresholds that lead to unnecessary cleanups. The EcoPRG methodology builds upon the protocol for Laboratory screening-level ecological risk assessments (LANL 2017, 602649) and uses site-specific studies

(bioassays, bioaccumulation) for plants, soil invertebrates, and wildlife as well as ecological information that can be used to define assessment population areas. Thus, the approach provides more realistic exposure and effects information and a spatial basis for assessing adverse effects on populations. Having EcoPRGs does not automatically trigger cleanup if site concentrations exceed these values. Risk managers can conduct empirical studies to verify that cleanup is warranted or evaluate the remedial alternatives based on EcoPRGs as the cleanup level. Empirical studies may include bioassays or direct ecological measurements and site observations of the flora and fauna within the contaminated site or area. If site observations are used as a line of evidence, an evaluation should be provided that includes the spatial distribution of contamination relative to available habitat and that correlates the observations with the contamination.

EcoPRGs can be used to estimate a potential for adverse ecological effects from contaminated soil or sediment (Efroymson et al. 1997, 070825; Fuji et al. 2000, 076076; Suter et al. 2000, 073480; Wentsel and Fairbrother 2014, 259136). Laboratory ecological assessments build upon the conceptual site model and the ecological scoping site visit that document the exposure pathways and receptors present at the site. Geologic material designated as soil has terrestrial receptors and pathways, but not all material designated as sediment has aquatic receptors and pathways. In some cases, terrestrial receptors and pathways are appropriate if the sediment is associated with a dry, nonaquatic system (e.g., a drainage or non-floodplain area). Sediment ESLs and lowest observed adverse effect level– (LOAEL-) based ESLs (L-ESLs) are applied where aquatic receptors and pathways exist, and sediment EcoPRGs are also applied to the locations.

The soil methodology builds upon information important to terrestrial wildlife evaluations, the Laboratory soil exposure model (LANL 2017, 602649), and the use of terrestrial wildlife receptors. The Laboratory approach to ecological risk screening (LANL 2017, 602649) employs hazard quotients (HQs) (i.e., ratios of exposure concentrations to safety thresholds) and is similar in several respects to EPA methodology for calculating wildlife ecological soil screening levels (Eco-SSLs) (EPA 2003, 076077) or the proposed wildlife ecological soil cleanup value approach (Wentsel and Fairbrother 2014, 259136). Receptors or endpoint species are representative in terms of the local environment as well as key feeding strategies, taxonomic groups, and exposure pathways.

Sediment EcoPRGs evaluate multimedia exposures because they are associated with aquatic communities and pathways where flowing or standing water as well as solid medium is available. In addition to the multimedia assessment of sediment and water, sediment EcoPRGs should also evaluate the potential for transport to downstream locations and potential for impacts on receptors at these locations. While there are cases where soil has a large potential to move off-site (e.g., steep hillslopes), in general, downstream transport is a more relevant concern for fluvial sediment. Given the need to consider multimedia exposures as well as potential for transport, sediment EcoPRGs should be developed on a case-by-case basis to reflect site-specific conditions. It is likely that biological monitoring would be useful in refining the need for sediment EcoPRGs and also in calculating appropriately protective sediment EcoPRGs.

This document presents the Laboratory's methods and approaches for developing soil and sediment EcoPRGs. Underlying assumptions are (1) that terrestrial species are the relevant ecological receptors for soil and some sediment conditions, (2) that off-site migration from soil to riparian and aquatic locations is either not occurring or has minimal impact on effects to downgradient species, and (3) riparian and aquatic species are the most relevant receptors in the approach for sediment.

2.0 SOIL ECOPRG METHODS

As discussed above, the development of soil EcoPRGs builds upon the original EcoPRG approach (Ryti et al. 2004, 600901) but has been updated with additional site-specific information and new approaches for evaluating soil ecotoxicity. This new information and its application to the development of Laboratory soil EcoPRGs are discussed below.

- Canyons field studies
    - ❖ Bioassays and other information from site-specific field studies and their application to EcoPRGs are discussed in detail in section 4. Information from Laboratory site-specific studies has been included in the development of EcoPRGs.
- Ecological soil cleanup value (Eco-SCV) approach for metals (inorganic chemicals). See overview papers by Wentsel and Fairbrother (2014, 259136) and Greenberg et al. (2014, 259132).
    - ❖ Plants and soil invertebrates (Checkai et al. 2014, 259131). The recommended approach in Checkai et al., species sensitivity distribution (SSD), maximizes the use of existing studies [and potentially includes a broader list of receptors to include microbial processes (Kuperman et al. 2014, 259133)]. The SSD approach is not relevant to EcoPRG development because dose-response information is not available for multiple species for Laboratory chemicals of potential concern (COPCs). Instead, the geometric mean of adverse effect level is used for EcoPRG development if three or more primary toxicity studies were available (LANL 2014, 252936). Although terrestrial biotic ligand models (tBLMs) are not recommended, empirical relationships of toxicity with soil properties (e.g., cation exchange capacity) were used when available. The tBLM is included as a source of refined toxicity information for plant and soil invertebrate EcoPRG development. An added component for inorganic chemicals is a comparison to background concentrations.
    - ❖ Wildlife toxicity reference values (TRVs) (Mayfield et al. 2014, 259134). Mayfield et al. made several recommendations concerning wildlife TRVs. One recommendation was to better refine species of concern and use TRVs specifically applicable to those species. If available, critical body or tissue residue TRVs should be used. Mayfield et al. also recommended developing TRVs for both acute and chronic exposures. Another recommendation was to consider biological significance and not only statistical significance (using effective dose for 5% to 20% effect rather than no observed adverse effect levels [NOAELs]). A final recommendation was to integrate field observations into the assessment as another line of evidence. For site-specific ecological risk assessments, Laboratory field studies are used as lines of evidence, and for EcoPRG development those field studies were considered a source of effects and exposure information. The literature ecotoxicity studies for the COPCs considered for the EcoPRG development generally do not have dose-response information, so a quantitative assessment of various effect levels is not practical.
    - ❖ Wildlife exposure (Sample et al. 2014, 259135). Recommendations from Sample et al. include the following: (1) collect additional data or apply adaptive management approaches if Eco-SCVs are calculated that are less than background because such values are unrealistic; (2) use site-specific data for incidental soil ingestion or dietary pathways; (3) understand the chemical form of the inorganic chemical to improve the exposure evaluation; (4) consider using tools and models for estimating the bioaccessibility and bioavailability of metals; and (5) apply the wildlife Eco-SCVs to the approach's spatial scale for individuals or the assessment population. Site-specific

information on bioaccessibility and bioavailability is not available for the Laboratory. For wildlife, spatial scale of the site relative to the assessment population area is considered.

- A tiered approach for soil EcoPRGs was developed for the Hanford Site, an arid site in eastern Washington State with similar ecosystems, pathways, and receptors to those of the Laboratory [CHPRC-00784, Revision 1, Tier 1 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site (CHPRC 2014, 261847)].
    - ❖ Generic screening values and Tier 1 wildlife EcoPRGs include published EPA or Washington State values and values calculated for Hanford Site receptors (CHPRC 2014, 261847). The Tier 1 wildlife EcoPRGs are developed for representative receptors at the Hanford Site with the use of either NOAELs or LOAELs as TRVs and are thus similar to the Laboratory ESLs and L-ESLs.
    - ❖ Tier 2 wildlife EcoPRGs use semi-site-specific information on bioaccumulation from soil into food, as well as LOAELs, as TRVs [CHPRC-01311, Revision 2, Tier 2 Risk-Based Soil Concentrations Protective of Ecological Receptors at the Hanford Site (CHPRC 2014, 261848)]. In some cases, the TRV is the geometric mean from available toxicity studies, although more typically it is a critical study because the number of studies (three or more) to calculate the geometric mean was not available. This approach is similar to what has been used for Laboratory EcoPRG development.
    - ❖ Tier 2 plant and soil invertebrate EcoPRGs use site-specific bioassays [ECF-Hanford-11-0158, Revision 1, Tier 2 Terrestrial Plant and Invertebrate Preliminary Remediation Goals (PRGs) for Nonradionuclides for Use at the Hanford Site (CHPRC 2014, 261849)]. Statistical analyses of these bioassays (seedling germination and collembolan) resulted in site-specific no observed effect concentrations (NOECs) to use as EcoPRGs. The Laboratory EcoPRGs for plants and soil invertebrates also make use of site-specific bioassays as discussed in detail in sections 4 and 5.

The remainder of this section describes receptors, COPCs, plant and soil invertebrate EcoPRGs, and wildlife exposure and EcoPRG calculations.

2.1 List of Receptors for Derivation of Soil EcoPRGs

The receptors considered for soil EcoPRGs started with those listed in SLERA Rev. 5 (LANL 2017, 602649). These receptors were selected to represent the variety of environments and feeding guilds present at the Laboratory, which has diverse land covers such as mixed conifer, ponderosa pine, piñon-juniper, and grassland across a vertical mile of elevation from the Rio Grande to the Jemez Mountains. More information on the environments present at the Laboratory, and the basis for selecting screening receptors is presented in SLERA Rev. 5 (LANL 2017, 602649). The screening and EcoPRG receptors are identical, with the exception of the inclusion of the Mexican spotted owl (rather than the American kestrel) as the top avian carnivore for the EcoPRG receptor (Table 2.1-1). The Mexican spotted owl was selected because it is the T&E species most often evaluated at Laboratory solid waste management units (SWMUs) and areas of concern (AOCs).

2.2 List of COPCs for Derivation of Soil EcoPRGs

The COPCs for derivation of soil EcoPRGs were compiled based on the results of previous Laboratory investigations and ecological risk assessments for SWMUs, AOCs, and canyons. The following suites were selected: inorganic chemicals, polychlorinated biphenyls (PCBs), dioxins/furans, phthalates,

polycyclic aromatic hydrocarbons (PAHs), and high explosives (HE). Table 2.2-1 lists the receptors that have ecotoxicity information available for the COPC selected. Sixty-one COPCs are included among these suites:

- Dioxins/furans: 1
- HE: 10
- Inorganic chemicals: 22
- PAHs: 17
- PCBs: 5
- Phthalates: 6

2.3 Plant and Soil Invertebrate EcoPRGs

Although the ESLs for plants and soil invertebrates are primarily obtained from the ecotoxicity literature with little to no calculations involved, the EcoPRGs for these receptors can involve some calculations. Given that multiple sources of information are available for these receptors and COPCs, it is also necessary to develop the logic to select the preferred information source for EcoPRGs.

Predicted no effect concentrations (PNECs) developed by the European Commission (Smolders et al. 2009, 260205) were considered for use as EcoPRGs. Currently, these PNECs are available for cadmium, copper, lead, nickel, and zinc from the PNEC calculator that accounts for soil properties (e.g., cation exchange capacity) and contaminant aging. However, these PNECs were determined to be less than Laboratory background concentrations, ESLs, or both, and do not provide practical EcoPRGs.

The plant and soil invertebrate EcoPRGs are selected as the maximum values from the following sources, if both are available:

- The effect concentration (EC) associated with a particular response (ECx). If an ECx cannot be calculated, then a site-specific NOEC was used. Such values are derived from the Laboratory-specific bioassays discussed in section 4.
- Geometric mean lowest adverse effect concentrations (LOECs) from the Laboratory ECORISK Database Release 4.1 (LANL 2017, 602538) or the most current version.

There may also be cases where the site-specific NOEC from the bioassays does not represent a biologically significant concentration. If no literature LOEC or site-specific ECx is available, then the site-specific NOEC is actually an unbounded value (meaning effects are possible at some unknown, higher concentration). In this case, it may not be appropriate to select this site-specific NOEC as an EcoPRG. However, the existing bioassays may be useful in planning additional bioassay studies with the site soil to determine a site-specific ECx from the dose-response relationship. Alternatively, a qualitative assessment of the habitat (e.g., type and amount of vegetative cover at the site compared with the surrounding habitat) may be performed. The assessment should consider the impacts of physical disturbance as well as the potential impacts of COPCs and any evidence for recovery. This habitat assessment should also include information (e.g., absence of biota associated with visual evidence of contamination) to help document the potential for impacts on populations of plants and/or soil invertebrates.

2.4 Wildlife Receptor Exposure Parameters and EcoPRG Calculations

The SLERA Rev. 5 provides an approach for calculating ESLs for wildlife receptors based on relevant ingestion exposure pathways (incidental ingestion of COPCs in soil plus ingestion of COPCs in food (LANL 2017, 602649). These exposure pathways were also considered in the development of the EPA's Eco-SSLs (EPA 2003, 076077). Other pathways from soil, such as exposure to vapors or particulates in air and dermal contact, are generally less important for terrestrial wildlife (EPA 2003, 076077). Risk associated with exposure through water pathways to terrestrial wildlife should be evaluated on a site-specific basis if contaminated water occurs at the site. EcoPRGs are calculated with the use of HQ methods (Equation 1) modified to account for adverse effects on populations of wildlife receptors or the more sensitive members of T&E species.

$$HQ_{ij} = Exposure_{ij} / TRV_{ij} \qquad \textbf{Equation 1}$$

where

HQ_{ij} = soil HQ for receptor *i* and COPC *j* (unitless)

$Exposure_{ij}$ = exposure dose for receptor *i* and COPC *j* (mg-COPC/kg-body weight/d)

TRV_{ij} toxicity reference value for receptor *i* and COPC *j* (mg-COPC/kg-body weight/d)

The ESLs are calculated with toxicity and exposure parameters designed to be protective of the more sensitive members of a receptor population (LANL 2017, 602649) (e.g., generally using empirical upper-bound estimates of food ingestion). Because central tendency estimates of ingestion rates are more appropriate for evaluating adverse effects on populations, the mean ingestion rates and mean body weights (EPA 1993, 059384) were incorporated in the calculation of EcoPRGs (Table 2.4-1). Where multiple studies were available, they were averaged based on the means reported for each study under the assumption that body weights might vary by location (habitat) and time (year or season). All the ESL food intake rates, except for the mountain cottontail, were empirical or measured food ingestion rates. Such studies are generally good upper-bound estimates of intake but are not reflective of either long-term or population average exposures. The paper by Nagy (2001, 253420) provides allometric equations of food ingestion based on body weight for mammal and bird diets (herbivore, omnivore, insectivore, or carnivore). The mean body weights were used for the representative receptors in these allometric equations (Table 2.4-1). There was no herbivore equation for birds, so the equation for "all birds" was used for the robin herbivore diet. With the exception of the deer mouse and shrew body weights, which are based on Laboratory field studies, the body weight information for other receptors is from the available literature and is broadly applicable to Laboratory sites. The deer mouse and shrew body weight data should be directly applicable to other Laboratory locations where these receptors are found and EcoPRGs may be required.

Considering exposure modeling to the soil EcoPRG T&E receptor, the Mexican spotted owl has exceptionally low energy requirements (Weathers et al. 2001, 073476) relative to other birds of prey and, consequently, the allometric-based intake might be overly protective for calculations of contaminant exposure. Empirical data were used to calculate more representative food ingestion rates for the Mexican spotted owl (Table 2.4-1). Note that for the calculation of ESLs, a variant of the kestrel with a 100% flesh diet was used; because the Mexican spotted owl is evaluated explicitly for the EcoPRGs, this additional diet variant of the kestrel was not necessary.

The EcoPRG TRVs presented in this document are obtained from information in the Laboratory ECORISK Database Release 4.1 (LANL 2017, 602538). (Note that if the database is subsequently revised, the TRVs in the most current version should be used.) TRVs were selected based on a variety of adverse effects, including differences in growth, survival, or reproduction. The database includes TRVs

based on EPA Eco-SSL reviews, in-depth reviews conducted by the Laboratory, and some secondary sources of ecotoxicity information. Where there are sufficient studies, the EcoPRG TRV is the geometric mean of available receptor- and analyte-specific NOAELs and associated LOAELs. Because Eco-SSLs are intended to be protective of rare, endangered, and threatened species, the geometric mean of NOAELs is appropriate for protecting more sensitive individuals (EPA 2003, 076077). To be protective of wildlife populations, the geometric mean of LOAELs for the EcoPRG was selected.

The default bioaccumulation models for EcoPRGs are also taken from the Laboratory's ECORISK Database Release 4.1 (LANL 2017, 602538) or the most current version. Bioaccumulation models or transfer factors may also be obtained from Laboratory site-specific studies (e.g., Podolsky 2000, 073477, or the published scientific literature).

EcoPRG derivations protective of wildlife populations and individuals of T&E species (Equations 2 and 3, respectively) are presented as general models that include herbivores, omnivores, invertivores, and carnivores.

$$EcoPRG_{ij} = \frac{TRV_{ij}}{I_i \cdot PAUF_i \cdot [fs_i + fp_i \times TF_{plant,j} + fi_i \times TF_{invert,j} + ff_i \times TF_{flesh,j}]} \qquad \textbf{Equation 2}$$

where

$EcoPRG_{ij}$ = soil EcoPRG for wildlife receptor *i* and COPC *j* (mg/kg)

TRV_{ij} = toxicity reference value (geometric mean LOAEL) for wildlife receptor *i* and COPC *j* (mg-COPC/kg-body weight/d)

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

$PAUF_i$ = population area use factor for wildlife receptor *i*

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

fp_i = fraction of plants in diet for wildlife receptor *i;* expressed as a fraction of the dietary intake

$TF_{plant,j}$ = transfer factor from soil to plant for COPC *j* (mg/kg-plant dry weight per mg/kg-soil dry weight)

fi_i = fraction of invertebrates ingested by wildlife receptor *i*, expressed as a fraction of the dietary intake

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

ff_i = fraction of flesh ingested by wildlife receptor *i*, expressed as a fraction of the dietary intake

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

$$EcoPRG_{ij} = \frac{TRV_{ij}}{I_i \cdot AUF_i \cdot [fs_i + ff_i \times TF_{flesh,j}]}$$

Equation 3

where

$EcoPRG_{ij}$ = soil EcoPRG for wildlife T&E receptor *i* and COPC *j* (mg/kg)

TRV_{ij} = toxicity reference value (geometric mean NOAEL) for wildlife T&E receptor *i* and COPC *j* (mg-COPC/kg-body weight/d)

I_i = normalized daily dietary ingestion rate for wildlife T&E receptor *i* (kg-food dry weight/kg-body weight/d)

AUF_i = area use factor for wildlife T&E receptor *i*

fs_i = fraction of soil ingested by wildlife T&E receptor *i*, expressed as a fraction of the dietary intake

ff_i = fraction of flesh ingested by wildlife T&E receptor *i*, expressed as a fraction of the dietary intake

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

Instead of parameters selected for wildlife populations (Equation 2), the parameters described above (Equation 3) are representative of a T&E receptor (e.g., I_i = normalized daily dietary upper-bound ingestion rate for T&E receptor *i*) and AUF_i (area use factor for T&E receptor *i*) replaces $PAUF_i$. The diet for the Mexican spotted owl is assumed to be 100% flesh (carnivore). The Mexican spotted owl food intake (I_i) is provided in Table 2.4-1 and its fraction of soil ingested (fs_i) is 0.02 or the same value used for the kestrel. The Mexican spotted owl AUF is calculated based on a home range (HR) of 545 ha (see discussion below).

HR Information. Unless otherwise indicated, data were compiled from EPA's Wildlife Exposure Factors Handbook (EPA 1993, 059384). The HR data in Table 2.4-1 were used to calculate population areas and EcoPRGs for all land cover types by accounting for the area potentially occupied by an individual or population relative to a site's areal extent. A site-specific HR can be developed to apply an HR to a SWMU/AOC for a species with information across several land covers. Justification for selecting HR data is provided below.

Mountain cottontail. The eastern cottontail is used as a surrogate for the mountain cottontail and the HR data are from Wisconsin woodlots over several seasons, ranging from 0.8 to 4 ha, along with data from a mixed habitat in Pennsylvania, ranging from 1.5 to 7.8 ha. All HRs were averaged (EPA 1993, 059384, p. 2-357), which results in a cottontail HR of 3.1 ha.

Deer mouse. Minimum HR data from Carlsen et al. (2004, 601149) for *Peromyscus maniculatus* are 0.16 and 0.63 ha. The average of these two values, 0.4 ha, is used as the HR to develop EcoPRGs. Information provided in the supporting appendix of Carlsen et al. (2004, 601149) was used because it represented a recent compilation of receptor spatial information.

Shrew. The short-tailed shrew is used as a surrogate for the montane shrew and the HR data are for males and females in a Michigan bluegrass environment (<0.1 to 0.36 ha female, <0.1 to 1.8 ha male) and for an old field in New York State in periods of high prey abundance (0.03 to 0.07 ha) and low prey abundance (0.1 to 0.2 ha). The mean HR data for all seasons is from a tamarack bog in Manitoba, Canada (0.39 ha [EPA 1993, 059384, p. 2-214]). Although a bog environment is uncharacteristic for much

of the Pajarito Plateau, there are uncertainties with deriving a point estimate from a range without prior knowledge of the data distribution. Consequently, the average HR of 0.39 ha was selected for the shrew.

Fox. Average HR data are for male and female foxes in alpine and subalpine British Columbia (from 1137 to 1967 ha), a variety of Minnesota settings (mean of 699 ha), and diverse environments in Wisconsin (from 96 to 717 ha). The average of the HRs is 1038 ha (EPA 1993, 059384, p. 2-226).

American robin. The average territory size over a campus setting in Tennessee (0.42 ha, with a range of 0.12 to 0.84 ha [EPA 1993, 059384, p. 2-199]) was selected instead of HR data for dense coniferous and unspecified forests in New York because the robin prefers open habitat.

American kestrel. Information on average territory sizes for the American kestrel is from California open areas (31.6 ha) and woods (13.1 ha), an agricultural area in Illinois (154 ha), Wyoming grasslands and forests (202 ha), and Michigan woodlots and fields (131 ha). The average of the values is 106 ha (EPA 1993, 059384, p. 2-114).

Mexican spotted owl. The mean breeding HR for the Mexican spotted owl is 545 ha for the canyon lands of Utah (Willey and Van Riper 2007, 601151). This study was selected based on its similarity to the Los Alamos environment (elevation range of 1500 to 2445 m and annual precipitation of 17 cm per year).

Area Use Factors (AUFs). AUFs are calculated as the ratio of the site area to the receptor's HR (EPA 2003, 076077). Individual AUFs and population area use factors (PAUFs) are appropriate to modify the estimate of risk to wildlife receptors. The introduction of area use reduces potential overestimation of risks to receptors whose HRs are larger than the area of contamination being evaluated. These AUFs/PAUFs may be applied to either individual organisms or populations.

EPA guidance recommends evaluating ecological effects at the population rather than at the individual level (EPA 1999, 070086), except when evaluating T&E species. Screening with ESLs generates HQs and hazard indexes (HIs) designed to estimate the potential for risk to individual ecological receptors, assuming continuous exposure to the representative concentration of the COPC in question. The AUF is calculated based on the ratio of the site area to the HR of an individual receptor to reflect the fact that a receptor actually moves around its HR and does not remain stationary in only the site area. Therefore, the individual AUF assesses the level of individual exposure based on the area of the HR. The modification of the EcoPRG with a PAUF uses the estimated area occupied by the population of a receptor species to assess the likelihood of any individual within the assessment population encountering the site. The PAUF assumes impacts to some individuals and estimates the average effect on the assessment population of that impact. Therefore, the wildlife EcoPRGs incorporate an evaluation of the potential for adverse effects on an assessment population.

PAUFs are developed based on investigations correlating the HR of a receptor with its dispersal distance (the distance an animal moves from its natal HR). The dispersal distance has been shown to affect population structure, demographics, and spacing patterns and can be used to determine the assessment population boundaries (Bowman et al. 2002, 073475). When HR is expressed as its linear dimension (the square root of HR), it has a good linear correlation with dispersal distance for the same species (Bowman et al. 2002, 073745). For mammals with similar HR sizes to the species used as screening receptors at the Laboratory, dispersal distance is equal to 3.5 times the square root of the HR. The relationship holds well for small mammals, such as mice and rabbits, but may overpredict dispersal distance for fossorial species and slightly underpredict dispersal distance for some large herbivores such as the white-tailed deer (Ryti et al. 2004, 600901). The mathematical relationship between HR and dispersal distance has been estimated only for mammals, but for the calculations at these sites, the same methodology was applied to avian receptors. Bird species have higher median and maximum dispersal

distances than similar-sized mammals (Sutherland et al. 2000, 073460; available at http://www.consecol.org/vol4/iss1/art16/index.html), so application of the mammalian relationship is protective of bird species because this relationship underestimates the dispersal distance and, therefore, the avian assessment population area. Using only spatial relationships to establish exposures for bird populations is also protective because most bird species inhabit the Pajarito Plateau seasonally and therefore are exposed for only part of the year. Birds selected as representative receptors for ESLs and EcoPRGs spend a large fraction of the year on the Pajarito Plateau and the TRVs used for exposure are based on daily exposure. Thus, average daily exposure is compared with the TRV for the purpose of determining the potential for adverse effects. For the purposes of classifying studies for wildlife exposure, 91 days or more is considered to represent chronic exposures (LANL 2014, 252936).

The dispersal distance from the center of the HR can be considered the radius of the animal's population area, with the area likely to be occupied by members of that population (the assessment population area) consisting of the circle described by the area covered by the dispersal distance. The assessment population area would therefore be equal to πr^2, which would be equal to π times (3.5 times the square root of the HR)^2. This mathematical relationship can be simplified to 40 times the HR as a representation of the assessment population area in hectares (Ryti et al. 2004, 600901). Once the population area is calculated for each receptor of interest, the area of the site can be divided by the population area to develop a site-specific PAUF for that population. HRs and population areas (40 times HRs) for the non-T&E wildlife EcoPRG receptors are presented in Table 2.4-2. A population area of 16 ha is used to be protective of middle trophic level wildlife, but a population area of at least 23,000 ha is used to be protective of upper trophic level wildlife. The middle trophic level population area (16 ha) was selected because the population areas for three of the four middle trophic level receptors (robin, deer mouse, and shrew) have population areas of 15 to 17 ha. The upper trophic level population area is the median and mean of the population areas for the two receptors (kestrel and fox). Potential impacts on T&E species are evaluated at an intermediate scale (the 545-ha HR for the Mexican spotted owl). Given these scales, the relative ingestion rates, and potential for bioaccumulation into various food items, it is likely that middle trophic wildlife populations will present the limiting wildlife EcoPRG for most, if not all, COPCs.

3.0 SEDIMENT ECOPRG METHODS

As discussed in SLERA Rev. 5 (LANL 2017, 602649, section 3.4.3, p. 36), sediment has a geological definition based on its being in a fluvial geomorphic setting. In the context of ecological exposure pathways, terrestrial is the sole or dominant pathway for the majority of fluvial sediment. For sediment with terrestrial receptors and pathways, the soil EcoPRGs are most relevant. Only sediment with persistent water has aquatic receptors and pathways, either directly to sediment-dwelling biota or indirectly via emergent insects to aerial insectivores. Sediment EcoPRGs are developed for locations with aquatic receptors and pathways.

Aquatic environments are not common at the Laboratory. The land-cover categories of Open Water and Aspen-Riparian-Wetland represent around 2% of the Laboratory's area (LANL 2015, 600982, Table 2.1-1, p. 7). Therefore, wetlands comprise a small but important component of the landscape. This means 98% of land area is terrestrial and the soil EcoPRGs are potentially applicable. The vast majority of the aquatic environments are located in the Laboratory's canyons, which have been extensively studied for potential ecological risks. To provide a context for sediment EcoPRGs, this document summarizes the relevant risk assessments, presents the approach for developing sediment EcoPRGs with a site-specific example, and provides some considerations for risk management.

3.1 Summary of Relevant Risk Assessments

Ecological risk assessments have been completed for the aquatic environments in four watersheds: Los Alamos and Pueblo, Mortandad, Pajarito, and Sandia (LANL 2004, 087390; LANL 2005, 089308; LANL 2006, 094161; LANL 2006, 093553; LANL 2009, 106939; LANL 2009, 107453). Chemicals of potential ecological concern (COPECs) were identified from the list of COPCs for each watershed based on comparisons of concentrations with ESLs. In addition, the Phase III RFI Report for SWMU 16-021(c)-99 (LANL 2003, 077965) addressed baseline human health and ecological risks in Cañon de Valle (part of the Water Canyon watershed) with an emphasis on risks associated with the area proximal to the 260 Outfall. No risks were noted to terrestrial receptors (bat and swallow) that feed on emergent insects. The ecological risk assessments summary statements related to the aquatic community are presented verbatim below.

Los Alamos and Pueblo Canyons (LANL 2004, 087390, section 8.1.4.1, pp. 8−30). "Results from toxicity tests [(*Chironomus tentans*)] and field surveys of macroinvertebrate abundance and diversity indicate no potential for adverse effects, which contradicts the screening level ecological risk assessment result indicating the potential for adverse ecological effects. There were no decreases in chironomid growth or survival compared with reference locations. Field surveys of macroinvertebrate abundance and diversity documented an impoverished fauna, which was related to the quality of the aquatic habitat in these canyons. Measures of chironomid deformity could not be correlated to contaminant concentrations with the exception of the DDX [(dichlorodiphenyltrichloroethane [DDT] plus metabolites)] toxic score (concentration normalized to organic matter), which indicates that contaminant levels are not high enough to yield a statistical predictor of exposure. Thus, the weight of evidence [(WOE)] from the two measures of effect and the measure of exposure indicates no adverse effects from COPECs on abundance or diversity of aquatic organisms in the more persistently wet segments of the Los Alamos and Pueblo watershed."

Mortandad Canyon (LANL 2006, 094161, section 8.1.4.1, p. 123). "The Laboratory toxicity test using *C. tentans* showed no difference in survival and no difference in growth correlated with COPEC concentration. The field bioassessment characterization indicated that chironomids dominate the aquatic community in sampled reaches and that the toxicity test using chironomids is therefore an appropriate measure of impacts to the aquatic community. The Laboratory algal toxicity test showed differences in cell growth with reaches, but these differences were attributable to water hardness and not to COPECs in water. The WOE for measures of the aquatic community indicates there are no adverse effects from COPECs in sediment and water on abundance and survival of the aquatic community in the reaches of the Mortandad watershed."

Pajarito Canyon (LANL 2009, 106939, section 8.1.4.1, p. 83). "The laboratory toxicity test using *C. tentans* showed no difference in survival and no difference in growth correlated with COPEC concentration. The field bioassessment characterization indicated that chironomids dominate the aquatic community in one of the sampled reaches and that the toxicity test using chironomids is an appropriate measure of impacts to the aquatic community. The WOE for measures of the aquatic community indicates there are no adverse effects from COPECs in sediment and water on abundance and survival of the aquatic community in the reaches of the Pajarito watershed."

Sandia Canyon (LANL 2009, 107453, section 8.1.4.1, pp. 101−102). "The laboratory toxicity test using *C. tentans* showed differences in growth and no difference in survival compared to reference site results. Both survival and growth were negatively correlated with some sediment COPEC concentrations (PCBs, barium, chromium, copper, cyanide, iron, lead, mercury, perchlorate, selenium, and silver) and with possible confounding factors for *C. tentans* growth and survival. The field bioassessment characterization indicated that chironomids are present in the aquatic community in one of the sampled reaches and that the toxicity test using chironomids is an appropriate measure of impacts to the aquatic community. The

field bioassessment indicated that the reaches S-2 and S-3E were impaired, [with impairment] possibly associated with physical conditions, water quality (runoff from developed areas), and contamination in sediments. The WOE for measures of the aquatic community indicates adverse effects from COPECs in sediment and water on abundance and survival of the aquatic community in the Sandia Canyon reaches."

Cañon de Valle (LANL 2003, 077965, Appendix L, section 11, pp. L-48−L-49). "The ecological assessment of the aquatic system in the canyon found some differences between benthic macro-invertebrates in Cañon de Valle and reference canyons. These differences were attributed to relative sizes of the streams (with Cañon de Valle being the smallest), reduced flows caused by the ongoing drought, and the elimination of effluent discharges to the canyon. One of the two rounds of toxicity testing using *C. tentans* for sediment and water in the canyon identified reduced survival for a site near the 260 [O]utfall and a site below Burning Ground Spring. These results were not replicated in a subsequent toxicity test. The presence of a viable benthic macroinvertebrate community in the canyon indicates that the reduced survival in the 2001 toxicity test for the site near the 260 outfall is not a spatially extensive condition. The lack of difference between that same site and the reference site in the 2002 toxicity testing further indicates that large-scale pervasive impacts to the aquatic system are not occurring. The benthic macroinvertebrate community is considered a more meaningful measure of the condition of the aquatic system in the canyon than the toxicity testing results. While toxicity testing identifies potential problems based upon the sampling locations and can be used to associate contaminant concentrations with measured effects for the samples, the endemic community condition gives a much larger scale indication of contaminant impacts that are integrated over long periods."

Cañon de Valle Supplemental Toxicity Testing (LANL 2010, 110508, p. 5). "Results of previous investigations indicated the need for further testing of toxicity at the SWSC Cut. The 2009–2010 CMI [corrective measures implementation] investigation and remediation activities included collecting sediment samples from the SWSC Cut area and submitting them for TAL [target analyte list] metal analysis [and toxicity testing]…. No significant reductions of *C. tentans* survival or growth occurred in the SWSC Cut sediment." The preceding statements led to the conclusion that there are no adverse effects of COPECs in sediment on chironomid survival and growth. These studies were conducted after the CMI source removal actions, and the 2003 report presented information on the conditions before the CMI.

3.2 Approaches for Developing Sediment EcoPRGs

The Laboratory uses an ecological designation for sediment for the purposes of established exposure pathways and receptors. In most situations, material designated as geological sediment has no aquatic receptors and is more appropriately evaluated for terrestrial receptors. Therefore, sediment EcoPRGs are more complex than soil EcoPRGs, primarily because sediment EcoPRGs evaluate multimedia exposures ((LANL 2017, 602649, Table 3.4-1, pp. 32−33). SLERA Rev. 5 states, "Because of the typical association of sediment with water, application of sediment ESLs leads to an incomplete evaluation of the potential ecological effects associated with contaminated sediment/water settings. Thus, surface water and multimedia exposure assessments are required in all cases where contaminated sediment is identified" (LANL 2017, 602649, section 3.4.3, p. 37). Exposure from water should be considered in combination with sediment when developing EcoPRGs. Water EcoPRGs would be water-quality standards or criteria.

In addition to the multimedia assessment of sediment and water, the potential for transport to downstream locations and the potential for impacts on receptors at these locations should also be evaluated. A wealth of information is available on contaminant spatial and temporal trends in the canyon watersheds (Reneau et al. 2004, 093174; Malmon et al. 2005, 093540; "Watershed Monitoring" in LANL 2010, 111232; "Watershed Monitoring" in LANL 2014, 261879). These studies have shown that contaminant concentrations in sediment decrease over time and with distance from the sources.

Given the need for considering possible multimedia exposures as well as potential for transport, sediment EcoPRGs should be developed on a case-by-case basis to reflect site-specific conditions. It is likely that biological monitoring would be useful in refining the need for sediment EcoPRGs and also in determining appropriately protective sediment EcoPRGs.

There are three complementary approaches for developing sediment EcoPRGs in locations with aquatic communities. The first approach is to base EcoPRGs on the geometric mean LOECs if sufficient studies (three or more) are available for a geometric mean. This approach is parallel to the method used to develop soil EcoPRGs. The second approach is to use the dose-response relationship of bioassay measures (e.g., *C. tentans* survival or growth) with COPC concentrations to calculate ECs. The third approach is to use aquatic macroinvertebrate surveys to document current conditions and the need for sediment EcoPRGs and to potentially develop ECs based on exposure-response relationships.

Another option is to integrate these approaches using the Sediment Quality Triad (Chapman 1989, 062902; Bay and Weisberg 2010, 601148). Each of the approaches and the triad approach are discussed below.

Literature-based effects thresholds. The sediment ESLs and L-ESLs are intended to be screening values and are not appropriate as EcoPRGs. Most of the sediment ESLs and L-ESLs are based on secondary literature sources, so there are no detailed records providing additional information on toxicity and effects for these COPCs. A geometric mean of effect levels can be calculated for those COPCs where the primary ecotoxicity literature was reviewed for the ECORISK database. The ecotoxicity information for those COPCs can be modified with literature searches and reviews based on the Laboratory TRV methodology development process (LANL 2014, 252936).

Bioassays. As summarized in the ecological risk assessments (section 3.1), toxicity testing using *C. tentans* or other appropriate test species provides useful site-specific information on the potential for adverse effects and the development of EcoPRGs in aquatic systems. The bioassay provides information on survival and growth of chironomids. If site survival and growth differ from that measured at the reference location, then this difference could be related to COPC concentrations. However, differences in growth or survival may be related to confounding factors (such as nutrients and particle size) as well as COPCs. If a dose-response relationship can be established, then an EC for survival and growth can be calculated. For example, the EC20 for growth would be a 20% reduction in growth from reference levels. As discussed in more detail in section 3.4, risk-management decisions will benefit from calculations of a range of values from EC5 to EC50 or greater.

Field surveys. Aquatic macroinvertebrate surveys are generally very useful as a line of evidence for sediment and water exposures in aquatic systems. They provide site-specific information on the biota present and in some cases can be used in a causal analysis to determine stressors that are acting on the system. At most locations, the habitat is scored and the Stream Condition Index (SCI) is calculated. However, these metrics are not applicable to wetlands. The result of the SCI is one of the following statements: “comparable to reference,” “slightly impaired,” “moderately impaired,” or “severely impaired.” The data behind the SCI can be analyzed with the use of multivariate statistical analyses and can be correlated to COPC concentrations or habitat measures. If an exposure-response relationship can be established, then it may be possible to calculate ECs, as recommended for the bioassays. However, it is more likely that field surveys can be used in a more semiquantitative or qualitative manner. For example, field surveys can be conducted over time to document changes or recovery.

Triad. The Sediment Quality Triad is a simple and established way to combine the results of literature-based effects with bioassays and field studies (Chapman 1989, 062902; Bay and Weisberg 2010, 601148). Both graphical and tabular methods are available for displaying triad information. For example, one may show sun-ray or ternary plots keyed to locations on a map. These plots may show concordance

or discordance with the literature, bioassay, and field measures and any spatial trends relative to contaminant sources. Such plots may depict the magnitude or the confidence in these measures. A tabular display is useful in that more details can be shown for locations or investigation reaches. Such detail could include the magnitude of the effect, the likelihood or persistence of recovery, and the quantitative or qualitative uncertainties related to each measure for each location. Therefore, the triad can be used in a qualitative manner to support development of sediment EcoPRGs.

3.3 Site-Specific Example

Sandia Canyon was selected as the site-specific example because its assessment identified unacceptable risks to aquatic endpoints. There were no unacceptable risks to the southwestern willow flycatcher (assessment endpoint [AE] 6) based on the WOE for the aerial insectivore feeding guild (LANL 2009, 107453, section 8.1.4.1, p. 100). The assessment identified the potential for effects on the abundance and survival of the aquatic community in the reaches of Sandia Canyon that retain surface water long enough to support aquatic communities (AE7). The Sandia Canyon Investigation Report (SCIR) identified both COPECs and confounding factors (including habitat condition) for measures associated with AE7 (decreased growth, survival, abundance, and diversity) (LANL 2009, 107453, section 8.1.4.1, pp. 101–102).

There are several factors to consider with regard to risk characterization and the need for EcoPRGs:

- What is the probability of adverse effects?
- What is the location and extent of contamination (in particular contamination exceeding a threshold)?
- Is the threshold likely to be exceeded in the future?
- What is the expected half-life (qualitative or quantitative) of COPCs, and what is the potential for recovery if the sources are removed?
- What are the sources of uncertainty and how are they propagated through the risk assessment?

These concepts are from step 7 of the Ecological Risk Assessment Guidance for Superfund (EPA 1997, 059370), and the information related to AE7 from the SCIR or other relevant documents is summarized below. The SCIR was prepared in 2009 based on studies completed in 2008, and 9 yr have elapsed (through 2017) since those data were collected. In addition, some site conditions have changed based on mitigation work associated with the chromium groundwater plume since the data were collected.

Likelihood of risk. The SCIR used a WOE approach to characterize risks for each of the AEs. The WOE for AE7 was summarized in the SCIR and included three measures (LANL 2009, 107453, Table 8.1-23). The bioassays using *C. tentans* were a high-weighted measure, and the conclusion from this measure was that differences noted in the bioassays could be related to COPECs or confounding factors (e.g., particle size). Specifically, there was no difference in chironomid survival between Sandia Canyon and the reference location, but Sandia Canyon growth was about 50% of reference location growth (LANL 2009, 107453, Figure 8.1-17 for survival and Figure 8.1-18 for growth). The field studies of aquatic invertebrate diversity and abundance were a medium-weighted measure and indicated impairment potentially related to the physical system, water quality, or sediment COPECs. Finally, the comparison with ESLs was a low-weighted measure and identified study-design COPECs (metals and PCBs) for this AE. Therefore, based on these measures, the risk characterization indicated some impacts to the aquatic community, but these effects could be from COPECs or non-COPEC factors.

Location and extent of contamination. Perennial surface water occurs in upper Sandia Canyon from effluent discharges of treated sanitary wastewater and cooling water from Technical Area 03 (TA-03). This area includes Reaches S-1S, S-2, S-3W, and S-3E (see Figure 8.1-1 in LANL 2009, 107453, for the locations of these investigation reaches). Downstream of the perennial segment, there is ephemeral flow down the remaining extent of Sandia Canyon to the Rio Grande. However, by definition of AE7, the spatial extent of potential concern to the aquatic community is limited to the segment of the canyon with perennial flow. Because the ecological screening conducted for the SCIR used the ECORISK Database, Version 2.3 (LANL 2008, 103352), the data from the report were screened with the ECORISK Database, Version 3.3 (LANL 2015, 600921) screening levels for this exercise. Table 3.3-1 presents the maximum detected active-channel sediment concentrations in the Sandia Canyon reaches with perennial surface water flow, along with the reference reach in Pajarito Canyon [PA-0] for comparison. Maximum concentrations of cadmium, chromium, copper, cyanide, mercury, selenium, silver, zinc, Aroclor-1254, and Aroclor-1260 are greater than the aquatic community L-ESLs, and the highest concentration of each COPEC is in Reach S-2 (the Sandia Canyon wetlands). The maximum concentration of selenium in Reach S-3W is basically equal to the aquatic community L-ESL. Therefore, the extent of contamination greater than the aquatic community L-ESLs is limited to Reach S-2. The total area of Reach S-2 is about 2.3 ha, and the total area of the Sandia Canyon watershed is about 1400 ha. Reach S-2 has a number of geomorphic deposits laterally and with depth. Concentrations of COPECs vary across these geomorphic deposits. However, it is worth noting that the average concentration of chromium for the entire reach is 600 mg/kg (LANL 2009, 107453, Figure 7.1-7), which is greater than the aquatic community L-ESL (less than or similar to the avian and mammalian aerial insectivore L-ESLs). Therefore, much of Reach S-2 has concentrations greater than the aquatic community L-ESL for at least this one key COPEC.

Projected future conditions. The SCIR discusses temporal trends in sediment concentrations and concludes concentrations of COPECs will be similar or lower in the future (LANL 2009, 107453, section 7.1.8). This conclusion is based on concentrations of COPECs measured over time as well as physical processes of burial of higher concentrations and mixing with cleaner sediment. The Laboratory's annual environmental reports have been documenting changes in concentrations in sediment and storm water over time for key constituents. The focus for Sandia Canyon has been on PCBs analyzed using the congener method, which has a record for comparison only since 2012 (LANL 2014, 261879, Figures 6-10c and 6-10d). Although no conclusions can be made based on that information, the conceptual site model for Sandia Canyon sediment deposits strongly indicates site conditions will improve or stay basically the same.

Half-life of contaminants and potential for recovery. The Laboratory has been monitoring Sandia Canyon since 2012. The purpose of this monitoring is to determine the effects of a grade-control structure installed at the east end of the wetland and changes in outfall chemistry and discharge volumes related to the Sanitary Effluent Reclamation Facility (SERF) expansion. Baseline conditions of geomorphology, flow, water chemistry, and vegetation were reported in 2014 (LANL 2014, 257590). The first year of post-baseline monitoring was 2014. The grade-control structure appears to have stabilized the distal end of the wetlands and reduced erosion and contaminant transport (LANL 2015, 600399, section 5.0). In addition, reducing conditions exist at depth through the wetland (LANL 2015, 600399, section 5.0), which means chromium exists in its trivalent and not in the more soluble, mobile, and toxic hexavalent form. The grade-control structure should also increase deposition of clean sediment over contaminated sediment. Therefore, the wetland appears to be moving toward a condition of less availability of COPECs, including chromium and PCBs. The grade-control structure and the changes related to the SERF are enhancing the potential for recovery of the wetland.

Sources of uncertainty. As stated in the SCIR, "For the aquatic environment, both the field macroinvertebrate surveys and the chironomid bioassay point toward differences that may be related to contaminants from Laboratory operations or other sources. The main uncertainty associated with this conclusion is that non-COPEC confounding factors are also correlated to bioassay measures" (LANL 2009, 107453, section 8.1.5, p. 104). Specifically, although chironomid larval survival was not different from reference, it was negatively correlated to the fraction of clay-sized particles in the sample. Because COPEC concentrations in sediment also tend to be higher with a greater fraction of fine particles, there are also correlations of chironomid measures and COPECs. Stream condition and biological diversity were assessed in November 2007 at one location upstream of the wetland and three locations downstream. All locations showed impairment using the SCIR, and the habitat quality at each location was suboptimal. Similar results were obtained in 10 other surveys conducted from 1998 to 2005. Biological condition in the wetland integrates habitat, confounding factors, and COPECs and does not quantify the contributions of each to impairment.

EcoPRGs. If sediment EcoPRGs for the Sandia Canyon wetland would be useful in evaluating remedial alternatives, then it would be appropriate to calculate ECs based on the chironomid bioassay results. Given that the Sandia Canyon assessment showed no differences in survival but showed differences in chironomid growth from the reference location growth, calculations for growth ranging from EC5 to EC50 might be appropriate. As field surveys and the stream condition index are not directly relevant to the wetlands, where the majority of the sediment with COPECs exist, the surveys would not be useful for calculating numerical EcoPRGs. However, field surveys can be used to document changes in the ecological condition of Sandia Canyon upstream and downstream of the wetlands. Without developing biological metrics and reference wetland conditions, the triad would not provide any additional useful information for risk managers to consider relative to the Sandia Canyon wetlands sediment EcoPRGs. There are difficulties in identifying reference wetlands with similar size, elevation, and flow because the remaining wetlands on Laboratory property, like Sandia Canyon, owe their existence to ongoing effluent releases.

4.0 APPLICATION OF LABORATORY-SPECIFIC STUDIES TO ECOPRGS

The COPCs, exposure pathways, and receptors considered for EcoPRGs are similar to those previously investigated in the ecological risk assessments for Laboratory "canyons," specifically, the Los Alamos and Pueblo, Mortandad, Pajarito, and Sandia watersheds (LANL 2004, 087390; LANL 2005, 089308; LANL 2006, 093553; LANL 2009, 106939; LANL 2009, 107453). The COPCs in the canyons investigations were evaluated using analytical suites; all these investigations characterized exposure to inorganic chemicals, PAHs, and PCBs in soil. Inorganic chemicals were evaluated in all tissue samples, and PCBs were measured in tissues collected from the Pajarito and Sandia watersheds. Exposure pathways addressed in the canyons studies included direct contact, food chain uptake, and incidental soil ingestion. The AE entities for the canyons studies encompassed the range of terrestrial ecological receptors evaluated in Laboratory terrestrial screening assessments. Therefore, aspects of the study designs and conclusions from biological investigations performed in these watersheds complement the EcoPRG process. Given the gradient in COPC concentrations measured in these canyons investigations, it will be possible to pool the results across these studies and determine if "no adverse effects" can be used as site-specific NOECs or ECx values for various receptors.

The following studies were conducted for the canyons ecological risk assessments:

- Seedling germination tests
- Earthworm toxicity tests

- *Chironomus tentans* toxicity test
- Avian nest box studies
- Small-mammal trapping and analysis of pelts and carcasses
- Rapid bioassessment characterization.

The following studies are most relevant to soil EcoPRGs because the effects measured can be tied directly to COPC concentrations:

Seedling germination tests. As part of the baseline ecological risk assessment for the Los Alamos and Pueblo, Mortandad, Pajarito, and Sandia watersheds, soil collected from the 0- to 30-cm (0- to 1-ft) depth interval was used for the plant toxicity tests. The plant toxicity tests used the standard American Society for Testing and Materials (ASTM) Method E1963-98. The plant toxicity tests compared survival rates and shoot and root mass in plants grown in soil from the same locations used for the earthworm toxicity tests with plants grown in the soil sampled from the reference sites. The tests also included positive and negative controls and weekly measurements of plant condition and biomass. The tests used either yarrow (*Achillea millefolium L*. var *occidentalis*) or perennial ryegrass (*Lolium perenne*). Ryegrass is one of the standard test species for the seedling germination test and was selected based on the availability of seeds and the experience of the bioassay laboratory in successfully completing tests with ryegrass. The results from the ryegrass tests from the Pajarito and Sandia watersheds may not be directly comparable with tests conducted with yarrow in the Los Alamos, Pueblo, and Mortandad watersheds (LANL 2004, 087390; LANL 2006, 094161). Samples associated with the seedling germination tests are tabulated in previous canyons investigation reports (LANL 2004, 087390; LANL 2006, 093553; LANL 2009, 106939; LANL 2009, 107453). The possible application of seedling germination tests to the development of EcoPRGs is considered in section 5.

Earthworm toxicity tests. As part of the baseline ecological risk assessment for the Los Alamos and Pueblo, Mortandad, Pajarito, and Sandia watersheds, soil collected from the 0- to 30-cm (0- to 1-ft) depth interval was used for the earthworm bioaccumulation tests. The earthworm tests, which had a 28-d duration, used the standard ASTM Method E1676-97 lumbricid earthworm *Eisenia fetida* and measured growth and survival in addition to concentrations of COPCs in worm tissues. Soil was homogenized, pH was measured, and the moisture content was made consistent among the samples. Worms were selected and their combined live weight was recorded before they were placed into each test unit. The tests also included positive and negative controls and observations of earthworm behavior. The toxicity tests were used to compare growth and mortality of earthworms from locations in Los Alamos and Pueblo, Mortandad, Pajarito, and Sandia watersheds with reference sites. The locations were selected to represent a gradient of concentrations for COPCs associated with both the soil invertebrate receptor and the mammalian and avian receptors that feed on soil invertebrates. Earthworms were sent to an analytical laboratory for chemical analyses, and this information can be used in assessing wildlife exposure. Samples associated with the earthworm toxicity tests are tabulated in previous canyons investigation reports (LANL 2004, 087390; LANL 2006, 093553; LANL 2009, 106939; LANL 2009, 107453). The possible application of earthworm toxicity test results to the development of EcoPRGs is considered in sections 5 and 6.

The following study is most relevant to sediment EcoPRGs because the effects measured can be tied directly to COPC concentrations.

***Chironomus tentans* toxicity test**. As part of the baseline ecological risk assessment for Cañon de Valle and the Los Alamos and Pueblo, Mortandad, Pajarito, and Sandia watersheds, sediment and in some cases paired water samples were collected from the 0- to 5-cm (0- to 0.16-ft) depth interval and used in

the EPA Method 100.2 (EPA 2000, 073776) 10-d growth and survival test with the larval insect *Chironomus tentans*. Each sediment sample was tested at 100% concentration only; dilution series were not run on the sites. Standard controls and reference toxicants were included. The endpoints for this test include both survival and growth (as ash-free dry weight). Samples associated with the *Chironomus tentans* toxicity tests are tabulated in previous canyons investigation reports (LANL 2004, 087390; LANL 2006, 093553; LANL 2009, 106939; LANL 2009, 107453). As discussed in section 3, these tests can be used on a site-specific basis to assist in developing sediment EcoPRGs.

The following studies may also be relevant to soil EcoPRGs because the effects and exposure studies for wildlife can be related to COPC concentrations.

Avian nest box studies. An avian nest box monitoring network has existed at the Laboratory and its vicinity since 1997; the network includes both potentially contaminated and noncontaminated areas. As part of the baseline ecological risk assessment for the Los Alamos and Pueblo, Mortandad, Pajarito, and Sandia watersheds, additional nest boxes were placed in the canyon bottoms or canyon bench areas and the tributary canyons. Both the western bluebird (*Sialia mexicana*) and the ash-throated flycatcher (*Myiarchus cinerascens*) occupied these boxes. The nest box study included field measures of effect on reproductive success of these avian species (including clutch size, fledgling success, growth of fledglings, etc.) and measures of exposure through analysis of COPC concentrations in unhatched western bluebird eggs and unconsumed prey (insects) collected within the boxes. Boxes in the Cañada del Buey watershed and boxes from two areas outside the Laboratory [the Los Alamos Golf Course and the Guaje Pines Cemetery (LANL 2004, 087390)] were also included in the study for reference. Eggs from individual boxes within a reach were submitted as samples. In some cases, individual boxes contained sufficient material for analysis, but in other cases, insects from more than one box in a reach were combined to obtain sufficient sample size for analysis. Because of sample size limitations, egg and insect samples were primarily analyzed for metals. The relevant soil samples are those collected from reaches with nest boxes occupied during these studies. Samples associated with the avian nest box studies are tabulated in previous canyons investigation reports (LANL 2004, 087390; LANL 2006, 093553; LANL 2009, 106939; LANL 2009, 107453). These studies include some information on exposure and effects, but the relevant sediment or soil exposure data are an uncertainty because although the nest boxes are placed near or within contaminated fluvial sediment deposits, some of the foraging occurs beyond these deposits and the exposure information is not known. Information on the contaminants in the diet is known for middle trophic level avian receptors, but it would be an extrapolation to estimate exposure from the primary contaminated medium (soil).

Small-mammal trapping and analysis of pelts and carcasses. Small mammals were trapped in reaches within the Los Alamos and Pueblo, Mortandad, and Sandia watersheds and background reaches. The results of the trapping determined measures of effect on the small-mammal population, including relative abundance, species composition, reproductive status, and body weight. The field measures of the small mammals were lines of evidence for the effects to the small mammals themselves. Small mammals were also collected for laboratory analysis to determine the concentration of COPCs in tissues. The concentrations in the tissues were lines of evidence for the exposure of the Mexican spotted owl as well as for the mammalian carnivore (the gray fox), which was not designated as an individual AE. For some watersheds, the individuals of most species from each reach were separated into pelts and carcasses; the pelts and carcasses were then combined so one pelt and one carcass sample from each species were sent for laboratory analysis for each reach. The exception is shrews, which were too small to allow separation into pelt and carcass; instead, whole-body samples were submitted. Analyses conducted on the carcass and pelt samples included EPA Method SW-846 6010B metals, perchlorate, mercury, PCBs, americium-241, cesium-137, and strontium-90. Composite soil samples were also collected from the trapping arrays and analyzed for the same contaminant suites. Samples associated

with the small-mammal trapping studies are tabulated in previous canyons investigation reports (LANL 2004, 087390; LANL 2006, 093553; LANL 2009, 107453). The small-mammal results can be used to evaluate both exposure and effects, although the range of COPC concentrations is relatively narrow because the composite samples from the trapping arrays represent the average concentration for that area. The possible use of the small-mammal exposure information as site-specific bioaccumulation information is considered in section 6.

The following study was also conducted for watershed risk assessments and may be relevant to sediment EcoPRGs.

Rapid bioassessment characterization. Rapid bioassessment characterization was conducted in reaches in the Los Alamos and Pueblo, Mortandad, Pajarito, and Sandia watersheds that had sufficient flow to potentially support aquatic macroinvertebrate communities with use of the EPA Rapid Bioassessment Protocol (EPA 1999, 073728). Aquatic macroinvertebrates were collected in association with the bioassessment. The approved biota investigation work plans specified use of a Hess sampler to collect aquatic macroinvertebrates when sufficient water was present. However, only 5 of 19 reaches had sufficient water to use the Hess sampler. The Hess sampler is necessary to collect data for comparison with the SCI; therefore, aquatic macroinvertebrates from only these 5 reaches could be compared with the SCI. The SCI compares sites with a reference condition, which is based on historical data from New Mexico streams. Semiquantitative sampling was conducted at the other sites with use of a D-frame dip net to determine taxonomic composition of macroinvertebrates. In addition to the general lack of enough water to permit use of the Hess sampler, habitat considerations were a predominant factor for the diversity and abundance of aquatic macroinvertebrates. Samples associated with the rapid bioassessment studies are tabulated in previous canyons investigation reports (LANL 2004, 087390; LANL 2006, 093553; LANL 2009, 106939; LANL 2009, 107453). As discussed in section 3, this information can be used on a site-specific basis to assist in developing sediment EcoPRGs.

5.0 APPLICATION OF LABORATORY-SPECIFIC SOIL BIOASSAYS TO ECOPRGS

Seedling germination tests and earthworm bioaccumulation tests were conducted between 2002 and 2008 for four Laboratory watersheds: Los Alamos/Pueblo, Mortandad, Pajarito, and Sandia. As mentioned in section 1.0, one may opt to conduct empirical studies to determine if cleanup is warranted. Such studies may include bioassays or direct ecological observations on the flora and fauna at the contaminated site. If site observations are used as a line of evidence an assessment of the spatial distribution of contamination correlated to the ecological observations should be included. This correlation should establish whether available habitat is collocated with areas of contamination or if the available habitat is located where contamination was not detected. The spatial analysis should be conducted whenever the EcoPRGs are applied, and field observations are used to determine whether or not cleanup of the site is warranted. Graphical comparison of these soil bioassay endpoints and a statistical comparison using the Wilcoxon rank sum test determined that the results from the analytical laboratories should not be pooled.

Simple linear regression of bioassay endpoints to concentrations in soil was evaluated as a screening tool to determine if evidence exists for dose-response relationship of bioassay endpoints to COPCs and confounding factors. Only those relationships indicating potential adverse effects, such as decreased survival or growth with increased COPC concentrations, were considered to be statistically significant. Either direction of slope of the bioassay endpoint versus the confounding factors was evaluated if statistically significant. First of all, the data (both COPC and potential confounding factor) were summarized. This summary was based on the chemistry analyses of the samples that pair with the bioassays (Table 5.0-1). Regarding these data, it should be noted that particle-size analysis, which

provides some potentially confounding factor measures, was not conducted on the samples collected from Los Alamos and Pueblo Canyons. One trend observed with particle size is that higher levels of silt and clay tend to be associated with higher concentrations of naturally occurring metals. Therefore, concentrations of aluminum and iron, a surrogate measure of particle size (in particular silt and clay), were also evaluated for all canyons. All COPCs with more than five detected concentrations and all potentially confounding factors were evaluated in the linear regressions.

An example of the results of these statistical analyses is provided for an inorganic COPC (chromium) in Figure 5.0-1 and for an organic COPC [benzo(a)pyrene] in Figure 5.0-2. These regressions initially evaluated all sample results—both detections and nondetections. The results from each toxicity testing laboratory were treated separately. Strong evidence for adverse effects of a COPC on plants or soil invertebrates would be concordance of the results from the different laboratories as well as effects on multiple endpoints. Attachment 1 (on CD included with this document) presents all the linear-regression models.

5.1 Seedling Germination COPC Results

Overall, the seedling germination rates were high (greater than 80% with 1 exception out of 43 samples), and growth rates were similar to references and within the range expected based on natural variation. Arsenic, chromium, copper, cyanide (total), lead, mercury, nickel, silver, vanadium, zinc, Aroclor-1254, and Aroclor-1260 had statistical relationships with seedling germination bioassay measures (Table 5.1-1). Dose-response is not indicated because of discordance of the results among the measures or between the toxicity testing laboratories. Based on this information, the maximum concentrations tested can be used as site-specific NOECs for plants.

5.2 Earthworm Bioassay COPC Results

Overall, earthworm survival rates were high (greater than 80% with 2 exceptions out of 42 samples), and weight changes were similar to the references and within the range expected based on natural variation. Only zinc and 8 PAH compounds had negative statistical relationships with earthworm survival (Table 5.2-1). Dose-response is not indicated because of discordance of the results among the measures or between the toxicity testing laboratories or the lack of biological significance of the results. Selenium was the only COPC that showed a statistical relationship with earthworm growth (weight change), but the correlation was in the positive direction and was not indicative of an adverse effect. Based on this information, the maximum concentrations tested can be used as site-specific NOECs for soil invertebrates. A lack of dose-response was also used as the basis for a site-specific NOEC for similar tests conducted for Hanford Site EcoPRG development (CHPRC 2014, 261847).

In addition to the canyons ecological risk-assessment studies discussed above, site-specific studies at SWMUs or AOCs may be used to supplement toxicity information available for specific COPCs. One such study was conducted for SWMU 32-002(b2) in the upper Los Alamos Canyon Aggregate Area for the High-Angle Remediation Project (HARP). The ecological receptors and pathways are similar to those at other Laboratory sites. The COPC of interest at this site was mercury, and the endpoint evaluated was soil invertebrates. A wide range of mercury soil concentrations was available from this site, and four locations were tested using the earthworm bioaccumulation test. The results of those tests are documented in a laboratory report (TRE Environmental Strategies 2015, 601279) that provided information on the test conditions as well as survival and weight change of the earthworm. These results were combined with the other earthworm bioaccumulation test results from the canyons investigations to evaluate a response of earthworm survival and growth to mercury concentrations. Linear and log-linear models did not reveal any adverse relationship. Therefore, the maximum mercury concentration of

395 mg/kg tested is a site-specific NOEC. This NOEC, including the results from SWMU 32-002(b2), is substantially greater than the maximum mercury soil concentration tested in Laboratory canyon studies (1.71 mg/kg) or the LOEC from the ECORISK database (0.5 mg/kg).

5.3 Bioassay Conclusions

Given the lack of dose-response relationships indicated for the majority of COPCs based on the linear-regression analyses and the additional evaluation for those significant regressions, the maximum concentration tested can be used as a site-specific NOEC for plants and soil invertebrates. A lack of dose-response was also used as the basis for a site-specific NOEC for similar tests conducted for Hanford Site EcoPRG development (CHPRC 2014, 261847). The lack of toxicity based on testing field-aged samples with a variety of COPCs is typical. As mentioned in section 2.3, the impact of contaminant ageing on reducing toxicity is also recognized in the PNECs developed by the European Commission (Smolders et al. 2009, 260205). These site-specific NOECs can be established for COPCs with at least five detected concentrations in the population of samples associated with the soil bioassays. Five or more detected concentrations were required to have more than the minimum number of results for regression analyses (minimum of three) and to encompass at least 10% of the samples tested for toxicity. Table 5.3-1 lists the COPCs with analytical data for seedling germination bioassay tests and indicates if a site-specific NOEC is available. The table also presents a brief summary of the dose-response information for each COPC. Table 5.3-2 provides the same information for the earthworm bioassay tests. To the extent the bioassay results are used to derive EcoPRGs for SWMUs and AOCs across the Laboratory, the use of the EcoPRGs is accompanied by a discussion of the applicability of the values to the specific SWMU or AOC. Such discussions should emphasize the similarities and differences between the SWMU/AOC and the sites of the field-laboratory studies used in the derivation of the EcoPRGs.

6.0 APPLICATION OF LABORATORY-SPECIFIC TISSUE DATA TO ECOPRGS

Evaluation of the earthworm and small-mammal canyons tissue data for possible development of site-specific bioaccumulation factors (BAFs) or tissue-to-soil ratios consisted of determining which COPCs were measured in both soil and tissues, and among those COPCs, which were detected in at least five or more paired measurements. The list of COPCs included most of the inorganic chemicals and the PCB Aroclor mixtures.

For the soil invertebrate tissue data (earthworm only), statistical models were developed for 9 of 23 COPCs. Statistical models were developed for only 5 of the 23 COPCs evaluated for mammal tissue. In some cases, the median BAF or regression slopes were larger than the literature-based transfer factors (TFs); and in others, they were smaller. The reason for using site-specific values was to better characterize wildlife exposure to COPCs through the food chain. These regressions are generally not statistically significant, and the site-specific values are not substantially different from the literature-based TFs. However, these site-specific bioaccumulation factors can be useful for the EcoPRG uncertainty evaluation if one considers the range of factors measured in comparison with the literature TFs.

Additional site-specific studies may also be used to better understand the potential for contaminant bioaccumulation. The earthworm bioaccumulation study conducted for SWMU 32-002(b2) in the Upper Los Alamos Canyon Aggregate Area for the HARP provides such additional site-specific information (TRE Environmental Strategies 2015, 601279). Mercury was measured in both earthworms and soil, and these paired data were combined with the same type of data from the canyons investigations. With these data, the resulting regression of mercury in earthworms was statistically significantly related to the concentration in soil. This statistical significance was from one large concentration in both earthworm and

soil concentrations. To better weight the influence of the earthworm results across a wide range of soil concentrations, the median BAF (ratio of earthworm to soil concentrations) was calculated. The samples with soil concentrations less than 0.2 mg/kg mercury were excluded to eliminate relationship influenced by soil background. The TFs associated with the greater than soil background mercury concentrations had a median of 0.47 mg-mercury-earthworm-dry weight/kg-mercury-soil (the interquartile range was 0.22 to 1.3). In comparison, the TF-invertebrate in the ECORISK database used for ESL and L-ESL calculations is 3.93 mg-mercury-earthworm-dry weight/kg-mercury-soil. Fresh-weight data were converted to dry weight using the 83.3% earthworm moisture content used elsewhere in the ECORISK database.

7.0 ECOPRGS IN RISK MANAGEMENT

Table 7.0-1 presents the soil EcoPRGs calculated for all receptors and the final (minimum) EcoPRG and associated receptor for a hypothetical 1-ha site. The final EcoPRG receptor is either the plant, earthworm, shrew, deer mouse, or robin. The wildlife EcoPRG calculations include site area (see equations 2 and 3 in section 2.4), which means that these EcoPRGs will change based on site area. The EcoPRGs for plants and earthworms do not vary with site area. All of the information supporting the EcoPRG calculations is in the ECORISK database (Release 4.0 or later). The final EcoPRG for inorganic COPCs should be compared to the soil background data, including the background value and the range of background concentrations. The cleanup level(s) should not be below background concentrations, and corrective actions do not require that concentrations be reduced to background to address potential risks. However, it is not clear how much above background concentrations must be before they result in risks to receptors. None of the final EcoPRGs listed in Table 7.0-1 are less than Laboratory soil background concentrations.

The EcoPRGs can be used in the corrective action process to evaluate remedial alternatives for Laboratory SWMUs and AOCs. The EcoPRGs were developed for common soil COPCs and address a range of terrestrial receptors. Although wildlife EcoPRGs are adjusted for site area, the plant and earthworm EcoPRGs are not adjusted for this important measure related to the spatial scale of ecological populations. If the final EcoPRG is based on LANL-specific laboratory or field studies not specific to the site being evaluated, a brief discussion of the similarities and differences of the SWMU/AOC and the site(s) of the studies used in the derivation of the EcoPRGs should be provided. Because the studies are specific to the Laboratory, the discussion should emphasize the applicability of the study results to the SWMU/AOC. Alternatively, it might be appropriate to conduct site-specific soil bioassays to determine if a dose-response relationship for plant and/or earthworm is present. The results of the site-specific soil bioassay(s) can be used to adjust or verify the cleanup level and quantitatively determine if remediation is required based on these receptors. In some cases, it might also be appropriate to consider the lowest wildlife EcoPRG for a COPC and look at the impacts of implementing this value as the cleanup level instead of the plant/earthworm EcoPRG(s). A qualitative assessment of plant community from the ecoscoping site visit might also provide information concerning the presence or absence of stressed vegetation. If site observations are used as a line of evidence, the correlation of the spatial distribution of contamination with available habitat (i.e., whether or not areas of contamination are collocated with areas of contamination) should be provided. This type of analysis should be provided whenever the EcoPRGs are applied and field observations are used to determine whether or not cleanup of a site is warranted. In all cases where EcoPRGs are used, they must be clearly documented and communicated, and they must be based on site conditions.

Wildlife soil EcoPRGs are based on PAUFs, and a major underlying assumption is that habitat quality is relatively uniform throughout the assessment population area. In particular, the site in question must be no more or less attractive to wildlife than the uncontaminated areas (attractiveness must be measured on

a per-unit basis). More site specificity can be incorporated into EcoPRGs by considering suitable habitat because receptors will use the fraction of the population area that provides adequate resources to meet their needs. In other cases, the calculated EcoPRG can be correspondingly adjusted to reflect the fraction of usable habitat for receptors occurring on-site (e.g., half the site is covered by structures). The site area used to calculate PAUFs can be adjusted for the nonhabitat fraction of the site. Such an adjustment would be documented by site field notes, photographs, and/or maps. Hope and Peterson (2000, 070087) employ this approach in performing population-level ecological risk assessments, where the site is described according to habitat quality.

For Laboratory SWMUs/AOCs, the Laboratory land cover map (LANL 2017, 602649, Plate 1) can be used to refine the potential risks associated with soil contamination. Animal density can be related to landscape features. For example, Raymer and Biggs (1994, 056038) showed that mammal diversity and density is higher at outfalls than at locations without water. One could use the relative density of wildlife in various landscape habitat elements to construct a weighted average COPC concentration. Such information can also be used to understand the potential benefits of various remedial alternatives, which can be compared with the costs of those alternatives.

Animal HR and the associated population area are key parameters for wildlife EcoPRGs. As discussed above, animal density varies with land cover and logically HR also varies. For some of the representative receptors (e.g., deer mouse), HR information is reported across a range of habitats. Thus, HR and the associated PAUF can be refined to reflect those land-cover types present at a SWMU/AOC. Because some receptors (e.g., the shrew) might not be broadly distributed across the Laboratory, the presence or absence of such species can be refined based on information on land cover or inferred from Laboratory field studies.

There is an explicit consideration of spatial scale in the wildlife soil EcoPRGs, and spatial extent of contamination should be part of the risk-management considerations for plants and soil invertebrates. An assumption is that the COPC has no additional sources within an area equal to the assessment population area around the SWMU/AOC. For middle trophic level wildlife, this area is16 ha, or a circle with a radius of about 225 m (738 ft) from the center of the SWMU/AOC. Therefore, evaluating spatially proximate sites as a group rather than as individual sites in evaluations of remediation options might be appropriate. The spatial extent of contamination, or the fraction of the assessment population area that is greater than the EcoPRG(s), is information that will be useful to risk management in making remediation decisions. Various statistical methods are available to interpolate between sample points and to estimate areas greater than the EcoPRG(s) for various receptors. This information can support remedial approaches or can be used to calculate the fraction of the assessment population area greater than the EcoPRG(s) and help to focus the remedial activities.

A primary distinction exists between how risk assessments and risk-management decisions based on EcoPRGs apply soil data for a SWMU/AOC or area. Risk assessments use the 95% upper confidence limit (UCL) of the mean for the entire site. Risk-management decisions using EcoPRGs consider the spatial distribution of the COPC(s) over the assessment population area. Maps should be prepared displaying COPC concentrations, source areas (SWMUs/AOCs), habitat or plant cover types, and the assessment population boundary (or boundaries). Such maps would show whether COPC concentrations are broadly distributed or localized. The 95% UCLs should also be recalculated based on proposed remedial alternatives to evaluate effectiveness and extent of the action.

Factors that can further refine soil EcoPRGs include an animal's seasonal use of a site and chemical bioavailability. These variables affect a receptor's exposure to COPCs through the proportion of time spent in a contaminated area and through the proportion of the total dose received through diet. Although some of the sensitive receptors are migratory (e.g., robin), seasonal use is currently assumed to be

100%. Because the robin and other birds representing those feeding guilds do not necessarily use the assessment population area year round, the soil EcoPRGs are protective. Considering bioavailability, detailed studies to characterize the assimilated fraction of the ingested dose for a COPC are ultimately necessary to fully characterize contaminant availability. However, it may be more practical to get a better estimate of trophic transfer by using the earthworm bioaccumulation test or by collecting terrestrial soil invertebrates from the site and analyzing them for COPCs.

Where aquatic communities exist, sediment EcoPRGs should evaluate multimedia exposures. In addition, the potential for transport to downstream locations and potential for impacts on receptors at these locations should be evaluated. Locations with aquatic sediment, which are rare at the Laboratory, provide habitat for aquatic biota and water for wildlife. However, the Sandia Canyon wetland example raises additional concerns and limitations related to application of EcoPRGs. One option for application of EcoPRGs is to document a volume of sediment that can be removed to mitigate exposure and risk to one or more AEs. For example, if a range of ECs from EC5 to EC50 is calculated from the bioassays, then these values can be equated to various levels of disturbance in the wetlands. That is, as the EC used as an EcoPRG decreases, the area requiring remediation to meet that level increases; therefore, the expected damage to habitat associated with cleanup could increase. This information can be balanced by risk managers when making recommendations on the need for, and the scale of, corrective action in wetlands or other such sensitive areas in the canyon bottoms.

Another consideration for sediment toxicity is whether it is based on contact and exposure to the solid phase (sediment bound) or the interstitial water (pore water). For many inorganic chemicals, including chromium (discussed for Sandia Canyon), the more toxic phase is the soluble form of the COPC found primarily in the sediment pore water. This results in a linkage between contaminant transport and chemical form, raising the potential for in situ remediation approaches. In situ approaches also have the potential added benefit of no physical impacts on sediment deposits.

There are regulatory concerns related to removal of wetlands, and in particular the Sandia Canyon wetland, one of the largest wetlands on Laboratory property. For example, the Sandia Canyon wetland has a significant inventory of chromium and PCBs, but the Laboratory has taken actions to improve the functioning of the wetland such that these contaminants are not mobile and do not represent a source for other receptors, locations, and media. Clearly, information on the biological conditions in the wetland, as well as monitoring of transport via surface water and other pathways, is important in making informed risk-management recommendations for such areas. It is also worth considering restoration goals early during problem formulation, as recommended by Kapustka et al. (2015, 601150).

The example of Sandia Canyon sediment also illustrates the limitations and conservatism of using L-ESLs as EcoPRGs. As noted above, it is likely that much, if not all, of the wetland (Reach S-2) exceeds the aquatic community L-ESL for chromium. Use of the L-ESL as an EcoPRG would indicate remediation of a large portion of the wetland. Targeted removals of the higher concentrations may not be feasible, given that the higher concentrations are fairly widely distributed at depth. This level of response does not appear commensurate with the observed ecological effects or the potential loss of habitat. Also, as discussed above regarding the potential for recovery, the wetland is performing its function of retaining contaminants, providing reducing conditions for chromium, and mitigating transport.

The question common to the remedial alternative evaluation is, “Will the cleanup cause more ecological harm than the current site condition?” (EPA 1999, 070086, Question 3) (i.e., “Is the impact of the remedial alternative more damaging to the environment than the ecological risks of the COPCs?”). This assessment may use a metric such as ecosystem services (e.g., pollination, mitigate erosion) to help quantify benefits and processes like Net Environmental Benefits Analysis (NEBA) as formal methods for

such evaluations. Such approaches are recommended in Executive Memorandum M-16-01, "Incorporating Ecosystem Services into Federal Decision Making" (October 7, 2015).

Efroymson et al. (2003, 601397) presented a framework for conducting NEBA for petroleum sites. Although aspects of their approach are specific to petroleum, the main steps they posit could be applied to any type of corrective-action site. NEBA for contaminated sites generally involves comparing management alternatives: leaving contamination in place, remediating using conventional methods (typically removal, but also more innovative in situ methods), performing ecological restoration activities not directly related to remediation, or some combination of the first three alternatives. Using NEBA, one can compare remedial alternatives, including natural attenuation, to hopefully make decisions that are ultimately more beneficial to the environment. The main components of the NEBA framework are planning (management goals, endpoints, reference conditions, conceptual model, analysis plan); determining the values of ecosystem services for the reference state(s); evaluating each management alternative; ranking and selecting an alternative, and monitoring to determine efficacy.

Information on the distribution of contamination across the site and the corresponding habitat quality can help to address costs and benefits of environmental remediation. Specific considerations in this regard are depth of contamination and the likelihood of ecological exposures. For example, if the higher concentrations are below 1 m (about 3 ft), then exposures and adverse effects are less likely. In this scenario, one would have to remove the less-contaminated soil and potentially viable ecological habitat to remove contamination. Another consideration is the potential for recovery of the habitat associated with the remedial action. Monitored natural remediation or other in situ remedial approaches may be the best alternatives in some circumstances. Such methods may lead to disturbing less of the environment compared with removal and restoration.

In general, PRGs may have a wide range of objectives, including protection of human health or meeting standards. The full range of PRGs and objectives needs to be evaluated, including the range of EcoPRGs for various ecological receptors. Ideally, remedial alternatives would be complementary, but there may be cases where one endpoint, such as ecological risk, indicates minimal to no impacts, but another endpoint indicates larger impacts and a greater scale of remediation. Such cases emphasize the complex nature of remedial decision-making.

8.0 REFERENCES

The following reference list includes documents cited in this report. Parenthetical information following each reference provides the author(s), publication date, and ERID or ESHID. This information is also included in text citations. ERIDs were assigned by the Associate Directorate for Environmental Management's (ADEM's) Records Processing Facility (IDs through 599999), and ESHIDs are assigned by the Environment, Safety, and Health Directorate (IDs 600000 and above). IDs are used to locate documents in the Laboratory's Electronic Document Management System and in the Master Reference Set. The New Mexico Environment Department Hazardous Waste Bureau and ADEM maintain copies of the Master Reference Set. The set ensures that NMED has the references to review documents. The set is updated when new references are cited in documents.

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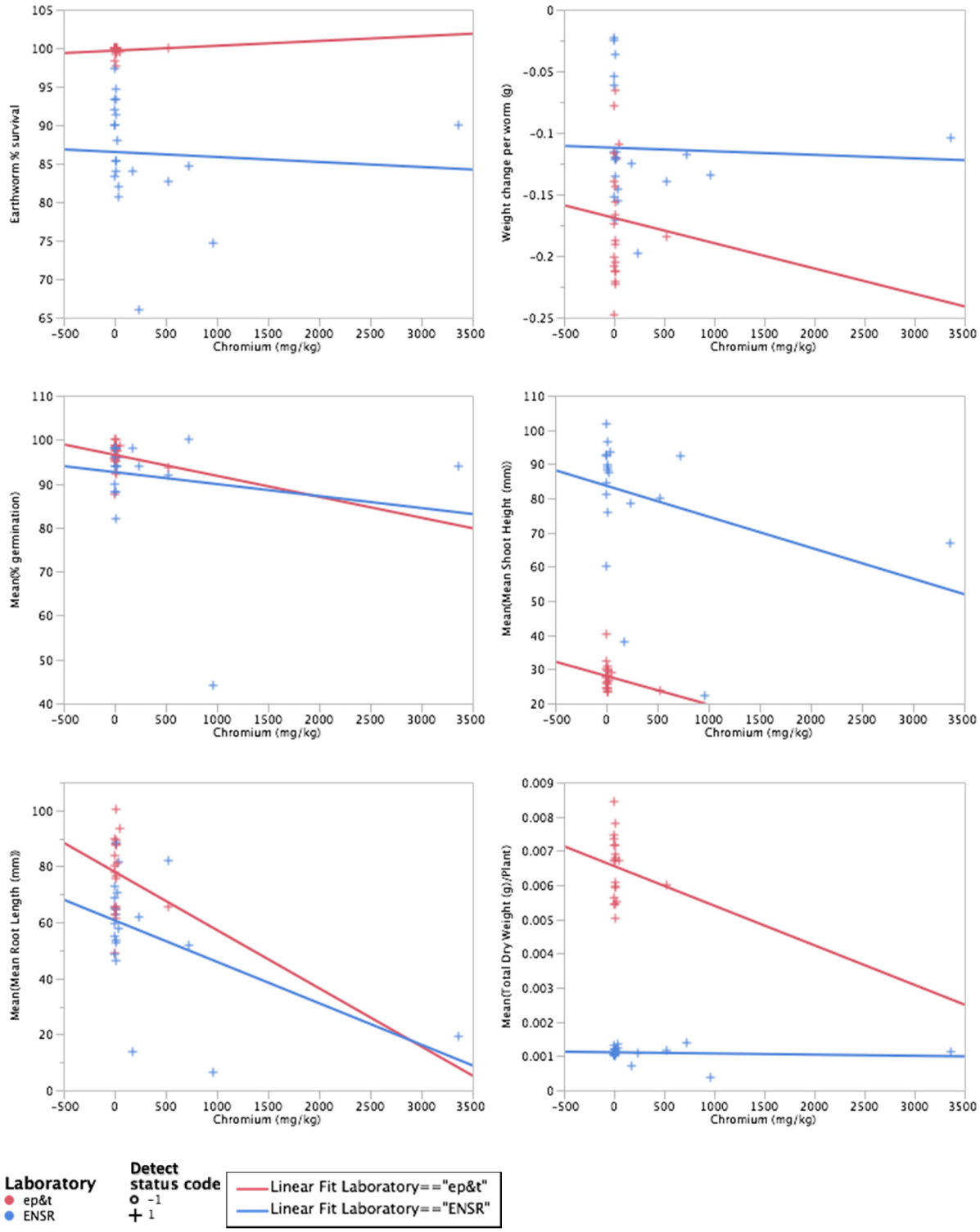

Figure 5.0-1 Bioassay measures versus chromium (the studies were from the reports cited in section 5)

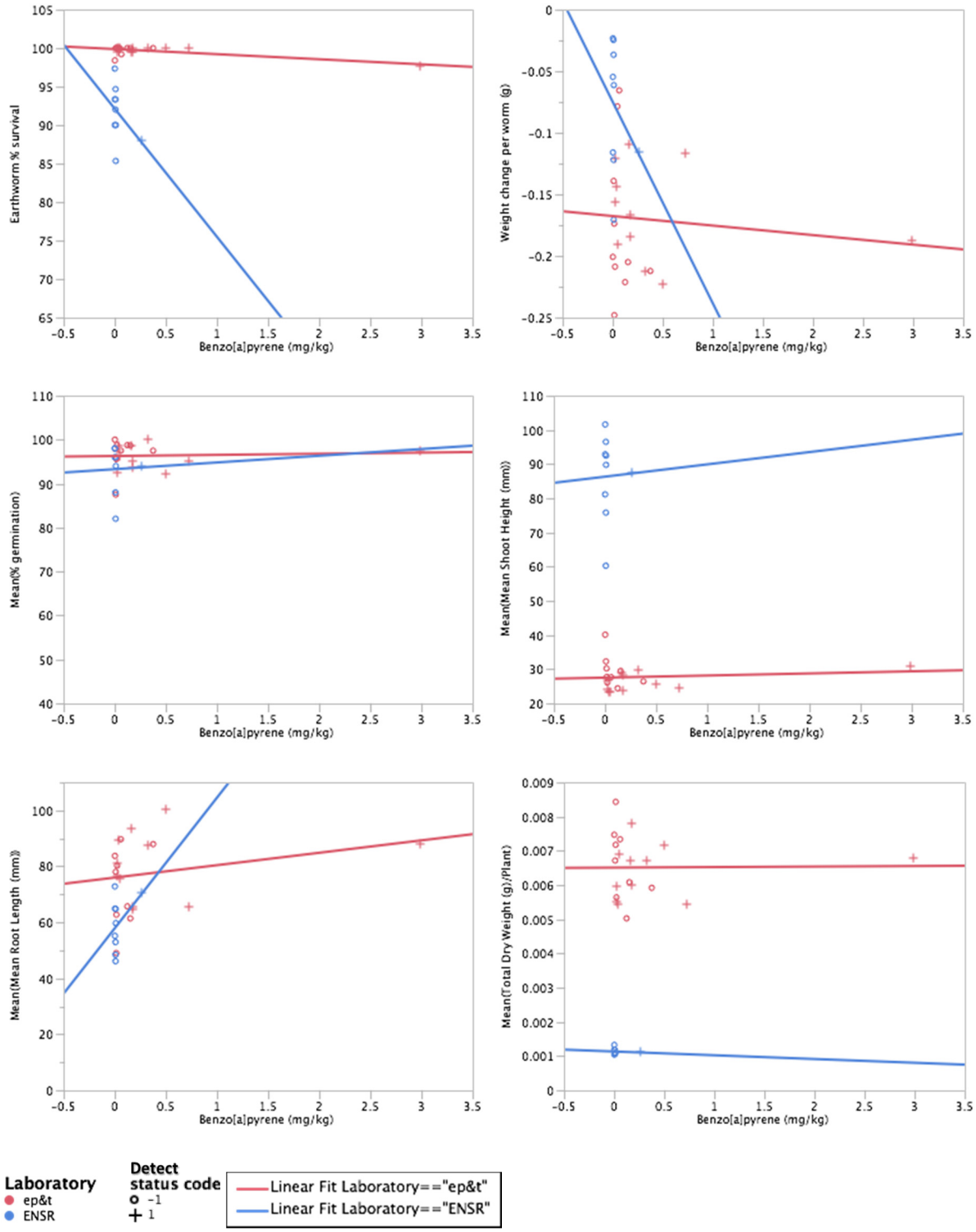

Figure 5.0-2 Bioassay measures versus benzo[a]pyrene (the studies were from the reports cited in section 5)

Table 2.1-1
Terrestrial Screening Receptors and EcoPRG Receptors

Taxonomic Group/ Trophic Level	Screening Receptor	EcoPRG Receptor
Primary producer	Plant	Plant
Detritivore	Soil invertebrate (earthworm)	Soil invertebrate (earthworm)
Mammalian herbivore	Mountain cottontail	Mountain cottontail
Mammalian invertivore	Montane shrew	Montane shrew
Mammalian omnivore	Deer mouse	Deer mouse
Mammalian carnivore	Gray fox	Gray fox
Avian herbivore	American robin (plant diet)	American robin
Avian invertivore	American robin (invertebrate diet)	American robin
Avian omnivore	American robin (omnivore diet)	American robin
Avian omnivore	American kestrel	American kestrel
Avian carnivore	American kestrel (flesh diet)	Mexican spotted owl

Table 2.2-1
Soil EcoPRG COPCs with Receptor Information

Analyte Group	Analyte Name	Soil, Plant	Soil, Invertebrate	Soil, Bird	Soil, Mammal
Dioxin/Furan	Tetrachlorodibenzodioxin[2,3,7,8-]	—[a]	X[b]	—	X
HE compounds	Amino-2,6-dinitrotoluene[4-]	X	—	—	X
	Amino-4,6-dinitrotoluene[2-]	X	—	—	X
	Dinitrotoluene[2,4-]	—	—	—	X
	Dinitrotoluene[2,6-]	—	—	—	X
	HMX[c]	X	X	—	X
	PETN[d]	—	—	—	X
	RDX[e]	—	X	X	X
	Tetryl	—	—	—	X
	Trinitrobenzene[1,3,5-]	—	—	—	X
	Trinitrotoluene[2,4,6-]	X	X	X	X
Inorganic Chemicals	Antimony	X	X	—	X
	Arsenic	X	X	X	X
	Barium	X	X	X	X
	Beryllium	X	X	—	X
	Boron	X	—	X	X
	Cadmium	X	X	X	X
	Chromium (total)	—	—	X	X

Table 2.2-1 (continued)

Analyte Group	Analyte Name	Soil, Plant	Soil, Invertebrate	Soil, Bird	Soil, Mammal
Inorganic Chemicals (cont.)	Chromium (hexavalent)	X	X	X	X
	Cobalt	X	—	X	X
	Copper	X	X	X	X
	Cyanide (total)	—	—	X	X
	Lead	X	X	X	X
	Manganese	X	X	X	X
	Mercury (inorganic)	X	X	X	X
	Mercury (methyl)	—	X	X	X
	Nickel	X	X	X	X
	Selenium	X	X	X	X
	Silver	X	—	X	X
	Thallium	X	—	X	X
	Uranium	X	—	X	X
	Vanadium	X	—	X	X
	Zinc	X	X	X	X
PAHs	Acenaphthene	X	—	—	X
	Acenaphthylene	—	—	—	X
	Anthracene	X	—	—	X
	Benzo(a)anthracene	X	—	X	X
	Benzo(a)pyrene	-	—	—	X
	Benzo(b)fluoranthene	X	—	—	X
	Benzo(g,h,i)perylene	—	—	—	X
	Benzo(k)fluoranthene	—	—	—	X
	Chrysene	—	—	—	X
	Dibenzo(a,h)anthracene	—	—	—	X
	Fluoranthene	—	X	—	X
	Fluorene	—	X	—	X
	Indeno(1,2,3-cd)pyrene	—	—	—	X
	Methylnaphthalene[2-]	—	—	—	X
	Naphthalene	X	—	X	X
	Phenanthrene	—	X	—	X
	Pyrene	—	X	X	X
PCBs	Aroclor-1016	—	—	—	X
	Aroclor-1242	—	—	X	X
	Aroclor-1248	—	—	X	X
	Aroclor-1254	X	—	X	X
	Aroclor-1260	—	—	X	X

Table 2.2-1 (continued)

Analyte Group	Analyte Name	Soil, Plant	Soil, Invertebrate	Soil, Bird	Soil, Mammal
Phthalates	Bis(2-ethylhexyl)phthalate	—	—	X	X
	Butylbenzylphthalate	—	—	—	X
	Di-n-butylphthalate	X	—	X	X
	Di-n-octylphthalate	—	—	—	X
	Diethylphthalate	X	—	—	X
	Dimethyl phthalate	—	X	—	X

[a] — = No toxicity information.

[b] X = Toxicity information available.

[c] HMX = Octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine.

[d] PETN = Pentaerythritol tetranitrate.

[e] RDX = Hexahydro-1,3,5-trinitro-1,3,5-triazine.

Table 2.4-1
Measures Required for the Wildlife Exposure Model

Receptor	Parameter	Value	Unit(s)	Reference	Notes
American robin	Body weight	0.081	kg	EPA 1993, 059384, p. 2-197	Mean of available values
Turdus migratorius	Food intake[a] Herbivore diet	0.160	kg-food dry wt/kg-body wt/d	Nagy 2001, 253420	Estimated using 0.081 kg body weight and Nagy (2001, 253420) allometric scaling formula for all birds (most appropriate diet and high r^2 for model)
	Food intake[a] Omnivore diet	0.148	kg-food dry wt/kg-body wt/d	Nagy 2001, 253420	Estimated using 0.081 kg body weight and Nagy (2001, 253420) allometric scaling formula for omnivorous birds (most appropriate diet and high r^2 for model)
	Food intake[a] Invertivore diet	0.130	kg-food dry wt/kg-body wt/d	Nagy 2001, 253420	Estimated using 0.081 kg body weight and Nagy (2001, 253420) allometric scaling formula for insectivorous birds (most appropriate diet and high r^2 for model)
	Fraction soil in diet: herbivore, omnivore, and insectivore	0.061, 0.063, 0.064	Unitless	EPA 2007, 602500, Attachment 4-1, Table 3	Used median dove value for herbivore diet, median woodcock value for insectivore diet, and average of these two species for omnivore diet
	Plant diet	0, 0.5, or 1[b,c]	Unitless	None	Modeled with three diets: herbivore, omnivore, and insectivore
	Soil invertebrate diet	1, 0.5, or 0[b,c]	Unitless	None	Modeled with three diets: herbivore, omnivore, and insectivore
	Home range	0.42[b]	ha	EPA 1993, 059384, p. 2-199	HR data represent average territory size in an open, semiurban environment
	Population area	16.8[b]	ha	Calculated	40 times HR (see text for explanation)
American kestrel	Body weight	0.116	kg	(Dunning 2008, 601245)	Mean of five environments with males and females reported separately
Falco sparverius	Food intake[a] (omnivore)	0.114	kg-food dry wt/kg-body wt/d	Nagy 2001, 253420	Estimated using 0.116 kg body weight and Nagy (2001, 253420) allometric scaling formula for omnivorous birds (most appropriate diet and high r^2 for model)
	Fraction soil in diet	0.02[b]	Unitless	None	Default value
	Soil invertebrate diet	0.5[b]	Unitless	EPA 1993, 059384, p. 2-113	Rounded EPA value to 50% to equally expose receptor to potentially contaminated invertebrates and flesh
	Flesh diet	0.5[b]	Unitless	EPA 1993, 059384, p. 2-113	Rounded EPA value to 50% to equally expose receptor to potentially contaminated invertebrates and flesh
	Home range	106[b]	ha	EPA 1993, 059384	Average of all HR data for woods, forests, and agricultural areas
	Population area	4240[b]	ha	Calculated	40 times HR (see text for explanation)

Table 2.4-1 (continued)

Receptor	Parameter	Value	Units	Reference	Notes
Mexican spotted owl *Strix occidentalis lucida*	Body weight	0.539	kg	Dunning 2008, 601245	Mean of both sexes from one study
	Food intake[a]	0.0350	kg-food dry wt/kg-body wt/d	Weathers et al. 2001, 073476	Based on 0.059 kg-fresh-woodrat/d (for the spotted owl) multiplied by (100–68)% to account for moisture content
	Flesh diet	1	Unitless	None	Strict carnivore diet
	Home range	545	ha	Willey and Van Riper 2007, 601151	Mean breeding HR is 545 ha
Mountain cottontail *Sylvilagus nuttallii*	Body weight	0.792	kg	Sowls 1957, 602507	Average of reported values (used desert cottontail as a surrogate)
	Food intake[a]	0.0717	kg-food dry wt/kg-body wt/d	Nagy 2001, 253420	Estimated using 0.792 kg body weight and Nagy (2001, 253420) allometric scaling formula for herbivorous mammals (most appropriate diet and high r^2 for model)
	Fraction soil in diet	0.063[b]	Unitless	Arthur and Gates 1988, 602506	For black-tailed jackrabbit at Idaho National Laboratory
	Plant diet	1[b]	Unitless	EPA 1993, 059384, p. 2-356	Assume strict herbivore diet
	Home range	3.1	ha	EPA 1993, 059384, p. 2-357	Average of all HR data for a woodlot and for mixed habitats (used eastern cottontail as surrogate)
	Population area	124	ha	Calculated	40 times HR (see text for explanation)
Deer mouse *Peromyscus maniculatus*	Body weight	0.0163	kg	(Foxx 1995, 050039; Robinson and Bennett 2003, 082663; Bennett et al. 2006, 093701; Bennett and Robinson 2008, 106938)	Mean of body weights from Laboratory-specific studies (Los Alamos and Guaje baseline, Los Alamos and Pueblo Canyons, Mortandad Canyon, and Sandia Canyon)
	Food intake[a]	0.176	kg-food dry wt/kg-body wt/d	Nagy 2001,253420	Estimated using 0.0163 kg body weight and Nagy (2001, 253420) allometric scaling formula for omnivorous mammals (most appropriate diet and high r^2 for model)
	Fraction soil in diet	0.02[b]	Unitless	(Beyer et al. 1994, 062785, Table 1)	For white-footed mouse, most closely related of species available
	Plant diet	0.5[b]	Unitless	EPA 1993, 059384, p. 2-297	Rounded EPA value to 50% to equally expose receptor to potentially contaminated plants and invertebrates
	Soil invertebrate diet	0.5[b]	Unitless	EPA 1993, 059384, p. 2-297	Rounded EPA value to 50% to equally expose receptor to potentially contaminated plants and invertebrates
	Home range	0.4	ha	Carlsen et al. 2004, 601149	Average of relevant minimum HRs (0.16 or 0.63 ha)
	Population area	16	ha	Calculated	40 times HR (see text for explanation)

Table 2.4-1 (continued)

Receptor	Parameter	Value	Units	Reference	Notes
Montane shrew *Sorex monitcolus*	Body weight	0.0054	kg	(Bennett et al. 1999, 082652)	Average of 17 males and females from Sandia Canyon
	Food intake[a]	0.197	kg-food dry wt/kg-body wt/d	Nagy 2001, 253420	Estimated using 0.0054 kg body weight and Nagy (2001, 253420) allometric scaling formula for insectivorous mammals (most appropriate diet and high r^2 for model)
	Fraction soil in diet	0.009	Unitless	EPA 2007, 602500, Attachment 4-1, Table 3	Used median of the calculated soil intake for the shrew
	Soil invertebrate diet	1[b]	Unitless	EPA 1993, 059384, p. 2-214	Assume strict insectivore diet
	Home range	0.39[b]	ha	EPA 1993, 059384, p. 2-214	Reported average HR for one environment (used short-tailed shrew as surrogate for montane shrew)
	Population area	15.6[b]	ha	Calculated	40 times HR (see text for explanation)
Gray fox *Urocyon cinereoargenteus*	Body weight	4.54	kg	EPA 1993, 059384, p. 2-224	Mean of four values used to provide a representative value for a population (used red fox as a surrogate)
	Food intake[a]	0.0378	kg-food dry wt/kg-body wt/d	Nagy 2001, 253420	Estimated using 4.54 kg body weight and Nagy (2001, 253420) allometric scaling formula for carnivorous mammals (most appropriate diet and high r^2 for model)
	Fraction soil in diet	0.028[b]	Unitless	Beyer et al. 1994, 062785, Table 1	For red fox, surrogate for gray fox
	Flesh diet	1[b]	Unitless	EPA 1993, 059384, p. 2-224	Rounded diet to 100% flesh
	Home range	1038[b]	ha	EPA 1993, 059384, p. 2-226	Average of all HR data for the red fox over a variety of unspecified environments
	Population area	41,520[b]	ha	Calculated	40 times HR (see text for explanation)

Note: The document "ESL_EcoPRG_HistorySummary2017.pdf" describes all changes made to Table 2.4-1, and is available on the ECORISK Database website at http://www.lanl.gov/environment/protection/eco-risk-assessment.php.

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

[b] Parameter is the same as used to calculate ESLs.

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

Table 2.4-2
Assessment Population Areas for EcoPRG Non-T&E Wildlife Receptors

Taxonomic Group/ Trophic Level	Receptor	Receptor HR (ha)	Assessment Population Area (ha)*
Mammalian herbivore	Mountain cottontail	3.1	124
Mammalian invertivore	Deer mouse	0.4	16
Mammalian omnivore	Vagrant shrew	0.39	15.6
Avian herbivore	American robin	0.42	16.8
Avian invertivore	American robin	0.42	16.8
Avian omnivore	American robin	0.42	16.8
Middle trophic level	Median	0.42	17
	Mean	0.86	34
	Geo. mean	0.57	23
Mammalian carnivore	Gray fox	1038	41,500
Avian omnivore	American kestrel	106	4240
Upper trophic level	Median	570	23,000
	Mean	570	23,000
	Geo. mean	330	13,000

*The assessment population area (A_{pop}) is equal to 40 × HR.

Table 3.3-1
Aquatic Community Sandia Canyon Ecological Screening by Investigation Reach

COPEC	ESL[a]	L-ESL[a]	Maximum Detected Active-Channel Sediment Concentration[a]				
			PA-0	S-1S	S-2	S-3W	S-3E
Cadmium	0.99	4.9	—[b]	—	**8.69**	0.191	—
Chromium (total)	43	110	6.23	15	**3580**	85.2	88
Chromium (hexavalent)	na[c]	na	0.049	0.0588	2.01	0.308	0.429
Copper	31	140	2.79	8.4	**223**	12.4	9.1
Cyanide (total)	0.1	1	—	1.11	**8.77**	—	0.124
Lead	35	120	7.15	22	74.4	8.26	15
Mercury	0.18	1	—	0.38	**5.57**	0.0152	0.0073
Methylmercury	na	na	—	—	0.0046	—	—
Perchlorate	na	na	—	0.000749	0.000997	—	—
Selenium	0.72	2.9	2.33	1.33	**11.9**	**2.92**	2.47
Silver	0.5	5	0.0475	0.315	**87.3**	0.383	0.99
Thallium	na	na	0.105	—	1.06	0.0445	0.105
Zinc	120	450	41.7	76.9	**1140**	75.3	73
Aroclor-1016	0.059	0.59	—	—	—	—	—
Aroclor-1242	0.059	0.59	—	—	0.366	—	—
Aroclor-1248	0.059	0.59	—	—	—	—	—
Aroclor-1254	0.06	0.34	—	0.221	**3**	0.0239	—
Aroclor-1260	0.059	0.59	—	0.479	**2.3**	0.0215	0.073
Total PCBs	na	na	—	—	13.9	—	—

[a] Units are mg/kg; bolded maximum reach active-channel sediment concentrations are greater than the L-ESL.

[b] — = Not detected.

[c] na = Not available.

Table 5.0-1
Summary of COPC Data Paired with Soil Bioassays

COPC	Number of Samples	Number of Detects	Minimum Detect[a]	Maximum Detect[a]	Number of Nondetects
Antimony	41	4	0.0533	0.198	37
Arsenic	43	43	0.882	13.8	0
Barium	43	43	28.8	500	0
Beryllium	43	43	0.228	2.82	0
Cadmium	43	37	0.0655	6.18	6
Chromium (total)	43	43	2.85	3360	0
Chromium (hexavalent)	10	5	0.0727	4.71	5
Cobalt	43	43	1.52	7.58	0
Copper	43	43	3.69	199	0
Cyanide (total)	35	29	0.0763	6.74	6
Lead	43	43	5.76	244	0
Manganese	43	43	127	1560	0
Mercury	43	43	0.0045	1.71	0
Nickel	43	43	2.65	23.1	0
Selenium	43	34	0.186	15	9
Silver	43	40	0.04	49.4	3
Thallium	43	43	0.0429	3.27	0
Vanadium	43	43	6.96	48.5	0
Zinc	43	43	19.5	332	0
Acenaphthene	33	2	0.0477	0.163	31
Acenaphthylene	33	1	0.0268	0.0268	32
Anthracene	32	7	0.01	0.104	25
Aroclor-1016	43	0	n/a[b]	n/a	43
Aroclor-1242	43	0	n/a	n/a	43
Aroclor-1248	43	1	0.0335	0.0335	42
Aroclor-1254	43	27	0.0041	1.61	16
Aroclor-1260	43	34	0.0024	1.86	9
Benzo[a]anthracene	32	16	0.01	2.22	16
Benzo[a]pyrene	31	12	0.02	2.99	19
Benzo[b]fluoranthene	31	5	0.1	2.54	26
Benzo[g,h,i]perylene	31	9	0.01	1.58	22
Benzo[k]fluoranthene	33	4	0.03	1.2	29
Bis[2-ethylhexyl]phthalate	19	3	0.0997	1.31	16
Butylbenzylphthalate	19	0	n/a	n/a	19
Chrysene	32	18	0.01	2.78	14

Table 5.0-1 (continued)

COPC	Number of Samples	Number of Detects	Minimum Detect[a]	Maximum Detect[a]	Number of Nondetects
Di-n-butylphthalate	19	0	n/a	n/a	19
Di-n-octylphthalate	19	0	n/a	n/a	19
Dibenz[a,h]anthracene	33	1	0.11	0.11	32
Diethylphthalate	19	0	n/a	n/a	19
Dimethyl phthalate	19	0	n/a	n/a	19
Dinitrotoluene[2,4-]	19	0	n/a	n/a	19
Dinitrotoluene[2,6-]	19	0	n/a	n/a	19
Fluoranthene	33	23	0.0174	7.12	10
Fluorene	32	4	0.0328	0.879	28
Indeno[1,2,3-cd]pyrene	33	3	0.06	0.44	30
Methylnaphthalene[2-]	19	5	0.0082	0.0822	14
Naphthalene	33	0	n/a	n/a	33
Phenanthrene	33	22	0.00993	6.03	11
Pyrene	30	24	0.0173	6.01	6

[a] Units are mg/kg.

[b] n/a = Not applicable.

Table 5.1-1
Statistically Significant Gradient Analyses—
Linear Regression of Plant Bioassay versus Soil Chemistry

COPC	Bioassay Measure	r^2	n	Intercept	Slope	Prob>|t|
Aroclor-1254	Mean (mean shoot height [mm])	0.214	21	85.9	−24	0.035
Aroclor-1254	Mean (mean root length [mm])	0.371	21	63.3	−34	0.003
Aroclor-1260	Mean (mean shoot height [mm])	0.217	21	85.5	−21.6	0.034
Aroclor-1260	Mean (mean root length [mm])	0.391	21	62.8	−31.2	0.002
Arsenic	Mean (% germination)	0.211	21	101	−1.64	0.036
Arsenic	Mean (mean root length [mm])	0.356	21	77.5	−3.84	0.004
Chromium (total)	Mean (mean root length [mm])	0.275	21	60.7	−0.0148	0.015
Copper	Mean (% germination)	0.341	21	96.2	−0.125	0.005
Copper	Mean (mean shoot height [mm])	0.478	21	89.7	−0.249	0.001
Copper	Mean (mean root length [mm])	0.592	21	66.7	−0.298	<.0001
Copper	Mean (total dry weight [g]/plant)	0.285	21	0.00117	−0.00000209	0.013
Cyanide (total)	Mean (% germination)	0.358	21	95.8	−3.67	0.004
Cyanide (total)	Mean (mean shoot height [mm])	0.303	21	87.2	−5.68	0.01
Cyanide (total)	Mean (mean root length [mm])	0.459	21	64.5	−7.51	0.001
Cyanide (total)	Mean (total dry weight [g]/plant)	0.221	21	0.00116	−0.0000528	0.032
Lead	Mean (% germination)	0.219	21	100	−0.332	0.032
Lead	Mean (mean shoot height [mm])	0.366	21	99	−0.721	0.004
Lead	Mean (mean root length [mm])	0.551	21	80	−0.951	<.0001
Lead	Mean (total dry weight [g]/plant)	0.205	21	0.00124	−0.00000588	0.039
Mercury	Mean (mean root length [mm])	0.19	21	62	−20.9	0.049
Nickel	Mean (% germination)	0.257	21	101	−1.02	0.019
Silver	Mean (mean shoot height [mm])	0.306	21	88.4	−0.67	0.009
Silver	Mean (mean root length [mm])	0.244	21	63.4	−0.642	0.023
Silver	Mean (total dry weight [g]/plant)	0.212	21	0.00116	−0.00000607	0.036
Vanadium	Mean (mean root length [mm])	0.241	21	73.8	−0.9	0.024
Zinc	Mean (% germination)	0.359	21	99.3	−0.0881	0.004
Zinc	Mean (mean shoot height [mm])	0.323	21	93	−0.14	0.007
Zinc	Mean (mean root length [mm])	0.385	21	70.4	−0.165	0.003
Zinc	Mean (total dry weight [g]/plant)	0.201	21	0.0012	−0.00000121	0.041

Table 5.2-1
Statistically Significant Gradient Analyses–
Linear Regression of Earthworm Bioassay versus Soil Chemistry

COPC	Bioassay Measure	r^2	n	Intercept	Slope	Prob>|t|
Zinc	Earthworm % survival	0.232	21	90	−0.0432	0.027
Benzo[a]anthracene	Earthworm % survival	0.483	21	99.9	−0.915	0.0005
Benzo[a]pyrene	Earthworm % survival	0.466	21	99.9	−0.664	0.0006
Benzo[b]fluoranthene	Earthworm % survival	0.408	21	99.9	−0.719	0.0018
Benzo[g,h,i]perylene	Earthworm % survival	0.446	21	99.9	−1.23	0.0009
Chrysene	Earthworm % survival	0.477	21	99.9	−0.728	0.0005
Fluoranthene	Earthworm % survival	0.503	21	99.9	−0.293	0.0003
Phenanthrene	Earthworm % survival	0.528	21	99.8	−0.353	0.0002
Pyrene	Earthworm % survival	0.489	21	99.9	−0.34	0.0004
Selenium	Weight change per worm (g)	0.206	21	−0.199	0.0319	0.039

Table 5.3-1
Summary of Seedling Germination Bioassay COPC Dose-Response Results

COPC	Dose-Response Notes	Site-Specific NOEC (mg/kg)
Antimony	Insufficient detections for statistical evaluation	—*
Arsenic	Negative slope for one of two laboratories; overall results are discordant and dose response is not indicated.	13.8
Barium	No statistical relationship	500
Beryllium	No statistical relationship	2.82
Cadmium	No statistical relationship	6.18
Chromium (total)	Negative slope for one of two laboratories; overall results are discordant and dose response is not indicated.	3360
Chromium (hexavalent)	No statistical relationship	4.71
Cobalt	No statistical relationship	7.58
Copper	Negative slope for one of two laboratories; overall results are discordant and dose response is not indicated.	199
Cyanide (total)	Negative slope for one of two laboratories; overall results are discordant and dose response is not indicated.	6.74
Lead	Negative slope for one of two laboratories; overall results are discordant and dose response is not indicated.	244
Manganese	No statistical relationship	1560
Mercury	Negative slope for one of two laboratories; overall results are discordant and dose response is not indicated.	1.71
Nickel	No statistical relationship	23.1
Selenium	No statistical relationship	15
Silver	Negative slope for one of two laboratories; overall results are discordant and dose response is not indicated.	49.4
Thallium	No statistical relationship	3.27
Vanadium	Negative slope for one of two laboratories; overall results are discordant and dose response is not indicated.	48.5
Zinc	Negative slope for one of two laboratories; overall results are discordant and dose response is not indicated.	332
Acenaphthene	Insufficient detections for statistical evaluation	—
Acenaphthylene	Insufficient detections for statistical evaluation	—
Anthracene	No statistical relationship	0.104
Aroclor-1016	Insufficient detections for statistical evaluation	—
Aroclor-1242	Insufficient detections for statistical evaluation	—
Aroclor-1248	Insufficient detections for statistical evaluation	—
Aroclor-1254	Negative slope for one of two laboratories; overall results are discordant and dose response is not indicated.	1.61
Aroclor-1260	Negative slope for one of two laboratories; overall results are discordant and dose response is not indicated.	1.86
Benzo[a]anthracene	No statistical relationship	2.22

Table 5.3-1 (continued)

COPC	Dose-Response Notes	Site-Specific NOEC (mg/kg)
Benzo[a]pyrene	No statistical relationship	2.99
Benzo[b]fluoranthene	No statistical relationship	2.54
Benzo[g,h,i]perylene	No statistical relationship	1.58
Benzo[k]fluoranthene	Insufficient detections for statistical evaluation	—
Bis[2-ethylhexyl]phthalate	Insufficient detections for statistical evaluation	—
Butylbenzylphthalate	Insufficient detections for statistical evaluation	—
Chrysene	No statistical relationship	2.78
Di-n-butylphthalate	Insufficient detections for statistical evaluation	—
Di-n-octylphthalate	Insufficient detections for statistical evaluation	—
Dibenz[a,h]anthracene	Insufficient detections for statistical evaluation	—
Diethylphthalate	Insufficient detections for statistical evaluation	—
Dimethyl phthalate	Insufficient detections for statistical evaluation	—
Dinitrotoluene[2,4-]	Insufficient detections for statistical evaluation	—
Dinitrotoluene[2,6-]	Insufficient detections for statistical evaluation	—
Fluoranthene	No statistical relationship	7.12
Fluorene	Insufficient detections for statistical evaluation	—
Indeno[1,2,3-cd]pyrene	Insufficient detections for statistical evaluation	—
Methylnaphthalene[2-]	No statistical relationship	0.0822
Naphthalene	Insufficient detections for statistical evaluation	—
Phenanthrene	No statistical relationship	6.03
Pyrene	No statistical relationship	6.01

*— = Less than five detections; insufficient data to evaluate a dose-response relationship.

Table 5.3-2
Summary of Earthworm Bioassay COPC Dose-Response Results

COPC	Dose-Response Notes	Site-Specific NOEC (mg/kg)
Antimony	Insufficient detections for statistical evaluation	—[a]
Arsenic	No statistical relationship	13.8
Barium	No statistical relationship	500
Beryllium	No statistical relationship	2.82
Cadmium	No statistical relationship	6.18
Chromium (total)	No statistical relationship	3360
Chromium (hexavalent)	No statistical relationship	4.71
Cobalt	No statistical relationship	7.58
Copper	No statistical relationship	199
Cyanide (total)	No statistical relationship	6.74
Lead	No statistical relationship	244
Manganese	No statistical relationship	1560
Mercury	No statistical relationship	395[b]
Nickel	No statistical relationship	23.1
Selenium	No statistical relationship	15
Silver	No statistical relationship	49.4
Thallium	No statistical relationship	3.27
Vanadium	No statistical relationship	48.5
Zinc	One statistical relationship, but difference of 1% in survival is not biologically significant; dose response is not indicated.	332
Acenaphthene	Insufficient detects for statistical evaluation	—
Acenaphthylene	Insufficient detects for statistical evaluation	—
Anthracene	No statistical relationship	0.104
Aroclor-1016	Insufficient detections for statistical evaluation	—
Aroclor-1242	Insufficient detections for statistical evaluation	—
Aroclor-1248	Insufficient detections for statistical evaluation	—
Aroclor-1254	No statistical relationship	1.61
Aroclor-1260	No statistical relationship	1.86
Benzo[a]anthracene	One statistical relationship, but difference of 1% in survival is not biologically significant; dose response is not indicated.	2.22
Benzo[a]pyrene	One statistical relationship, but difference of 1% in survival is not biologically significant; dose response is not indicated.	2.99
Benzo[b]fluoranthene	One statistical relationship, but difference of 1% in survival is not biologically significant; dose response is not indicated.	2.54
Benzo[g,h,i]perylene	One statistical relationship, but difference of 1% in survival is not biologically significant; dose response is not indicated.	1.58
Benzo[k]fluoranthene	Insufficient detections for statistical evaluation	—
Bis[2-ethylhexyl]phthalate	Insufficient detections for statistical evaluation	—
Butylbenzylphthalate	Insufficient detections for statistical evaluation	—

Table 5.3-2 (continued)

COPC	Dose-Response Notes	Site-Specific NOEC (mg/kg)
Chrysene	One statistical relationship, but difference of 1% in survival is not biologically significant; dose response is not indicated.	2.78
Di-n-butylphthalate	Insufficient detections for statistical evaluation	—
Di-n-octylphthalate	Insufficient detections for statistical evaluation	—
Dibenz[a,h]anthracene	Insufficient detections for statistical evaluation	—
Diethylphthalate	Insufficient detections for statistical evaluation	—
Dimethyl phthalate	Insufficient detections for statistical evaluation	—
Dinitrotoluene[2,4-]	Insufficient detections for statistical evaluation	—
Dinitrotoluene[2,6-]	Insufficient detections for statistical evaluation	—
Fluoranthene	One statistical relationship, but difference of 1% in survival is not biologically significant; dose response is not indicated.	7.12
Fluorene	Insufficient detections for statistical evaluation	—
Indeno[1,2,3-cd]pyrene	Insufficient detections for statistical evaluation	—
Methylnaphthalene[2-]	No statistical relationship	0.0822
Naphthalene	Insufficient detections for statistical evaluation	—
Phenanthrene	One statistical relationship, but difference of 1% in survival is not biologically significant; dose response is not indicated.	6.03
Pyrene	One statistical relationship, but difference of 1% in survival is not biologically significant; dose response is not indicated.	6.01

[a] — = Less than five detections; insufficient data to evaluate a dose-response relationship.

[b] Canyons investigations plus study conducted for SWMU 32-002(b2).

Table 7.0-1
Soil EcoPRGs Calculated for a Hypothetical 1-ha Site

COPC	Gray Fox	Mexican Spotted Owl	American Kestrel (flesh/invert diet)	American Robin (plant diet)	American Robin (invert/plant diet)	American Robin (invert diet)	Mountain Cottontail	Montane Shrew	Deer Mouse	Earthworm	Plant	Final Soil EcoPRG	Final Soil EcoPRG Receptor
Tetrachlorodibenzodioxin[2,3,7,8-]	18	na[a]	na	na	na	na	0.0049	0.000032	0.00007	10	na	0.000032	Montane Shrew
Amino-2,6-dinitrotoluene[4-]	1.8E+09	na	na	na	na	na	5.9E+04	2000	4200	180	330	180	Earthworm
Amino-4,6-dinitrotoluene[2-]	2.6E+09	na	na	na	na	na	2.0E+04	2600	4300	430	140	140	Generic Plant
Dinitrotoluene[2,4-]	5.6E+08	na	na	na	na	na	1.3E+04	2400	3700	180	60	60	Generic Plant
Dinitrotoluene[2,6-]	3.6E+08	4.3E+07	2.0E+08	1.8E+04	2.9E+04	6.1E+04	1200	1200	740	44	na	44	Earthworm
HMX[b]	4.3E+09	na	na	na	na	na	2.0E+04	5.0E+04	1.4E+04	160	3500	160	Earthworm
PETN[c]	1.2E+10	na	na	na	na	na	2.3E+04	1.8E+05	1.8E+04	na	na	1.8E+04	Deer Mouse
RDX[d]	6.0E+08	1.8E+06	6.7E+05	150	170	200	2200	870	940	15	360	15	Earthworm
Tetryl	1.2E+08	na	na	na	na	na	160	5700	130	na	na	130	Deer Mouse
Trinitrobenzene[1,3,5-]	2.8E+09	na	na	na	na	na	2.7E+04	1.5E+05	2.1E+04	28	na	28	Earthworm
Trinitrotoluene[2,4,6-]	3.3E+09	7.2E+06	7.2E+07	490	1000	17000	9900	1.9E+05	8000	58	120	58	Earthworm
Antimony	1.2E+09	na	na	na	na	na	4.7E+04	2.7E+05	40000	780	58	58	Generic Plant
Arsenic	2.0E+08	1.7E+06	3.2E+07	2.0E+04	1.1E+04	9100	1.9E+04	3200	5400	68	91	68	Earthworm
Barium	5.3E+09	5.7E+07	4.0E+08	6.2E+04	7.7E+04	1.0E+05	2.5E+05	1.9E+05	1.5E+05	3200	1400	1400	Generic Plant
Beryllium	1.1E+08	na	na	na	na	na	1.6E+04	8000	1.0E+04	400	25	25	Generic Plant
Boron	5.9E+09	2.2E+06	5.6E+06	350	610	1600	1.5E+04	2.2E+04	1.0E+04	na	86	86	Generic Plant
Cadmium	2.0E+08	9.9E+05	2.2E+05	910	110	70	2500	58	120	760	160	58	Montane Shrew
Chromium (hexavalent)	1.2E+09	8.3E+06	4.4E+08	1.6E+05	1.2E+05	1.0E+05	1.8E+05	6.9E+04	1.0E+05	4.7	4.7	4.7	Earthworm
Chromium (total)	5.0E+09	2.0E+06	1.6E+07	1.2E+04	6100	4500	7.5E+05	1.1E+05	2.0E+05	na	na	4500	A. Robin (invert diet)
Cobalt	3.9E+08	5.3E+06	4.2E+07	2.1E+04	1.3E+04	1.1E+04	5.1E+04	1.1E+04	1.9E+04	na	130	130	Generic Plant
Copper	3.0E+09	2.7E+06	2.1E+07	1.4E+04	8200	6400	1.3E+05	1.9E+04	3.2E+04	530	490	490	Generic Plant
Cyanide (total)	9.0E+08	1300	1.0E+05	37	40	46	1.4E+05	5.5E+04	6.1E+04	na	na	37	A. Robin (plant diet)
Lead	2.9E+09	8400000	1.6E+07	6100	5000	4600	1.7E+05	4.7E+04	6.5E+04	8400	570	570	Generic Plant
Manganese	4.1E+09	130000000	1.5E+09	1.2E+05	1.8E+05	3.7E+05	1.3E+05	2.2E+05	9.9E+04	4500	1500	1500	Generic Plant
Mercury (inorganic)	9.8E+07	58000	2.1E+05	41	51	68	4300	2300	2100	390	64	41	A. Robin (plant diet)
Mercury (methyl)	2.0E+04	20	4.6E+02	32	0.26	0.15	170	0.25	0.56	12	na	0.15	A. Robin (invert diet)
Nickel	7.6E+08	4700000	1.3E+07	3.5E+04	6300	3900	1.1E+05	3900	8200	1300	270	270	Generic Plant
Selenium	1.7E+07	170000	8.0E+05	270	240	240	290	80	100	41	15	15	Generic Plant
Silver	1.2E+09	1400000	3.9E+06	4300	1600	1100	2.9E+04	2300	4400	na	2800	1100	A. Robin (invert diet)
Thallium	1.3E+06	230000	1.4E+07	5300	4100	3600	230	91	130	na	3.2	3.2	Generic Plant
Uranium	3.2E+08	60000000	4.3E+09	1.1E+06	1.0E+06	9.8E+05	4.6E+04	2.9E+04	3.3E+04	na	250	250	Generic Plant
Vanadium	1.8E+08	260000	1.5E+07	4700	3900	3700	2.8E+04	1.3E+04	1.8E+04	na	80	80	Generic Plant
Zinc	2.5E+09	6100000	1.7E+07	3.5E+04	8600	5500	3.3E+05	1.5E+04	3.1E+04	930	810	810	Generic Plant
Acenaphthene	7.9E+09	na	na	na	na	na	9.6E+04	2.1E+04	3.0E+04	na	2.5	2.5	Generic Plant

COPC	Gray Fox	Mexican Spotted Owl	American Kestrel (flesh/invert diet)	American Robin (plant diet)	American Robin (invert/plant diet)	American Robin (invert diet)	Mountain Cottontail	Montane Shrew	Deer Mouse	Earthworm	Plant	Final Soil EcoPRG	Final Soil EcoPRG Receptor
Acenaphthylene	7.7E+09	na	na	na	na	na	9.8E+04	2.0E+04	2.9E+04	na	na	2.0E+04	Montane Shrew
Anthracene	1.0E+10	na	na	na	na	na	2.2E+05	3.5E+04	5.5E+04	na	8.9	8.9	Generic Plant
Benzo(a)anthracene	3.1E+07	6.6E+04	1.9E+06	310	390	530	1100	730	620	na	180	180	Generic Plant
Benzo(a)pyrene	3.0E+08	na	na	na	na	na	1.5E+04	3300	4800	na	na	3300	Montane Shrew
Benzo(b)fluoranthene	6.6E+08	na	na	na	na	na	2.3E+04	7500	9400	na	180	180	Generic Plant
Benzo(g,h,i)perylene	9.8E+08	na	na	na	na	na	8.6E+04	4200	8500	na	na	4200	Montane Shrew
Benzo(k)fluoranthene	1.1E+09	na	na	na	na	na	6.0E+04	1.1E+04	1.8E+04	na	na	1.1E+04	Montane Shrew
Chrysene	3.0E+07	na	na	na	na	na	1100	550	560	na	na	550	Montane Shrew
Dibenzo(a,h)anthracene	2.3E+08	na	na	na	na	na	1.5E+04	2500	4000	na	na	2500	Montane Shrew
Fluoranthene	1.0E+09	na	na	na	na	na	4.9E+04	3700	6900	23	na	23	Earthworm
Fluorene	2.7E+09	na	na	na	na	na	4.3E+04	8300	12000	19	na	19	Earthworm
Indeno(1,2,3-cd)pyrene	1.2E+09	na	na	na	na	na	9.2E+04	1.1E+04	2.0E+04	na	na	1.1E+04	Montane Shrew
Methylnaphthalene[2-]	1.3E+09	na	na	na	na	na	2.0E+04	2600	4400	na	na	2600	Montane Shrew
Naphthalene	4.4E+08	5.3E+07	2.4E+07	1300	2300	7600	730	1200	490	na	10	10	Generic Plant
Phenanthrene	5.3E+08	na	na	na	na	na	1.1E+04	1800	2800	12	na	12	Earthworm
Pyrene	8.5E+08	7.0E+06	4.9E+07	2.6E+04	1.8E+04	1.5E+04	2.1E+04	3700	5700	20	na	20	Earthworm
Aroclor-1016	1.9E+07	na	na	na	na	na	2500	50	100	na	na	50	Montane Shrew
Aroclor-1242	1.1E+07	1.4E+04	5.8E+04	430	30	17	2000	25	54	na	na	17	A. Robin (invert diet)
Aroclor-1248	5.2E+05	4.2E+05	1.7E+05	1300	89	52	98	1.1	2.5	na	na	1.1	Montane Shrew
Aroclor-1254	1.9E+06	2.5E+05	8.4E+04	860	44	25	4400	40	88	na	620	25	A. Robin (invert diet)
Aroclor-1260	4.2E+06	9.2E+05	1.7E+05	3400	94	54	8.2E+04	390	880	na	na	54	A. Robin (invert diet)
Bis(2-ethylhexyl)phthalate	1.3E+08	2.1E+04	2.8E+04	9900	15	8.7	3.6E+05	96	210	na	na	8.7	A. Robin (invert diet)
Butylbenzylphthalate	6.4E+09	na	na	na	na	na	4.4E+05	1.4E+04	3.0E+04	na	na	1.4E+04	Montane Shrew
Di-n-butylphthalate	4.0E+09	4600	1.5E+04	140	8.2	4.8	7.3E+05	7200	1.5E+04	na	600	4.8	A. Robin (invert diet)
Di-n-octylphthalate	3.7E+08	na	na	na	na	na	1.5E+06	140	330	na	na	140	Montane Shrew
Diethylphthalate	6.8E+11	na	na	na	na	na	1.6E+06	5.9E+05	6.6E+05	na	1000	1000	Generic Plant
Dimethyl phthalate	1.3E+10	na	na	na	na	na	1.1E+04	1.3E+04	6900	100	na	100	Earthworm

Note: Units are mg/kg.

[a] na = Not available.

[b] HMX = Octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine.

[c] PETN = Pentaerythritol tetranitrate.

[d] RDX = Hexahydro-1,3,5-trinitro-1,3,5-triazine.

Attachment 1

Linear Regression Model
(on CD included with this document)