Final Baseline Ecological Risk Assessment for Upland Terrestrial Habitat

Upper Columbia River Site

Stevens County, Washington

Prepared for:

United States Environmental Protection Agency – Region 10

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Executive Summary

Introduction

This document presents the final baseline ecological risk assessment (BERA) for the upland terrestrial habitat of the Upper Columbia River site (UCR site or site) in northeast Washington State. This document is referred to as the "Upland BERA" and has been prepared by the U.S. Environmental Protection Agency (EPA) pursuant to the Settlement Agreement between Teck American Incorporated (TAI) and EPA (EPA, 2006), the UCR Remedial Investigation (RI)/Feasibility Study (FS) Work Plan (EPA, 2008), and EPA comments on the draft interim Partial Upland BERA (EPA, 2022a) and draft final Upland BERA (EPA, 2023) documents prepared by TAI (TAI, 2021, 2023). EPA assumed responsibility for finalization of the Upland BERA in May 2023 (EPA, 2023) as part of a work takeover in accordance with the Settlement Agreement.

The upland portion of the site[1] is designated as Operable Unit 3 (OU3) in the RI/FS Work Plan Schedule Addendum (EPA, 2022b). Though it has yet to be fully defined, the upland area is commonly described as land above the elevations of historical Columbia River flood events and within the approximate footprint of metals deposition associated with historical smelter aerial emissions. For the purposes of the Upland BERA, the upland area is operationally defined as the spatial extent of the upland soil data set used for ecological risk analysis and is termed the "Terrestrial Study Area." The actual spatial extent of OU3 is expected to be established by analyses presented in the Draft Final Upland RI Report (TAI, 2023), which is currently under EPA review.

Consistent with the objectives described in the RI/FS Work Plan (EPA, 2008) and the UCR BERA Work Plan (referred to herein as the "BERA Work Plan") (TAI, 2011), the objective of the Upland BERA is to assess risk from hazardous substances in soils to ecological assessment endpoints (EAEs) within the Terrestrial Study Area of the site, under both current conditions and expected future conditions assuming no steps are taken to remediate the environment.

The Upland BERA identifies which hazardous substances, exposure pathways, exposure media, and ecological receptors are associated with unacceptable risks to EAEs and provides a basis for informed discussions with risk managers about the causes, nature, and extent of any such risks.

Hazardous substances that might pose risks to EAEs are called chemicals of potential concern (COPCs). The Upland BERA evaluates whether COPCs identified during the previously conducted screening-level risk assessments (SLERAs) present unacceptable risk of adverse effects to representative ecological receptor communities or populations selected as EAEs. COPCs determined in this BERA to pose unacceptable risk to EAEs are identified as ecological chemicals of concern (COCs). The distribution of ecological and human health COCs[2] in the upland portion of the site is further evaluated in the Upland RI Report and protection of both human and ecological receptors will be considered in the development of remedial action objectives and preliminary remediation goals in the Upland FS.

[1] As defined within the Settlement Agreement of June 2, 2006, the site consists of the areal extent of hazardous substances contamination within the United States in or adjacent to the Upper Columbia River, including the Franklin D. Roosevelt Lake, from the U.S.-Canada border to the Grand Coulee Dam, and all suitable areas in proximity to the contamination necessary for implementation of response actions.

[2] Human health COCs are identified in the final sitewide Human Health Risk Assessment (EPA, 2021)

Summary of Previous Screening-Level Evaluations

This Upland BERA builds upon screening level- risk evaluations done previously for the site, which were designed to focus the list of COPCs evaluated in the Upland BERA. Specifically, these include the SLERA (TAI, 2010) and the COPC refinement (TAI, 2020b).[3]

The SLERA (TAI, 2010) was conducted following procedures described in EPA's ecological risk assessment (ERA) Guidance for Superfund (EPA, 1997). The primary purpose of the SLERA was to determine whether there were adequate data to make a determination of the potential for unacceptable risks posed to ecological receptors by chemicals of interest (COIs) in environmental media (water, sediment, and soil) at the site. The SLERA also reduced the number of COIs through the use of conservative, risk-based assumptions; if adequate data were available for COI benchmarks and environmental media concentrations, and if risks were determined to be acceptable, then no further assessment was warranted. If adequate data were not available, or if the available information indicated that the COI may pose an unacceptable risk, then further evaluation of those COIs was required.

Following the SLERA, a refined screening-level analysis for the selection of COPCs for the site was conducted (TAI, 2020b). This COPC refinement satisfied the EPA requirement in its dispute resolution letter (EPA, 2013) regarding the Phase 2 Sediment Study Quality Assurance Project Plan, namely that the refinement would include and evaluate any and all EPA-approved data as collected for the RI/FS to date [and] "… refine the assumptions and methods used in the SLERA."

The objectives of the COPC refinement were to identify COPCs to be carried forward to the Upland BERA, COIs to be eliminated from further evaluation based on exposure concentrations less than benchmarks, and COIs to be retained for the Upland BERA because of uncertainty (for example, no benchmarks were available or there were insufficient exposure measurements). COPCs and COIs (that could not be screened in the COPC refinement) include aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, molybdenum, nickel, selenium, silver, thallium, vanadium, and zinc.

These COPCs and COIs identified in the screening level evaluations are further evaluated in the Upland BERA and for simplicity all are referred to as COPCs. Most of the COPCs are metals, but also include two metalloids (arsenic and antimony) and a nonmetal (selenium). For simplicity, all COPCs are collectively referred to as "metals."

Media and Spatial Limitations

This final Upland BERA focuses solely on upland soils and does not address risks in any of the upland aquatic habitats (streams, lakes, and or wetlands) that are present in the Terrestrial Study Area. Risks to ecological receptors in upland aquatic habitats within the Terrestrial Study Area will be assessed in an addendum to this Upland BERA after the upland lakes/wetlands sampling and analysis program is completed.

Additionally, the data evaluated in this BERA do not include riparian soil and beach sediment samples from locations within historically flooded areas adjacent to the Columbia River. Because these areas may have been impacted by river-borne contaminants, ecological risks will be evaluated in the Aquatic BERA for the UCR site. A map depicting pre-1973 maximum flood extent topographic elevations, as defined for

[3] The COPC refinement (TAI, 2019b) included an addendum that was finalized in January 2020 (TAI, 2020b). This citation, (TAI, 2020b), is used in the remainder of the BERA to cite the COPC refinement document.

the UCR Final Soil Study Quality Assurance Project Plan (TAI, 2014a), was used to determine the areas and data evaluated in the Upland BERA and those that will be evaluated in the Aquatic BERA.

Summary of Site and Terrestrial Study Area Characteristics

The overall UCR site is located in northeastern Washington State and encompasses approximately 150 river miles (RMs) along the Columbia River, from the U.S.-Canada border to Grand Coulee Dam. Grand Coulee Dam impounds Lake Roosevelt, a large reservoir extending approximately 133 RMs upstream of the dam at full pool. The Columbia River is free flowing upstream of Lake Roosevelt to the U.S.-Canada border. The upland portion of the UCR site that makes up the Terrestrial Study Area is commonly defined as land above the elevations of historical Columbia River flood events and within the approximate footprint of metals deposition associated with historical smelter aerial emissions (operationally defined as the spatial extent encompassed by existing upland soil data).

The Terrestrial Study Area is sparsely populated and includes federal, state, and private forest lands and farmland. The northern end of the site is largely undeveloped, and there is partial overlap with the Colville National Forest to the west and northeast (EPA, 2008). The region is known to have bedrock formations that, in some areas, are enriched with metal ores, and there are placer deposits in the UCR and its tributaries. Mining, milling, smelting, and prospecting have occurred in this region since the mid-1800s (EPA, 2008).

The Terrestrial Study Area is located in the Okanogan Highlands province of northeastern Washington. Elevations in the Terrestrial Study Area range from 1,290 feet above mean sea level (amsl) (the full-pool elevation of Lake Roosevelt) to more than 4,700 feet amsl. Topography consists of glaciofluvial terraces adjacent to the UCR, tributaries to the UCR, and surrounding hills, mountains, and ridges to the east and west that are characterized by variable terrain and elevations. The terraces and stream-cut scarps, with varying slopes from gentle to steep, have also been modified by gullying, land-sliding, and creep (Jones et al., 1961).

The hydrological setting for the overall UCR site includes the Upper Columbia River and the areas it drains between the U.S.-Canada border and Grand Coulee Dam, including its major tributaries. River flows are controlled by the operation of upstream dams in addition to operations at Grand Coulee Dam. A regulated flow regime has been in effect on the Columbia River since construction of Grand Coulee Dam was completed in 1942 and other upstream dams in Canada were constructed and began operating in the 1970s. In general, reservoir elevation is reduced from January to April to prevent flooding during spring runoff; allowed to increase during May and June; allowed to decrease slightly through July and August, when recharge is diminished; and allowed to increase from September to December as a response to increased autumn precipitation.

The Terrestrial Study Area encompasses land within Washington's Water Resource Inventory Areas (WRIAs)[4] 61 and the northwestern part of WRIA 62. Major WRIA 61 drainages within the Terrestrial Study Area include Cedar Creek, Deep Creek, Quartz Creek, Sheep Creek, Onion Creek, Rattlesnake Creek, Crown Creek, and Flat Creek, all of which discharge into the Columbia River. Terrestrial Study Area drainages in WRIA 62 discharge to the Pend Oreille River.

Surficial geology of the Terrestrial Study Area is primarily composed of the following: glacial fluvial and lacustrine sediments and till; recent unconsolidated fluvial deposits, alluvium, and colluvium; Paleozoic

[4] WRIAs are areas established in the state of Washington to be used as the basis for natural resource management purposes in the state. WRIA boundaries are based on natural watershed and drainage characteristics. The geographic definitions for WRIAs across the state were formalized under Washington Administrative Code (WAC) 173-500-040.

metasedimentary, metacarbonate, and metavolcanic rocks; and Eocene igneous intrusive and extrusive rocks. Localized mineralization in the Terrestrial Study Area is primarily associated with Paleozoic metasedimentary, metacarbonate, and metavolcanic rocks.

The Terrestrial Study Area is predominantly rural and forest lands. The area is largely undeveloped and there is partial overlap with the Colville National Forest to the west and the northeast. Land uses include management of federal, state, and private forest lands; residential, agricultural production; outdoor recreation; and cultural uses, with some commercial and retail services in established towns and crossroads. Timber harvesting is prevalent in the area, occurring mostly on private forest land with a small percentage of trees harvested from state and federal lands.

Ore mining and mineral processing has been occurring in the UCR region since at least the late 1800s. Most of the operations in the U.S. took place in Stevens and Ferry counties (Orlob and Saxton, 1950; Wolff et al., 2005). Mining activities in the drainage basin also occurred in the Metaline mining district. Most of these operations became inactive or were closed in the early to mid-20th century. Only the Teck Smelter in Trail, British Columbia is known to be currently active.

The climate of the forested areas is characterized by warm, dry summers and moist, cold winters with heavy snowfall. Grasslands tend to experience hotter, drier climatic conditions. Wildfires may occur seasonally, and fire intensity may vary depending on drought conditions and temperature. Wind direction distributions have strong seasonal variation and are affected by local topographic conditions, and winds may flow in directions different from prevailing winds. Diurnal variations in slope and valley winds are also likely to occur in the Columbia River valley.

Known and Potential Sources of Chemicals of Potential Concern

The Upland BERA evaluates risk from metals (that is, COPCs) that have been introduced by one or more of the sources noted in this section, without attributing any particular source(s) to COPCs. The source activities all present varying degrees of potential stress on natural habitats in the Terrestrial Study Area, and in many locations there may be multiple stressors (chemical and/or physical).

Broadly, the sources of COPCs to the Terrestrial Study Area can be classified into the following categories by order of magnitude and importance: (1) historical smelter operations, (2) mining and mining-related operations, (3) other anthropogenic sources, (4) geology and local mineralization (geogenic sources), and (5) ambient atmospheric constituents.

Ecological Setting

The Terrestrial Study Area is characterized by dominantly coniferous forest and montane grasslands. Natural areas, consisting of public and private forest and parks/undeveloped areas are interspersed with towns, residential neighborhoods, residences on large rural parcels, roads, railroad and utility rights-of-way, agriculture, retail, other services, manufacturing, and land designated for cultural or recreational use. Natural areas comprise most of the Terrestrial Study Area.

Ecosystems of the lower-elevation shorelines and terraces adjacent to the Columbia River are a mix of grasslands, shrub lands, savanna, and forest. Ecosystems of the higher-elevation terrain are predominantly conifer forest, pine woodlands, and savanna, with some grasslands.

Approximately 100 species of upland mammals and 250 species of birds are potentially present in the Terrestrial Study Area. Four amphibian and seven reptile species might occur within the Terrestrial Study Area. The only federally and/or state-listed endangered or threatened species reported in the Terrestrial Study Area is the gray wolf (*Canis lupus*).

Summary of the Upland BERA Approach

The approach for the Upland BERA was described in the BERA Work Plan (TAI, 2011) and expanded problem formulation (TAI, 2012). The methodology follows EPA's guidance for ecological risk assessment, including Superfund (EPA 1992, 1997, 1998) and other relevant guidance documents. The Upland BERA incorporates the five principles of the Framework for Metals Risk Assessment (EPA, 2007a) (e.g., incorporating background in risk characterization and evaluating soil characteristics that influence metal bioaccessibility and bioavailability).

An important aspect of the Upland BERA is the identification of valued aspects of the site ecology that are susceptible to contamination and should be protected through risk analysis and remedial management. These values are evaluated in the Upland BERA using specific EAEs which include an ecological entity and its attributes to be protected. Based on the BERA planning documents, including the BERA Work Plan (TAI, 2011) and the expanded problem formulation (TAI, 2012), as well as EPA's ecological risk assessment guidance (EPA 1997, 1998), the environmental values identified for protection include plants, soil invertebrates, and wildlife (birds, mammals, reptiles, and amphibians).[5] These receptor groups are the entities to be protected. The aspects of these entities are the plant and invertebrate communities and the wildlife populations, with the exception of the threatened and endangered species EAE (gray wolf), for which protection of individual organisms was identified. For birds and mammals, populations of representative species in five feeding guilds were selected as EAEs:

- Herbivore populations: California quail and meadow vole
- Omnivore populations: black-capped chickadee and deer mouse
- Invertivore populations: American robin and masked shrew
- Aerial-feeding insectivore populations: Tree swallow and little brown bat
- Carnivore populations: American kestrel, short-tailed weasel, and gray wolf (individuals)

The types of data used in this Upland BERA include surficial soil chemistry data, chemical bioaccessibility data, chemical bioaccumulation data, and toxicological effects data. These data are incorporated into models used in the Upland BERA to estimate the bioavailable fraction of COPC concentrations in soils, COPC concentrations in plants and prey items consumed by wildlife, dietary doses of COPCs to the wildlife receptors, and toxicological effects. The soil chemistry data set is composed of data collected during three studies: aerial deposition area (ADA) samples collected during the 2014 UCR Upland Soil Study (TAI, 2015)[6], soil data collected for the 2012 Ecology Upland Soil Study (Ecology, 2013), and the 2015 Bossburg Study (TAI, 2016). In the three soil studies, soil samples were analyzed for EPA's target analyte list metals (aluminum, antimony, arsenic, barium, beryllium, cadmium, calcium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, potassium, selenium, silver, sodium, thallium, vanadium, and zinc), plus molybdenum (2014 UCR Upland Soil Study only).

In addition to metal concentration data, data were collected from the Terrestrial Study Area and compiled from the literature to determine the availability of metals for uptake by plants and animals. Plant tissue sample data and co-located soil data from the UCR Final Plant Tissue Study Data Summary Report (referred to herein as the "2018 Plant Tissue Study") (TAI, 2019a) were used to model COPCs

[5] Risk to reptiles and amphibians will be assessed in an addendum to this Upland BERA after the upland lakes/wetlands sampling and analysis program is completed.

[6] Although originally assigned to the ADA, DU ADA-140, located in the northern part of the site, adjacent to the western shoreline of the UCR, has approximately 80 percent of its spatial extent below the pre-1973 maximum flood extent elevation, so will be evaluated in the Aquatic BERA.

concentrations in plant tissue consumed by wildlife. COPC concentrations in prey tissue (e.g., invertebrates, small mammals) were modeled using bioaccumulation data reported in scientific literature.

For plants and invertebrates, exposure is assumed to be based on direct contact with soil (i.e., plants absorb metals from soil directly through their roots, and invertebrates through dermal contact and ingestion). In this approach, toxicological effects concentration thresholds for metals in soil (referred to as benchmarks in the Upland BERA) developed from laboratory-based toxicity experiments are compared to measured soil COPC concentration data to predict the potential for adverse effects. Two benchmark types are used in the Upland BERA. The first type is conservative soil screening-level benchmarks (Eco-SSLs developed by EPA to be protective of effects from contaminants in soils nationally or SSLs developed for the Upland BERA using the same methodology). The second type, referred to as bioavailability-adjusted benchmarks (BABs), consist of soil COPC concentrations adjusted to account for the effect of site soil properties on bioavailability, in cases where the data are available to do so. If a BAB is unavailable for a particular COPC, then only SSLs are used to evaluate that COPC, which conservatively assume 100 percent bioavailability.

For wildlife (birds and mammals), exposure to metals in soils is through their diets (plants, invertebrates, and wildlife prey, and incidental ingestion of soil). Average daily dietary doses of COPCs are calculated using models that predict the amount of metal taken up from the soil by plants and prey animals and passed through the food chain. Dietary dose-based thresholds (referred to as toxicity reference values, or TRVs) are selected based on laboratory toxicity tests where ≥ 20% increases in adverse effects occur on test organism growth, reproduction, or survival (mortality). The calculated dietary COPC doses are compared to the TRVs to assess whether site COPC exposure may adversely affect wildlife growth, reproduction, or survival.

The comparison of a soil concentration to a benchmark (for plants and soil invertebrates) or a modeled dose to a TRV (for wildlife) results in a ratio called a hazard quotient (HQ).

The HQ based on the benchmark or TRV is first used as a simple binary metric; either the HQ is ≥ 1 or it is < 1. It represents a decision point.

- HQ < 1: The estimated exposure is unlikely to cause an adverse effect leading to unacceptable risk to the EAE. Unacceptable risk can be ruled out at this point in the evaluation, and further analysis is not warranted.
- HQ ≥ 1: Unacceptable risk cannot be ruled out at this point in the evaluation. Additional analysis and evaluation are warranted to quantify the nature and magnitude of potential adverse effects associated with the estimated exposure at each applicable location and to determine whether unacceptable risk is present.

In addition to using plant and invertebrate benchmarks and wildlife TRVs to generate HQs, the severity of effects is estimated where data are available. For plants and invertebrates, sample-specific potentially affected fraction (PAF) values are calculated for COPCs that have BABs. The PAF is an estimate of the number (percent or proportion) of species in a community that could experience ≥20% adverse effects at the measured soil COPC concentration. BABs are equivalent to a PAF of 5 percent. For wildlife, dose-response models are incorporated into the risk characterization. Where dose-response relationships could be developed from the available toxicity data, effective doses (EDxs) are estimated. The EDxs provide an estimate of the magnitude of the adverse effect (as a percent, x, increase in an adverse effect in the COPC exposed population relative to an unexposed control population). Use of PAF and EDx estimates help connect the effects estimated from COPC exposed laboratory populations to community-level impacts (PAFs) and population-level impacts (EDxs), which cannot be evaluated by simply reviewing the magnitude of an HQ when it is greater than 1.

It is not uncommon in ERAs for HQs to be ≥ 1 (equal to or greater than 1) at naturally occurring (background) concentrations of metals in soil. Plants and animals in the environment are adapted to tolerate background concentrations of naturally occurring constituents in the soil where they live and forage. ERAs account for this by assuming that the risks from exposure to background soil concentrations are acceptable. The Upland BERA accounts for background through the use of soil background threshold values (BTVs) (TAI, 2020a). Locations where soil concentrations are less than or equal to BTVs (defined as 95 percent upper confidence limit on the 95th percentile of background sample concentrations) are assumed to not present unacceptable risk, regardless of whether the calculated HQ is ≥ 1.

Typically, when multiple lines of evidence (LOEs) are used to assess risk to an EAE, a weight of evidence (WOE) framework is needed to reconcile any inconsistencies as well as to determine the reliability of all available LOEs for a given receptor-COPC pair (EPA, 2016b). In the Upland BERA two receptor groups, terrestrial plants and soil invertebrates, employ two LOEs to assess risk. However, because the two LOEs used to assess risk are progressive with the second LOE superseding the first, no formal process is needed to weigh the LOEs relative to one another. Therefore, no WOE framework was needed nor used in this Upland BERA. However, one important concept related to WOE guidance, a statement of confidence in the overall risk conclusions, is synthesized from the uncertainty analysis and presented in the risk conclusions for each EAE. Confidence is derived from the reliability, relevance, and strength of the data including the collective properties (number, diversity, sufficiency; absence of bias; and coherence) of the LOEs.

Summary of Findings

Nine of the 19 COPCs evaluated are determined to be COCs that present unacceptable risk for one or more EAE within the receptor group: arsenic, barium, cadmium, copper, lead, manganese, mercury, selenium, and zinc. The outcome for all COPCs is presented in Table ES-1. Ten of the 19 COPCs evaluated are found to pose negligible risk to EAEs so are not identified as COCs: aluminum, antimony, chromium, cobalt, iron, molybdenum, nickel, silver, thallium, and vanadium. Risk characterization could not be completed for antimony, beryllium, and thallium for one or more EAE, due to the absence of sufficient toxicity information for some receptor groups so risks posed by these metals are uncertain. Uncertainty analyses indicate that areas with elevated concentrations of antimony, beryllium, and thallium co-occur with identified COCs suggesting that the extent of risks from these metals is captured by that of the identified COCs. Although risk due to interaction of elevated concentrations of multiple metals cannot be quantitatively assessed with the available data, for the purposes of this risk assessment, it is assumed that locations with multiple COCs with HQs ≥ 1 pose a greater risk to receptors than those locations with fewer exceedances and that risk at a specific location is at least as great as that associated with the COC with the highest HQ.

Chemical of Concern and Ecological Assessment Endpoint Pairs

For plant and invertebrate receptor groups, the HQs presented in Table ES-1 for each COC are based on comparison of soil concentrations with BABs protective of 95% of plant or invertebrate species if available, and with Eco-SSL or SSLs (below which no adverse effects are expected) if BABs were not available. For the bird and mammal receptor groups, HQs calculations are based on estimates of average daily dietary COPC doses compared with dietary dose-based TRVs protective of ≥20% increases in the incidence of survival, growth, or reproduction in sensitive species. Table 10-1 presents bird and mammal HQs for only the most sensitive endpoint (survival, growth, or reproduction) for the most sensitive species. COPCs and respective EAEs (and respective representative species) retained as COCs include:

- Arsenic: plants EAE
- Barium: invertebrates EAE

- Cadmium: invertivore birds EAE (American robin) and invertivore mammal EAE (masked shrew)
- Copper: invertivore mammals EAE (masked shrew)
- Lead: plants EAE, invertivore birds EAE (American robin), herbivore mammals EAE (meadow vole), invertivore mammals EAE (masked shrew), and omnivore mammals EAE (deer mouse)
- Manganese: plants EAE, invertebrates EAE
- Mercury: invertivore birds EAE (American robin)
- Selenium: plants EAE, invertivore mammals EAE (masked shrew), aerial insectivore mammals EAE (little brown bat)
- Zinc: plants EAE, invertebrates EAE, invertivore birds EAE (American robin), carnivore birds EAE (black-capped chickadee).

All COPCs (including the COCs) pose negligible risks to the herbivore birds EAE (California quail), aerial insectivore birds EAE (tree swallow), and carnivore birds EAE (American kestrel).

Uncertainty Analyses

Uncertainty analyses for each line of evidence in the risk assessment consider the uncertainty and variability of the data to determine whether risk analyses may over- or underestimate the potential for adverse effects. The risk characterization and risk conclusions for each receptor group are based on limited site-specific data and rely on several assumptions. As such, it is unknown whether many of the uncertainties identified for each receptor group contribute to over- or underestimates of site exposures. In general, reasonable conservative assumptions are used to ensure that exposure and effects are not underestimated. Quantitative uncertainty analysis is performed to evaluate some uncertainties for each receptor group. Sources of uncertainty that are quantitatively evaluated include:

- Uncertainty associated with using a single incremental composite sample at most locations or the maximum of triplicate incremental composite samples from a subset of locations (2014 UCR Upland Soil Study and the 2015 Bossburg Study) rather than using an estimated 95 percent upper confidence limit. This was evaluated for all receptor groups.
- Uncertainty around median BABs as soil ecological screening benchmarks for plant and invertebrates determined from the results of the predicted no-effect concentration calculator.
- Uncertainty associated with the relative bioavailability assessments for bird and mammal dietary exposure.
- Uncertainty associated with earthworm bioaccumulation models for dietary exposure to birds.
- Uncertainty associated with several dietary exposure and effects assumptions for lead and mercury for birds.

The quantitative uncertainty analyses indicate that none of the sources of uncertainty are substantial enough to change the determination of unacceptable risk from any COC for all EAEs.

Chemicals of Concern Summary

Maps showing HQ ranges for each receptor group (plants, invertebrates, birds, and mammals) for each COC are presented on Maps ES-1 through ES-9. Multiple symbols are shown at each location, with symbol shape indicating receptor group, symbol size and color indicating HQ ranges, closed symbols indicating exceedance of the BTV, and open symbols indicating concentrations below the BTV. Maps ES-10a through ES-10d show the greatest HQ at each sample location among the nine COCs, endpoints, and receptors for each assessment endpoint.

Risks associated with each COC are as follows:

- Arsenic – Arsenic poses unacceptable risk to plants based on exceedances of the Eco-SSL which is based on the plant growth endpoint (Table ES-1). Arsenic concentrations are greater than the BTV and have HQs ≥ 1 in many locations (Map ES-1), particularly in the north-central portion of the Terrestrial Study Area near the U.S.-Canada border. Arsenic poses negligible risk to invertebrates, birds, or mammals.
- Barium – Barium poses unacceptable risk to invertebrates based on exceedances of the Eco-SSL, which is based on the threshold for 20 percent reductions in invertebrate reproduction (Table ES1). Because the barium BTV is close to the Eco-SSL, most locations with HQs ≥ 1 also exceed the BTV. Sample locations with barium HQs ≥ 1 for invertebrates are scattered throughout the central and northern portion of the Terrestrial Study Area (Map ES-2). Although three locations have HQs ≥ 1 for plants, because the HQs are low and the Eco-SSL is conservative, barium poses negligible risk to plants. Barium poses negligible risk to birds and mammals because all samples have concentrations resulting in HQs < 1.0.
- Cadmium – Cadmium poses unacceptable risk to birds and mammals (Table ES-1). There is risk of reduced growth and reproduction to invertivore birds (American robin) with EDxs ≥ 20 throughout the Terrestrial Study Area (Map ES-3). Cadmium poses risk of reduced survival, and to a lesser extent reduced growth, with EDxs ≥ 20 for invertivore mammals (masked shrew) throughout the Terrestrial Study Area. All site soil concentrations exceed the BTV. Cadmium is not a COPC for plants or invertebrates and poses negligible risk to these receptor groups.
- Copper – Copper poses unacceptable risk of reduced survival and growth to invertivore mammals (masked shrew) (Table ES-1). Sample locations with HQs ≥ 1, and EDxs ≥ 50, with copper concentrations exceeding the BTV occur throughout the Terrestrial Study Area but are most frequent near the river in the northern portion of the Terrestrial Study Area (Map ES-4). Copper poses negligible risk to plants and birds because all samples have concentrations resulting in HQs < 1.0 and poses negligible risk to invertebrates because few samples result in HQs ≥ 1.
- Lead – Lead poses unacceptable risk to plants, birds, and mammals. Lead concentrations result in BAB HQs ≥ 1 for reduced plant growth in approximately one-fourth of sample locations in the Terrestrial Study Area. Lead poses unacceptable risk to invertivore birds (American robin) based on concentrations resulting in HQs ≥ 1 or EDxs ≥ 20 throughout a substantial portion of the Terrestrial Study Area and are greatest in the center of the Terrestrial Study Area near the U.S.-Canada border (Map ES-5) where concentrations result in HQs ≥ 5 for reproduction and EDxs ≥ 50 for survival. Lead poses unacceptable risk to multiple mammal feeding guilds including invertivores (masked shrew), omnivores (deer mouse), and herbivores (meadow vole). For mammals, lead exposure poses risk of reduced reproduction throughout most of the Terrestrial Study Area for masked shrew, throughout a substantial portion of the study area for deer mouse, and in a more limited area for meadow vole. Risk of reduced survival in mammals occurs over a more limited extent. Lead is not a COPC for soil invertebrates and poses negligible risk to this receptor group.
- Manganese – Manganese poses unacceptable risk to plants and invertebrates in the Terrestrial Study Area (Table ES-1). Manganese concentrations result in HQs ≥ 1 for the Eco-SSL (based on reduced growth of plants) in nearly all sample locations and HQs ≥ 1 for the Eco-SSL (based on reduced reproduction in invertebrates) in about three-fourths of sample locations (Map ES-6). The BTV is five- and three-fold greater than the plant and invertebrate Eco-SSLs. Roughly one-third of sample locations have concentrations resulting in both HQs ≥ 1 for plants and invertebrates and exceedance of the BTV (Map ES-6). Manganese is not a COPC for birds and mammals and poses negligible risk to these receptor groups.
- Mercury – Mercury poses unacceptable risk to birds (Table ES-1). Mercury has dietary dose EDxs ≥ 20 for invertivore bird (American robin) reproduction at numerous locations in the Terrestrial Study

Area. Locations with EDxs ≥ 20 for reproduction and concentrations greater than BTV occur primarily in northern portion of the Terrestrial Study Area (Map ES-7). An evaluation of alternative more realistic assumptions regarding mercury exposure and effects on bird reproduction indicates that American robin may be adversely affected over a substantial extent in the northern portion of the Terrestrial Study Area. Mercury is not a COPC for plants and invertebrates so poses negligible risk to these receptor groups and poses negligible risk to mammals because all samples have concentrations resulting in HQs < 1.0.

- Selenium – Selenium poses unacceptable risk to plants and mammals (Table ES-1). Sample locations with selenium concentrations resulting in HQs ≥ 1 for plants (based on the Eco-SSL representing plant growth) are scattered throughout the Terrestrial Study Area and are concentrated near the U.S.-Canada Border on the east side of the river (Map ES-8). Selenium HQs ≥ 1 for mammal invertivores (masked shrew) and aerial insectivores (little brown bat) for reduced growth occur primarily on the east side of the river between Northport and the U.S.-Canada Border. In some cases, selenium concentrations reported in the 2012 Ecology Study that result in HQs ≥ 1 are below analytical detection limits. This adds uncertainty to the risk characterization for a portion of the 2012 Ecology Study locations (nondetects are shown with hashed symbols in Map ES-8) but TRV exceedances by detected concentrations are of sufficient magnitude and extent to result in unacceptable risk. Selenium is not a COPC for invertebrates. For birds, HQs ≥ 1 occur over a limited spatial extent and uncertainties (including elevated detection limits in some samples and TRVs based on more bioavailable forms of selenium than likely occur in natural bird diets) likely result in overestimates of the magnitude of risk. Thus, selenium poses negligible risk to invertebrates and birds.
- Zinc – Zinc poses unacceptable risk for all four receptor groups (Table ES-1). Zinc has HQs ≥ 1 in about one-third of locations for multiple bird feeding guilds for principally reduced growth and reproduction but also reduced survival in some locations; about two-thirds of locations for invertivore mammal for reduced growth and reproduction; three quarters of locations for reduced plant growth, and in approximately one-third of locations for reduced growth of invertebrates (Map ES-9). Zinc concentrations were greater than background in all but a few locations.

Conclusions

Nine COCs in the UCR Terrestrial Study Area present unacceptable risk to one or more EAE representative of site plants, invertebrates, birds, and mammals. For each of the receptor groups, at least one COC-EAE pair has HQs ≥ 1 in nearly every sample location (Maps ES-10a through ES-10d). For birds and mammals, HQs ≥ 1 and concentrations > BTV occur in every sample location for at least one COC. Locations near the U.S.-Canada border generally have greater COC concentrations in soils and more COCs with HQs ≥ 1 for multiple receptor groups, suggesting a greater likelihood and severity of adverse effects in this area.

Cadmium, lead, and zinc present the greatest and most widespread risk in the Terrestrial Study Area. These COCs exceed background in nearly all sample locations in all three soil studies evaluated. Conclusions about risk due to cadmium, lead, and zinc have a moderate level of confidence due to the availability of significant toxicity information, use of BABs and estimates of PAFs for plant and invertebrate benchmarks, and dietary dose-response information for multiple bird and mammal effects endpoints (survival, growth, and reproduction).

Copper presents widespread unacceptable risk primarily to invertivore mammals (represented by masked shrew). The risk characterization has a similar level of confidence to that of cadmium, lead, and zinc, but copper concentrations are less elevated relative to background. Locations with HQs ≥ 1 or EDxs ≥ 20 occur primarily in the northern portion of the Terrestrial Study Area.

Mercury presents unacceptable risk to invertivore birds (represented by American robin) in the northern portion of the Terrestrial Study Area. The risk characterization has a similar level of confidence to that of cadmium, copper, lead, and zinc, due to the availability of a robust toxicity data set, although mercury does not have sufficient dose-response information to incorporate dose-response estimates into the risk estimates.

Selenium presents widespread unacceptable risk primarily to invertivore mammals (masked shrew) and to a lesser extent to aerial insectivore mammals (represented by the little brown bat). The risk characterization has a similar level of confidence to that of cadmium, lead, and zinc. Locations with HQs ≥ 1 or EDxs ≥ 20 occur mainly along the river valley and adjacent valley terraces and in the northeastern most portion of the Terrestrial Study Area.

Arsenic, barium, manganese, and selenium also present unacceptable risk to plant and/or invertebrate EAEs in the Terrestrial Study Area, but the risk characterization has a lower level of confidence due to the lack of bioavailability adjustments to benchmarks and dose-response information.

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Acronyms and Abbreviations

°F	degree(s) Fahrenheit
µm	micrometer(s)
95 UCL	95 percent upper confidence limit of the mean
ADA	aerial deposition area
amsl	above mean sea level
AUF	area use factor
BAB	bioavailability-adjusted benchmark
BAF	bioaccumulation factor
BERA	baseline ecological risk assessment
BTV	background threshold value
CEC	cation exchange capacity
COC	chemical of concern
COI	chemical of interest
COPC	chemical of potential concern
CSM	conceptual site model
DQO	data quality objective
DSR	data summary report
DU	decision unit
DUA	data usability assessment
EAE	ecological assessment endpoint
eCEC	effective cation exchange capacity
Ecology	Washington State Department of Ecology
Eco-SSL	ecological soil screening level
ED	effective dose

ED20	effective dose with 20 percent reduction in the response relative to the control
ED50	effective dose with 50 percent reduction in the response relative to the control
ED80	effective dose with 80 percent reduction in the response relative to the control
EDx	effective dose with an x percent reduction in the response relative to the control
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment
FS	feasibility study
HQ	hazard quotient
IC	incremental composite
ICS	incremental composite sample
ITRC	Interstate Technology Regulatory Council
IVBA	in vitro bioaccessibility
kg	kilogram(s)
kg dw/day	kilogram(s) (dry weight) per day
km^2	square kilometer(s)
LCL	lower confidence limit
LOAEC	lowest-observable-adverse-effect concentration
LOAEL	lowest-observed-adverse-effect level
LOAEL ≥ 20	lowest-observed-adverse-effect level with ≥ 20 percent reduction in the response relative to the control
LOE	line of evidence
LOEC	lowest observed effect concentration
MATC	maximum acceptable toxicant concentration

MDL	method detection limit
mg/kg	milligram(s) per kilogram
mg/kg-dw	milligram(s) per kilogram-dry weight
mg/kg bw/day	milligram(s) per kilogram of body weight per day
mm	millimeter(s)
MRL	method reporting limit
MTCA	Model Toxics Control Act
NOAEC	no-observable-adverse-effect concentration
NOAEL	no-observed-adverse-effect level
OC	organic carbon
OU	Operable Unit
PAF	potentially affected fraction
POE	piece of evidence
QA	quality assurance
QAPP	quality assurance project plan
QC	quality control
RAO	remedial action objective
RBA	relative bioavailability
RBC	risk-based concentration
RFDA	relict floodplain deposition area
RI	remedial investigation
RM	river mile(s)
RSD	relative standard deviation
SLERA	Screening-Level Ecological Risk Assessment
SSD	species sensitivity distribution
SSL	soil screening level

TAI	Teck American Incorporated
TAL	target analyte list
TOC	total organic carbon
TRV	toxicity reference value
UCL	upper confidence limit
UCR	Upper Columbia River
USACHPPM	U.S. Army Center for Health Promotion and Preventative Medicine
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
WAC	Washington Administrative Code
WDNR	Washington Department of Natural Resources
WEF	wildlife exposure factor
WOE	weight of evidence
WRIA	Water Resource Inventory Area
WSDA	windblown sediment deposition area
ww	wet-weight

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1. Introduction

This document presents the final Baseline Ecological Risk Assessment (BERA) for the upland terrestrial habitat of the Upper Columbia River Site (UCR site or site)[7] in northeast Washington (Map 1-1). This document is referred to as the "Upland BERA" and has been prepared by the U.S. Environmental Protection Agency (EPA) pursuant to the Settlement Agreement between Teck American Incorporated (TAI) and EPA (EPA, 2006a), the UCR Remedial Investigation (RI)/Feasibility Study (FS) Work Plan (EPA, 2008), and EPA comments on the draft Interim Partial Upland BERA (EPA, 2022a) and draft final Upland BERA (EPA, 2023) documents prepared by TAI (TAI, 2021, 2023a). EPA assumed responsibility for finalization of the Upland BERA in May 2023 (EPA, 2023) as part of a work takeover per the Settlement Agreement.

The contents of this BERA are based in part on data, analyses, and other information presented in the Draft Final Upland BERA (TAI, 2023). However, EPA did not agree with certain analyses, interpretations, and conclusions presented in that document and has conducted significant reanalysis of the data to support the interpretations and conclusions presented in this report. The Final Upland BERA is the document of record regarding risks to upland ecological receptors and will be in the Administrative Record for the UCR site.

The upland area was designated Operable Unit (OU) 3 in the RI/FS Work Plan Schedule Addendum (EPA, 2022b). Though it has yet to be fully defined, the upland area is commonly described as land above the elevations of historical Columbia River flood events and within the approximate footprint of metals deposition associated with historical smelter aerial emissions. For the purposes of the Upland BERA, the upland area is operationally defined as the spatial extent of the upland soil data set used for ecological risk analysis (Map 1-1) and is termed the "Terrestrial Study Area." The actual spatial extent of OU3 is expected to be established by analyses presented in the Draft Final Upland RI Report (TAI, 2023b), which is currently under EPA review.

1.1 Objective of the Upland BERA

Consistent with the objectives described in the RI/FS Work Plan (EPA, 2008) and the Upper Columbia River BERA Work Plan (referred to herein as the "BERA Work Plan") (TAI, 2011) for the UCR site, the objective of the Upland BERA is to assess risk from hazardous substances in soils to ecological assessment endpoints (EAEs) within the Terrestrial Study Area of the site, under both current conditions and expected future conditions assuming no steps are taken to remediate the environment.

The Upland BERA identifies which hazardous substances, exposure pathways, exposure media, and ecological receptors are associated with unacceptable risks to EAEs and provides a basis for informed discussions with risk managers about the causes, nature, and extent of any such risks.

Hazardous substances that might pose risks to EAEs are called chemicals of potential concern (COPCs). The Upland BERA evaluates whether COPCs identified during the previously conducted screening-level risk assessments (SLERAs) present unacceptable risk of adverse effects to representative ecological receptor communities or populations selected as EAEs. COPCs determined to pose unacceptable risk to EAEs will be identified as ecological chemicals of concern (COCs). The distribution of ecological and human health COCs[8] in the upland area will be further evaluated in the Upland RI Report and protection

[7] As defined within the Settlement Agreement of June 2, 2006, the site consists of the areal extent of hazardous substances contamination within the United States in or adjacent to the Upper Columbia River, including the Franklin D. Roosevelt Lake, from the U.S.-Canada border to the Grand Coulee Dam, and all suitable areas in proximity to the contamination necessary for implementation of response actions.

[8] Human health COCs are identified in the final sitewide Human Health Risk Assessment (EPA, 2021)

of both human and ecological receptors will be considered in the development of remedial action objectives (RAOs) and preliminary remediation goals in the Upland FS.

1.2 Media and Spatial Limitations

This Final Upland BERA focuses solely on upland soils and does not address risks in any of the upland aquatic habitats (streams, lakes, and or wetlands) that are present in the Terrestrial Study Area. Risks to ecological receptors in upland aquatic habitats within the Terrestrial Study Area will be assessed in an addendum to this Upland BERA after the upland lakes/wetlands sampling and analysis program is completed.

Additionally, the data evaluated in this BERA do not include riparian soil and beach sediment samples from locations within historically flooded areas adjacent to the Columbia River. Because these areas may have been impacted by river-borne contaminants, ecological risks will be evaluated in the Aquatic BERA for the UCR site. A map depicting pre-1973 maximum flood extent topographic elevations, as defined for the UCR Final Soil Study Quality Assurance Project Plan (TAI, 2014a), was used to determine the areas and data evaluated in the Upland BERA and those that will be evaluated in the Aquatic BERA.

1.3 Ecological Risk Assessment Guidance

The need for an ecological risk assessment (ERA) was described in the Settlement Agreement (EPA, 2006a) and expanded upon in the RI/FS Work Plan (EPA, 2008). The methodology for the Upland BERA was described in the BERA Work Plan (TAI, 2011) and expanded problem formulation (TAI, 2012) and follows EPA guidance for Superfund ERAs (EPA, 1997a). The methodology is consistent with EPA's framework for ERAs (EPA, 1992, 1998; Figure 1-1) and is consistent with other relevant EPA risk assessment and related guidance documents (for example, EPA, 1999, 2007a, 2016a). EPA's framework describes ERA as an iterative three-phase process embedded within a risk management framework. The risk management framework is where risk assessors and risk managers interact. The three phases of an ERA are (1) problem formulation, (2) analysis (exposure and effects assessments), and (3) risk characterization. Subsequent guidance for the Framework for Metals Risk Assessment (EPA, 2007a), provides information about current ERA practices pertinent to this BERA.

The Framework for Metals Risk Assessment (EPA, 2007a) was identified in the Settlement Agreement (EPA, 2006a) as relevant guidance for this Upland BERA. Five key principles describing characteristics of metals and metalloids[9] make them unique from other chemicals (EPA, 2007a). These principles and their application to this Upland BERA are as follows.

- **Principle 1.** Metals are naturally occurring constituents in the environment and vary in concentrations across geographic regions. The natural occurrence of metals in the Terrestrial Study Area is relevant for this Upland BERA and the forthcoming Upland RI for the following reasons identified in the metals framework (EPA, 2007a):
  - Because of the regional variation in naturally occurring metals concentrations in the environment, plants and animals exhibit variable requirements for and/or tolerance to certain metals. These regional differences in requirements and tolerances should be kept in mind when evaluating and extrapolating risks across large areas.

[9] EPA (2007a) identifies the following as metals and metalloids of primary interest: aluminum, antimony, arsenic, barium, beryllium, boron, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury (inorganic), molybdenum, nickel, selenium, silver, strontium, tin, thallium, vanadium, and zinc.

- Because remediation of metals less than background concentrations is neither feasible nor desirable, care should be taken to understand and distinguish among naturally occurring levels, current background levels (that is, natural and anthropogenic sources), and contributions to current levels from aerial deposition originating from historical aerial emissions released from smelter operations.

- **Principle 2.** All environmental media have naturally occurring mixtures of metals, and metals are often introduced into the environment as mixtures.
  - Interactions among and between metals are important aspects of assessing exposure and effects on ecological receptors because some metals can act additively, independently, synergistically, or antagonistically. These different interactions may occur when metals compete for binding locations on specific enzymes or receptors within organisms during absorption, excretion, or sequestration, or at the target site (EPA, 2007a). The accuracy of ecological risk estimates, therefore, can be improved by accounting for the mixtures of metals present in the Terrestrial Study Area if reliable information on metal mixtures toxicity to terrestrial species is available.[10]
- **Principle 3.** Some metals are essential for maintaining proper health of humans, animals, plants, and microorganisms.
  - Essentiality needs to be considered as part of the overall dose-response relationship for those metals where it is relevant. In addition, the necessary doses of essential metals can vary over an organism's life cycle and may differ between male and female organisms of the same species and life stage. Therefore, reference doses designed to protect populations from toxicity from excess metals should not be set below doses identified as essential (EPA, 2007a).
- **Principle 4.** The environmental chemistry of metals strongly influences their fate and effects on human and ecological receptors. Some of the ways environmental chemistry can influence the bioaccessibility, bioavailability, fate, and effects of metals in ways that are relevant to this Upland BERA are identified in the metals framework (EPA, 2007a), as follows:
  - The form of the metal is influenced by environmental properties, such as pH, particle size, moisture, redox potential, organic matter, cation exchange capacity (CEC), and acid volatile sulfides.
  - Certain forms of metals are used for evaluating exposure and effects. For example, the free metal ion is used for exposure assessments based on competitive binding of metals to specific sites of action. However, information developed on the fate and effects of one form of a metal may not be directly applicable to other forms. For example, organometallic forms have different characteristics from inorganic metals and metal compounds, and the same general principles and approaches for risk assessment do not apply.
- **Principle 5.** The toxicokinetics and toxicodynamics (or absorption, distribution, transformation, and excretion) of metals within an organism depend on the metal, the form of the metal or metal compound, and the organism’s ability to regulate and/or store the metal. These highly dynamic processes directly impact the risks of adverse effects from metals on ecological receptors. Some of the implications of these processes for the Upland BERA are identified in the metals framework (EPA, 2007a):
  - Certain metal compounds are known to bioaccumulate in tissues, and this bioaccumulation can be related to their toxicity.

[10] The statement made in EPA (2007a), “accurate prediction of joint toxicity of metal mixtures to terrestrial organisms remains a significant challenge,” is still true today.

- Trophic transfer can be an important route of exposure for metals, although biomagnification of inorganic forms of metals in food webs is generally not a concern in metals assessments.
- Many organisms have developed physiological or anatomical means for regulating and/or storing certain metals up to certain exposure levels such that metals may not be present in organisms in a concentration, form, or place that can result in a toxic effect.
- The organ or tissue in which metal toxicity occurs may differ from the organ or tissue(s) in which the metal bioaccumulates and may be affected by the metal's kinetics. Adverse effects that occur at the point of entry to an organism, however, are less dependent on the internal kinetic processes of the organism.
- Finally, sensitivity to metals varies based on life stage, gender, pregnancy status, nutritional status, and genetics of the organism.

The implications of these five principles on the problem formulation, analysis, and risk conclusions for the Terrestrial Study Area, including available data, assumptions, and uncertainties, are discussed in relevant sections of this Upland BERA.

1.4 Summary of Previous Screening-Level Evaluations

This Upland BERA builds upon screening -level risk evaluations done previously for the site, which were designed to focus the list of COPCs evaluated in the Upland BERA. Specifically, these include the SLERA (TAI, 2010) and the COPC refinement (TAI, 2020b).[11]

The SLERA (TAI, 2010) was conducted following procedures described in EPA's ERA Guidance for Superfund (EPA, 1997a). The primary purpose of the SLERA was to determine whether there were adequate data to make a determination of the potential for unacceptable risks posed to ecological receptors by chemicals of interest (COIs) in environmental media (water, sediment, and soil) at the site. The SLERA also reduced the number of COIs through the use of conservative, risk-based assumptions; if adequate data were available for COI benchmarks and environmental media concentrations, and if risks were determined to be acceptable, then no further assessment was warranted. If adequate data were not available, or if the available information indicated that the COI may pose an unacceptable risk, then further evaluation of those COIs was required.

Following the SLERA, a refined screening -level analysis for the selection of COPCs for the site was conducted (TAI, 2020b). This COPC refinement satisfied the EPA requirement in its dispute resolution letter (EPA, 2013) regarding the Phase 2 Sediment Study Quality Assurance Project Plan, namely that the refinement would include and evaluate any and all EPA-approved data as collected for the RI/FS to date [and] "… refine the assumptions and methods used in the SLERA."

The objectives of the COPC refinement were to identify COPCs to be carried forward to the Upland BERA, COIs to be eliminated from further evaluation based on exposure concentrations less than benchmarks, and COIs to be retained for the Upland BERA because of uncertainty (for example, no benchmarks were available or there were insufficient exposure measurements). COPCs and COIs (that could not be screened in the COPC refinement) identified for the Upland BERA are discussed further in Section 2.4.

These COPCs and COIs identified in the screening level evaluations are further evaluated in the Upland BERA and for simplicity all are referred to as COPCs. Most of the COPCs are metals, but also include

[11] The COPC refinement (TAI, 2019b) included an addendum that was finalized in January 2020 (TAI, 2020b). This citation, (TAI, 2020b), is used in the remainder of the report to cite the COPC refinement document.

two metalloids (arsenic and antimony) and a nonmetal (selenium). For simplicity all COPCs are collectively referred to as “metals.”

1.5 Organization of the Upland BERA Report

The remaining sections of this Upland BERA are organized as follows:

- **Section 2 Problem Formulation**. Presents the focus of the assessment, including the physical characteristics of and land uses in the Terrestrial Study Area, ecological settings, COPCs, receptors, receptor-specific exposure pathways, and receptor-specific EAEs.
- **Section 3** Upland BERA **Data Set.** Provides the data usability assessment and a summary of the overall data set for environmental chemistry used in the BERA.
- **Section 4 Analysis Approach**. Provides descriptions of the risk assessment approaches used for each EAE, including the exposure and effects assessments, and underlying assumptions.
- **Section 5 Risk Characterization Approach.** Outlines the approach used for risk characterization, including comparison to background, uncertainty evaluation, and risk description.
- **Section 6 Risk Characterization for Terrestrial Plants**. Presents the risk characterization for terrestrial plants that grow in the surface and shallow subsurface soil layers in the Terrestrial Study Area.
- **Section 7 Risk Characterization for Soil Invertebrates**. Presents the risk characterization for invertebrate receptors that live and forage in the surface and shallow subsurface soil layers in the Terrestrial Study Area.
- **Section 8 Risk Characterization for Birds**. Presents the risk characterization for birds that live and forage in the Terrestrial Study Area.
- **Section 9 Risk Characterization for Mammals**. Presents the risk characterization for mammals that live and forage in the Terrestrial Study Area.
- **Section 10 Summary and Conclusions**. Provides a summary of the risk evaluation findings for the Terrestrial Study Area.
- **Section 11 References.**

The following appendices are also provided with this Upland BERA:

- **Appendix A Upland BERA Environmental Chemistry Data Usability Assessment**. Presents the data usability evaluation used to assemble the Upland BERA data set.
- **Appendix B Soil Chemistry Data Sets Used in the Upland BERA**. Provides the soil chemistry data sets evaluated in the Upland BERA.
- **Appendix C Terrestrial Bioaccumulation Models**. Presents the approach and data used to generate a subset of the bioaccumulation models used in the Upland BERA.
- **Appendix D Soil Invertebrate and Plant Toxicity Benchmarks**. Presents soil invertebrate and plant benchmarks derived for use in the Upland BERA along with the derivation approach.
- **Appendix E Wildlife Toxicity and Bioavailability Methods Used in the Upland BERA**. Presents bird and mammal toxicity reference values (TRVs) derived for use in the Upland BERA and an evaluation of metal bioavailability to wildlife from metals in soil.
- **Appendix F Hazard Quotients**. Presents the direct contact and dose model hazard quotient (HQ) calculations used to support soil invertebrate, plant, bird, and mammal risk evaluations for the Upland BERA. Also provides detailed information on HQ calculations, including supplemental

tables, figures, and maps that summarize and compare HQ results for every soil chemistry data set evaluated in the Upland BERA.

2. Problem Formulation

A preliminary problem formulation was developed during the SLERA (TAI, 2010), which was refined in the BERA Work Plan (TAI, 2011) and then further refined in the Expansion of the Problem Formulation section of the BERA Work Plan (TAI, 2012). The problem formulation presented in this Upland BERA has been refined further to include information collected since 2012 and is focused on describing how COPCs in site soils might result in adverse effects on terrestrial EAEs.

The Upland BERA problem formulation is used to develop the ecological exposure conceptual site model (CSM) for exposures to soils and to provide the following information:

- Description of the ecological setting
- Identification of COPCs for ecological receptors
- Identification of receptor groups and EAEs
- Description of exposure pathways linking COPCs in surface soils to receptor groups

Figure 2-1 provides a diagram of the ecological exposure CSM that is discussed in the following sections.

This section provides a brief description of the UCR site and its characteristics pertinent to the Upland BERA, including the physical geography and land uses in the region, which will be described in more detail in the Upland RI report. A detailed description of the Terrestrial Study Area ecological characteristics follows. The latter parts of this section summarize COPCs, their potential sources, and the ecological exposure CSM before outlining the EAEs and the risk questions and evidence used to evaluate risk for each EAE.

2.1 Physical Characteristics of the Site

The overall UCR site is located in northeastern Washington State and encompasses approximately 150 river miles (RMs) along the Columbia River, from the U.S.-Canada border to Grand Coulee Dam (Map 1-1). Grand Coulee Dam impounds Lake Roosevelt, a large reservoir extending approximately 133 RMs upstream of the dam at full pool. The Columbia River is free flowing upstream of Lake Roosevelt to the U.S.-Canada border. The upland portion of the UCR site that makes up the Terrestrial Study Area is commonly defined as land above the elevations of historical Columbia River flood events and within the approximate footprint of metals deposition associated with historical smelter aerial emissions (operationally defined as the spatial extent encompassed by existing upland soil data). The actual spatial extent of OU3 is expected to be established by analyses presented in the Draft Final Upland RI Report (TAI, 2023b), which is currently under EPA review.

The Terrestrial Study Area is sparsely populated and includes federal, state, and private forest lands and farmland. The northern end of the site is largely undeveloped, and there is partial overlap with the Colville National Forest to the west and northeast (EPA, 2008). The region is known to have bedrock formations that, in some areas, are enriched with metal ores, and there are placer deposits in the Columbia River and its tributaries. Mining, milling, smelting, and prospecting have occurred in this region since the mid-1800s (EPA, 2008).

The Terrestrial Study Area is operationally defined as the areas represented by soil sample locations used in the Upland BERA data set, which includes aerial deposition area (ADA) locations sampled for the 2014 UCR Final Soil Study Data Summary and Data Gap Report (referred to herein as the "2014 UCR Upland Soil Study") (TAI, 2015), the Upper Columbia River Upland Soil Sampling Study (referred to herein as the "2012 Ecology Upland Soil Study") (Ecology, 2013), and the 2015 Bossburg Flat Beach Refined Sediment and Soil Study (hereinafter referred to as the "2015 Bossburg Study") (TAI, 2016)

(Section 3). As shown by the black outline and red outlines on Map 2-1, the 2014 UCR Upland Soil Study and the 2012 Ecology Upland Soil Study (TAI, 2015; Ecology, 2013) were conducted in the northernmost 100 square miles extending from the U.S.-Canada border south to approximately the latitude of China Bend, between the ridge located east of the river and a western border that is less distinct because it is bisected by relatively large drainages, but in general follows a line north to the U.S.-Canada border. The area represented by the 2015 Bossburg Study (TAI, 2016) consists of small plots of land on both sides of the river, located just upriver of the Bossburg Flat Beach Area (Map 2-1).

The Terrestrial Study Area is located in the Okanogan Highlands province of northeastern Washington. Elevations in the Terrestrial Study Area range from 1,290 feet above mean sea level (amsl) (the full-pool elevation of Lake Roosevelt) to more than 4,700 feet amsl. Topography consists of glaciofluvial terraces adjacent to the Columbia River, tributaries to the Columbia River, and surrounding hills, mountains, and ridges to the east and west that are the characterized by variable terrain and elevations. The terraces and stream-cut scarps, with varying slopes from gentle to steep, have also been modified by gullying, landsliding, and creep (Jones et al., 1961).

The hydrological setting for the overall UCR site includes the Columbia River and the areas it drains between the U.S.-Canada border and Grand Coulee Dam, including its major tributaries. River flows are controlled by the operation of upstream dams in addition to operations at Grand Coulee Dam. A regulated flow regime has been in effect on the Columbia River since construction of Grand Coulee Dam was completed in 1942 and other upstream dams in Canada were constructed and began operating in the 1970s. Upstream dams in Canada and the U.S. that influence discharge into the Columbia River are shown on Map 2-2. In general, reservoir elevation is reduced from January to April to prevent flooding during spring runoff; allowed to increase during May and June; allowed to decrease slightly through July and August, when recharge is diminished; and allowed to increase from September to December as a response to increased autumn precipitation.

The Terrestrial Study Area encompasses land within Washington's Water Resource Inventory Areas (WRIAs)[12] 61 and the northwestern part of WRIA 62 (Map 2-2). Major WRIA 61 drainages within the Terrestrial Study Area include Cedar Creek, Deep Creek, Quartz Creek, Sheep Creek, Onion Creek, Rattlesnake Creek, Crown Creek, and Flat Creek, all of which discharge into the Columbia River. Terrestrial Study Area drainages in WRIA 62 discharge to the Pend Oreille River.

Surficial geology of the Terrestrial Study Area is shown on Map 2-3. It is primarily composed of the following:

- Glacial fluvial and lacustrine sediments and till
- Recent unconsolidated fluvial deposits, alluvium, and colluvium
- Paleozoic metasedimentary, metacarbonate, and metavolcanic rocks (for example, Carboniferous through Devonian marine metasedimentary rocks of the Grass Mountain, Pend Oreille, and Flagstaff Mountain sequences as well as the Metaline Formation, Maitlen phyllite, and Ledbetter slate)
- Eocene igneous intrusive rocks (for example, Sheppard granite) and extrusive rocks (for example, O'Brien Creek Formation and Sanpoil volcanics).

[12] WRIAs are areas established in the state of Washington to be used as the basis for natural resource management purposes in the state. WRIA boundaries are based on natural watershed and drainage characteristics. The geographic definitions for WRIAs across the state were formalized under Washington Administrative Code (WAC) 173-500-040.

Localized mineralization in the Terrestrial Study Area is primarily associated with Paleozoic metasedimentary, metacarbonate, and metavolcanic rocks of the Grass Mountain, Pend Oreille, and Flagstaff Mountain sequences as well as the Metaline Formation, Ledbetter slate, and Maitlen phyllite.

Meteorologically, the site exhibits a predominantly continental climate, characterized by warm, dry summers and cold, moist winters. Precipitation and temperature vary substantially across the region.). During the summer months, temperatures range from 75 to 100 degrees Fahrenheit (°F) in the daytime, dropping to 50 to 60°F at night (TAI, 2011). Fall and spring seasons are characterized by sunshine and mild temperatures, which vary between 50 and 80°F during the daytime and between 30 and 50°F in the night (TAI, 2011). Winters are cold, with daytime temperatures between 25 and 40°F, and nighttime temperature ranges as low as 15 to 20°F. Wildfires may occur seasonally, and fire intensity may vary depending on drought conditions and temperature. Strong, gusty winds occur mainly in spring and summer (Ferguson, 1999). Meteorological monitoring data collected along the UCR indicate the dominant wind direction is from the southwest for much of the year, with northeast winds prevailing in the fall and early winter (DOI, 2006). However, wind direction distributions have strong seasonal variation and are affected by local topographic conditions. The many steep-walled valleys and canyons along the Columbia River Valley can channel and accelerate winds to very high speeds (Ferguson, 1999). Locally, these winds may occur in directions that are different from prevailing winds. Diurnal variations in slope and valley winds are also likely to occur in the Columbia River Valley. An evaluation of air movement in the vicinity of Trail, British Columbia, found that up-slope and up-valley winds occurred during the day, while down-slope and down-valley winds occurred at night. These patterns were more pronounced in the summer than the winter (Cross, 1950). A more thorough evaluation of diurnal air movement will be provided in the Upland RI report.

2.2 Land Uses

As shown on Map 2-5, the Terrestrial Study Area is predominantly rural and forest lands. The area is largely undeveloped and there is partial overlap with the Colville National Forest to the west and the northeast. Land uses include management of federal, state, and private forest lands; residential, agricultural production; outdoor recreation; and cultural uses, with some commercial and retail services in established towns and crossroads.

Various agricultural activities, including orchards, have been documented around Northport extending northward to the U.S.-Canada border since the early 1900s (Dominion of Canada, 1936). Based on aerial imagery covering the period from 2015 to 2023 (Esri, 2023), land uses on some of these lands appear to remain agricultural, mostly being either single-family farms or large plots of lightly developed or undeveloped private land. Timber harvesting is prevalent in the area, occurring mostly on private forest land with a small percentage of trees harvested from state and federal lands.

Ore mining and mineral processing has been occurring in the UCR region since at least the late 1800s. Most of the operations in the U.S. took place in Stevens and Ferry counties (Orlob and Saxton, 1950; Wolff et al., 2005). Mining activities in the drainage basin also occurred in the Metaline mining district. The locations of the mines, mills, and smelters in the UCR drainage basin, including those north of the border, and along tributaries to the UCR are shown on Map 2-4. Most of these operations became inactive or were closed in the early to mid-20th century. Only the Teck Smelter in Trail, British Columbia is known to be currently active.

2.3 Habitat and Species Present

The Terrestrial Study Area is within the Northern Rockies ecoregion (Omernik, 1987). The native vegetation is primarily coniferous forest ecosystems interspersed with montane grasslands (McNab et al., 2007). Ecological habitat comprises most of the Terrestrial Study Area and is interspersed with residential neighborhoods and rural residential lots, utility and railroad rights-of-way, agriculture, retail and other services, manufacturing, and land designated for cultural or recreational use (Map 2-5).

Sixteen ecosystem types[13] have been described for the Terrestrial Study Area (Map 2-6) (NatureServe, 2009). As shown on Map 2-6, the shorelines and lower elevation terraces adjacent to the Columbia River are a mix of grasslands, shrub lands, savanna, and forest. The higher slopes are predominantly conifer forest with some areas of savanna. Portions of the Terrestrial Study Area are logged, sometimes by clear cutting, and the region has a history of wildfires.

Regional information on the presence and distribution of terrestrial wildlife has been compiled by several resource agencies (Hebner et al., 2000; Creveling and Renfrow, 1986; CTFWD, 2006; McCaffrey et al., 2003; Rodhouse, 2005). Hebner et al. (2000) listed the wildlife species in the area extending from Grand Coulee Dam to the U.S.-Canada border and the reservoir to the surrounding ridges, while Quigley et al. (2001) and Marcot et al. (2003) have reported species data for the interior Columbia Basin region. Species that only occupy the sagebrush grasslands and shrub-steppe habitats characteristic of the southern section of the Columbia Basin region are not relevant to the Terrestrial Study Area. Expanded discussions of terrestrial plants, soil invertebrates, birds, mammals, herpetofauna, and threatened or endangered species are presented in the sections that follow.

2.3.1 Upland Aquatic Habitat

Approximately 25 lakes or open water areas and 120 wetlands have been mapped in the Terrestrial Study Area (USFWS, n.d.). These lakes and wetlands provide habitat for a number of aquatic species including plants, invertebrates, and fish, and are integral to maintaining the connectivity of food webs between aquatic and terrestrial environments. Aquatic-dependent upland species include riparian and wetland plant communities, animals that forage on aquatic fish and invertebrates (for example, belted kingfisher, American mink) and aerial insectivores (for example, violet-green swallow, little brown bat) that consume swarms of emergent aquatic insects. The upland aquatic habitats are especially important for the resident amphibians (for example, Columbia spotted frog) and migrating and resident waterfowl that depend on these lakes and wetlands for suitable breeding habitat (Green et al., 1997).

2.3.2 Terrestrial Plants

Native terrestrial vegetation in the Terrestrial Study Area contains coniferous forest ecosystems interspersed with montane grasslands. Riparian and wetland ecosystems occur along watercourses and waterbodies or where soils are saturated for part or all of the growing season. The distribution of plant communities in the Terrestrial Study Area reflects unique combinations of edaphic conditions, temperature, precipitation, evaporation, and solar radiation (that is, slope and aspect). Riparian and wetland plant communities are also influenced by proximity to water and extent, frequency, and duration of flooding (Manning and Engelking, 1997).

As noted earlier in this section, there are 16 ecosystem types identified for the Terrestrial Study Area (Map 2-6). These ecosystems, or groups, are predominantly defined by sets of diagnostic plant species, as well as by regional mesoclimate, geology, substrates, hydrology, and disturbance regimes (USNVC, 2022). These ecosystems may be further subdivided into finer-scale alliances and associations using diagnostic species and more localized conditions (USNVC, 2022). Of the 16 ecosystems identified,

[13] The 16 ecosystem types included in the count are those that make up at least 0.1 square kilometer (km^2) (0.04 square mile) of the Terrestrial Study Area. Excluded from this count are ecosystems composing less than 0.1 km^2, as well those that are nonvegetated (for example, developed area, cliff, bedrock, open water) grouped as “Other” on Map 2-6.

the most common ecosystem types (those that cover 5 percent or more of the Terrestrial Study Area) are as follows:

- **Northern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest.** This ecosystem is characterized by a mix of Douglas fir (*Pseudotsuga menziesii*) and ponderosa pine (*Pinus ponderosa*), with ponderosa pine typically occurring at lower elevations or in drier areas. The forest understory is dominated by grasses and sedges, including bluebunch wheatgrass (*Pseudoroegneria spicata*), pine grass (*Calamagrostis rubescens*), and Ross's sedge (*Carex rossii*), with a variety of shrubs, including juniper (*Juniperus communis*) and mallow-leaf ninebark (*Physocarpus malvaceus*).
- **Northern Rocky Mountain Ponderosa Pine Woodland and Savanna.** This ecosystem occurs at the lower tree line at the ecotone between grasslands and coniferous forests, typically on warmer, drier sites. The tree canopy is dominated by ponderosa pine, although Douglas fir may also occur. The understory can be shrubby and may include big sagebrush (*Artemisia tridentata*), bearberry (*Arctostaphylos* spp.), antelope brush (*Purshia tridentata*), rose (*Rosa* spp.), and Saskatoon berry (*Amelanchier alnifolia*) as common species. In more open savanna areas, shrubs are uncommon, and the understory comprises fire-resistant cool season bunchgrasses and forbs, including bluebunch wheatgrass, fescue (*Festuca* spp.), and needle-and-thread grass (*Hesperostipa comata*).
- **Northern Rocky Mountain Mesic Montane Mixed Conifer Forest.** This forested ecosystem is typically found on moist areas, such as on toe slopes. The tree species canopy in this ecosystem is dominated by western red cedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*) with Douglas fir. Common shrub species include twinflower (*Linnea borealis*), Oregon boxwood (*Paxistima myrsinites*), gray alder (*Alnus incana*), thimbleberry (*Rubus parviflorus*), and huckleberry (*Vaccinium membranaceum*).
- **Northern Rocky Mountain Lower Montane Foothill and Valley Grassland.** The grassland ecosystem is characterized by the same cool-season perennial bunchgrasses that occur in the low-elevation pine woodlands and savannah, including bluebunch wheatgrass, fescue, and needle-and-thread grass, with a sparse shrub layer that may include rose, Saskatoon berry, juniper, and snowberry or buckbrush (*Symphoricarpos* spp.) (NatureServe, 2009). Limited areas of sagebrush shrubland and sagebrush steppe also occur in the Terrestrial Study Area.
- **Harvested forest-tree regeneration.** This ecosystem is anthropogenically impacted and consists of regenerated areas of forest after lumber harvesting. Depending on land ownership and forest management practices, tree species in formerly harvested lands may be less diverse if active replanting of specific species has occurred.
- **Northern Rocky Mountain Montane Foothill Deciduous Shrubland.** This shrubland ecosystem is found in the lower montane and foothill regions around the Columbia Basin, typically below the treeline and on steep slopes of canyons. The most common dominant shrubs include mallow-leaf ninebark, cherry (*Prunus* spp.), rose, sumac (*Rhus glabra*), maple (*Acer glabrum*), Saskatoon serviceberry, snowberry or buckbrush, and ocean spray (*Holodiscus discolor).*
- **Northern Rocky Mountain Lower Montane Riparian Woodland and Shrubland.** Consisting of deciduous, coniferous, and mixed conifer-deciduous forest, this ecosystem occurs on streambanks and river floodplains of the lower montane and foothill zones of the northern Rocky Mountains with hydric soils and experiences annual flooding throughout the growing season. This ecosystem is identified by the key indicator species, balsam poplar (*Populus balsamifera*), but other trees such as birch (*Betula* spp.) and spruce (*Picea* spp.) can be mixed in the canopy. Shrubs, ferns, and forbs such as red osier dogwood (*Cornus sericea*), alder, lady fern (*Athyrium filix-femina*), and arrowleaf ragwort (*Senecio triangularis*) are commonly present.

Common vascular plant species that are representative of the ecosystems in the Terrestrial Study Area are summarized in Table 2-1 (NatureServe, 2009). Plants provide many functions in the ecosystem. As primary producers, terrestrial plants form the foundation of the food web in the upland habitat. For example, cone-producing trees, including pines and spruce, and seed- and berry-producing shrubs and forbs, such as blueberries (for example, *Vaccinium* spp.) and cherries, are a key food source for herbivorous and omnivorous birds, including upland game birds and a variety of songbirds (for example, sparrows, thrushes, chickadees, and nuthatches). Woody shrubs like willows (*Salix* spp.) and dogwoods (*Cornus* spp.) provide browse for ungulates, including moose (*Alces alces*), deer, and elk (*Cervus canadensis*). Moose also rely on aquatic wetland plants (for example, sedges [*Carex* spp.]) during the summer, and elk and deer graze on sedges and grasses in the forest understory and in grasslands. Smaller mammals, including white tailed jackrabbit (*Lepus townsendii*), porcupine (*Erethizon dorsatum*), mice, and voles rely on the foliage, bark, fruits, and seeds of a wide variety of herbaceous plants, grasses, shrubs, and trees. Terrestrial plants sustain a diverse invertebrate community, including foliage-feeding, wood boring, and nectar or pollen feeding insects and root-feeding invertebrates in the soil (Johnson and Rasmann, 2015). Dead plant material, including coarse woody debris, leaf litter, and roots, supports a variety of detritivores and is the primary input of organic matter back into the soil (Gross and Harrison, 2019).

2.3.3 Soil Invertebrates

Detailed information on the diversity of the soil invertebrate community in the Terrestrial Study Area is not available; however, species diversity for some orders has been reviewed for the Columbia Basin as a whole (Table 2-2). These reviews catalogued three species of earthworms (James, 1995), 32 species of true bugs (*Hemiptera*) (Lattin, 1995), and 1,300 predaceous beetle species (*Coleoptera*) (McIver et al., 1994). Given the Columbia Basin's large geographic extent, the species counts previously noted may overestimate the quantity of species but may underestimate the diversity of soil invertebrates present in the Terrestrial Study Area.

Some invertebrates are semiaquatic, using both terrestrial and aquatic habitats. These include mollusks (*Stylommatophora*), worms (*Lumbriculata* and *Platyhelminthes*), and wood lice (*Isopoda*).

Within the soil invertebrate community, there is a wide variety of functional feeding groups that play different ecological roles in the terrestrial ecosystem. These include the following guilds:

- **Detritivores.** Detritivores feed on dead and decaying plant and animal material, sometimes also consuming bacteria, lichens, and fungi. Coarse woody debris chewers consume woody material, feeding primarily on dead stems, branches, or twigs. Detritivores often inhabit the soil and are important contributors to decomposition and nutrient cycling. Detritivores include species from the worm invertebrate classes *Lumbriculata* and *Platyhelminthes* and species from insect groups, including beetles (*Coleoptera*), termites (*Blattodea*), and ants (*Hymenoptera*).
- **Root-feeding herbivores.** This functional group inhabits the soil and consumes plant roots. Insect orders with root-feeding herbivorous species include beetles and true flies (*Diptera*). Root-feeding herbivores interact with many other functional groups to shape populations and communities of plants, soil microbes, herbivores, and aboveground food webs (Johnson and Rasmann, 2015). Many root-feeding species are considered plant pests.
- **Foliage-feeding herbivores.** This guild feeds on the foliage of grasses, forbs, shrubs, or trees. Feeding is confined to high-quality portions of the plant (that is, growing tips, developing seeds, or other reproductive parts) (Lattin, 1995). Herbivory contributes to nutrient cycling by adding nitrogen and other elements to the soil through nitrogen rich fecal pellets and parts of foliage that are clipped off during feeding (Miller, 1994). Foliage-feeding herbivores that are also soil invertebrates include those that specialize on understory plants (that is, low-growing herbs and grasses). Insect orders that include soil-dwelling, foliage-feeding herbivores include true bugs (*Hemiptera* and *Homoptera*), grasshoppers and crickets (*Orthoptera*), and beetles.

- **Predators.** Predators are carnivorous invertebrates that can be classified as either chewing or suctorial. Chewing predators consume whole adult or larval invertebrates or their eggs. Suctorial predators capture and inject prey with salivary enzymes then ingest the liquefied contents (Johnson, 1995). They can be found in the soil, on the ground, and throughout all vegetation layers (that is, in the understory and in the tree canopy). In forest ecosystems, predation is thought to be responsible in large part for regulating invertebrate populations, including pest populations under endemic conditions (McIver et al., 1994).
- **Parasitoids.** One of the orders in Table 2-2 includes parasitoids (*Coleoptera*). These species are restricted to host-parasite relationships for food (Johnson, 1995). The larvae develop and feed within another invertebrate species (the host) that will eventually die. The parasitoid adult is free living and may be predaceous. Much like predators, parasitoids also play a role in regulating invertebrate populations in a variety of ecosystems.

Most of these invertebrate functional feeding groups are detritivores and primary consumers that consume the terrestrial plant material that forms the foundation of the food web. Soil invertebrates sustain a diverse community of secondary consumers, or mesopredators, including invertivorous arthropods, birds, mammals, amphibians, and reptiles. Ground-dwelling invertebrates (that is, detritivores and root-feeding herbivores) are a key food source for invertivorous birds (for example, soil probing thrushes and warblers), amphibians, lizards, and small mammals (for example, shrews). Many predatory invertebrates, including beetle larvae, centipedes, and some true flies, also consume soil-dwelling invertebrates.

2.3.4 Birds

There are 250 species of birds identified in the UCR region, of which 137 terrestrial-dependent species (Table 2-3) are known to occur in the Terrestrial Study Area (TAI, 2012). The diets of terrestrial-dependent birds are derived from multiple trophic levels, including terrestrial vegetation (for example, fruits, seeds), invertebrates (for example, worms, insects, and other arthropods), amphibians, reptiles, and small to medium-sized birds and mammals. The avian trophic groups, or feeding guilds, include herbivores, invertivores, aerial insectivores, omnivores, and carnivores.

The diet of herbivorous birds (for example, California quail) primarily comprises vegetative material, such as fruits and seeds, though they may occasionally consume invertebrates. Invertivorous birds consume predominantly invertebrates, such as worms, insects, and other arthropods. This group of birds was divided into two guilds based on their feeding habits: soil invertivores (for example, American robin) and aerial insectivores (for example, tree swallow). Soil invertivores feed on the ground, vegetation, or debris and may also consume vegetative material or incidentally ingest soil while probing for invertebrates. Aerial insectivores feed, primarily while flying, on swarms of terrestrial and emerged aquatic insects[14] and have minimal to no soil ingestion. The omnivore guild (for example, black-capped chickadee) forms a great portion of terrestrial birds in the Terrestrial Study Area, which consumes both vegetative material and invertebrates. Carnivorous birds (for example, American kestrel) are top predators and feed on vertebrate prey, including mammals, birds, reptiles, and amphibians.

Some bird species will use both aquatic and terrestrial habitats for feeding; however, this Upland BERA focuses on birds exposed to COPCs in terrestrial habitats. Exposures of birds to COPCs in aquatic habitats will be evaluated in the Aquatic BERA.

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[14] For the Upland BERA, it is assumed that aerial insectivores are exposed to COPCs in flying insects with entirely terrestrial life cycles. Exposure to COPCs in aquatic emergent insects will be evaluated in an addendum to this Upland BERA that will be prepared after the Upland Lakes Sediment Study is completed and in the Aquatic BERA.

2.3.5 Mammals

The Upland BERA focuses on mammalian species that use terrestrial resources and forage and spend most of their time in terrestrial habitats in and around the site. In total, there are 98 mammalian species known to use the site, and 65 of these are considered terrestrial-dependent mammals that may be found within the Terrestrial Study Area (TAI, 2012). A list of these terrestrial mammals and their feeding guilds is presented in Table 2-4.

In the Upland BERA, terrestrial mammals are divided into five feeding guilds: herbivores, invertivores, aerial insectivores, omnivores, and carnivores. Herbivorous mammals (for example, meadow vole) consume fruits, seeds, leaves, twigs, roots, and other plant parts. Some of these species consume vegetation close to the ground and burrow or dig; because of this, these species may incidentally ingest soil while foraging. Both soil invertivorous and aerial insectivorous mammals consume primarily invertebrates, such as insects and arthropods. Soil invertivores (for example, masked shrew) may also consume some vegetation or soil while foraging and eating invertebrates. Like birds, aerial insectivores (for example, little brown bat) consume mainly swarms of terrestrial and emergent aquatic insects and have little to no soil ingestion. Omnivorous mammals (for example, deer mouse) consume both vegetation, such as seeds and fruits, and invertebrates, such as arthropods and insects. Carnivorous mammals (for example, short-tailed weasel) feed on other vertebrates, such as mammals, birds, reptiles, and amphibians. This feeding guild includes the top predators (for example, gray wolf) in the Terrestrial Study Area.

2.3.6 Herpetofauna

Herpetofauna of the Terrestrial Study Area are divided between the amphibian and reptile taxonomic classes. There are 10 amphibian and 15 reptile species reported to occur in the site (TAI, 2012); of these species, four amphibian and seven reptile species may occur within the Terrestrial Study Area (Table 2-5). The general ecology of both taxonomic classes is described in Sections 2.3.6.1 and 2.3.6.2.

2.3.6.1 Amphibians

There are four species of amphibian that may occur in the Terrestrial Study Area (Table 2-5): Columbia spotted frog (*Rana luteiventris*), long-toed salamander (*Ambystoma macrodactylum*), Pacific tree frog (*Pseudacris regilla*), and western toad (*Anaxyrus boreas*). Amphibians occupy both terrestrial and aquatic habitats, depending on their life stage. They lay their eggs in water and attach the egg mass to aquatic vegetation or in woody/rocky substrate. Larval stages (tadpoles) have gills and are confined to aquatic environments; their exposure to COPCs will be evaluated in the Aquatic BERA and in a separate Addendum to the Upland BERA that will be prepared after the Upland Lakes Sediment Study is completed. Breeding habitats include shallow waterbodies that are temporary or permanent, such as wetlands, lakes, slow-moving streams and rivers, backwaters, and marshes (WDFW, 2023b). Tadpoles undergo metamorphosis where their bodies transform to obtain limbs and lungs; they can then emerge to terrestrial habitats. This Upland BERA focuses on terrestrial exposures to adult amphibians. All amphibians found in the Terrestrial Study Area are terrestrial as adults, spending much of their life away from water. However, as amphibians they are dependent on water to complete their life cycles, and the timing of emergence from winter hibernacula to spring breeding sites is largely dependent on temperature and, therefore, elevation (Smith-Gill and Berven, 1979). Temperature, elevation, and hydroperiod are also responsible for the duration of embryo development and metamorphosis (Blaustein et al., 2001; Smith-Gill and Berven, 1979).

Terrestrial habitats used by amphibians include meadows, woodlands, forests, and sometimes disturbed or urban areas. Adults and juveniles forage in densely vegetated riparian areas and under logs, rocks, and other debris. They sometimes dig burrows in the soil for cover. Adult and juvenile amphibians are freeze-intolerant species that overwinter underground. Overwintering habitats can be communal but are often individual dens dug in streambanks, under downed wood, or in used burrows of other animals (for example, rodents and moles) (Mullally, 1952).

Amphibians are invertivores that mainly consume terrestrial and aquatic arthropods (predominantly spiders and beetles), snails, and earthworms. The long-toed salamander also consumes aquatic larvae. Based on their foraging techniques, amphibians may incidentally ingest soil or water while feeding. Predators of amphibians include other herpetofauna as well as omnivorous or carnivorous birds and mammals.

Amphibians are sensitive to environmental threats due to their heavy reliance on cutaneous (skin surface) respiration. They have moist, thin skin that is permeable; this allows liquids and gases to pass easily through the skin.

2.3.6.2 Reptiles

Terrestrial reptiles comprise snakes and lizards (which include skinks). There are eight reptile species potentially occurring in the Terrestrial Study Area: common garter snake (*Thamnophis sirtalis*), gopher snake (*Pituophis catenifer*), North American racer (*Coluber constrictor*), rubber boa (*Charina bottae*), terrestrial garter snake (*Thamnophis elegans*), northern alligator lizard (*Elgaria coerulea*), western rattlesnake (*Crotalus oreganus*) and western skink (*Eumeces skiltonianus*) (Table 2-5). Because toxicity data for reptiles are being compiled with those for amphibians, risk to reptiles will be assessed in an addendum to this Upland BERA that will be prepared after the Upland Lakes Sediment Study is completed.

Snakes are predominantly found in forests and woodlands in the foothills and open grassy areas at lower elevations (MFWP, 2020a). They prefer moist habitats, such as along borders of streams, ponds, and lakes. They are found under logs, rocks, and leaf litter or basking in the sun in open areas (for example, on roads or flat rocks) (Baxter and Stone, 1980). Snakes hibernate during winter in hibernacula, such as rock piles, debris-filled wells, caves, crevices, building foundations, unused burrows made by other animals, and ant mounds.

Snakes have small home ranges. For example, the North American racer has a home range of 330 feet. Snakes are carnivorous and feed primarily on invertebrates and small vertebrates, such as mice, rabbits, birds, shrews, amphibians, lizards, and occasionally other smaller snakes. The terrestrial garter snake and common garter snake are generalists, consuming both terrestrial and aquatic prey. Prey items most commonly include earthworms and adult frogs and toads but have also included amphibians in metamorphosis, leeches, slugs, and fish (Anderson, 1977; Arnold, 1977; Halliday, 2016). Predators of snake species include carnivorous mammals, birds, and sometimes fish and larger snakes.

Lizards (which include skinks) occur in cool, humid habitats within coniferous forests, woodlands, and clear cuts. They prefer locations with numerous hiding places provided by rocks, decaying logs, leaf litter, and vegetation; they can be particularly abundant along riverbanks (Government of Canada, 2005). Lizards also are associated with rocky outcrops and talus. They hibernate in underground burrows below the frost line that they construct themselves or that small mammals have constructed. The western skink constructs hibernation burrows up to 20 inches long (Nussbaum et al., 1983). Lizards do not migrate. They are invertivores that mainly consume insects, ticks, spiders, millipedes, earthworms, slugs, and snails (Stebbins, 2003; St. John, 2002). While foraging, they may also incidentally ingest soil. Mammals, birds, and sometimes snakes will prey on lizards.

2.3.7 Threatened and Endangered Species of the Terrestrial Study Area

There is the potential for one[15] state-listed endangered plant species to occur in the Terrestrial Study Area: Gastony's cliffbrake (*Pellaea gastonyi)* (Table 2-6). Only one occurrence of this rare species has been documented by Washington Department of Natural Resources (WDNR) (Map 2-6). Gastony's cliffbrake has a narrow habitat preference of calcareous rock, often limestone or dolomite (Brunton, 1979; Windham, 1993; Friesen and Murray, 2015). This species lives in montane and subalpine areas, on often dry calcareous cliffs, in crevices, and on edges of limestone outcrops in cracks in the rock (Friesen and Murray, 2015; NatureServe, 2009; WDNR, 2023a). Because Gastony's cliffbrake grows in areas that do not contain a significant amount of soil (that is, cliffs, crevices, ledges), exposure to soil is expected to be minimal; thus, this species is not evaluated further in the Upland BERA.

There is the potential for two federally listed threatened mammal species to occur in northeastern Washington State: Canada lynx (*Lynx canadensis*), and grizzly bear (*Ursus arctos*), respectively (Table 2-4) (USFWS, n.d.c). Gray wolf (*Canis lupus*) was federally listed as endangered until February 10, 2022, when, by court order they were delisted in the Northern Rocky Mountain region including throughout the UCR site (USFWS, 2023). Gray wolf listing status is under review by U.S. Fish and Wildlife Service and a proposed ruling is expected by February 2024. There are four state-listed endangered species, gray wolf, fisher (*Pekania pennanti)*, grizzly bear, and Canada lynx, that also potentially occur in northeastern Washington State (Table 2-4) (USFWS, n.d.a; WDFW, 2023b). However, the only state-listed mammal species regularly observed in the Terrestrial Study Area is the gray wolf (Table 2-6).

One federally listed threatened and state-listed endangered bird species, the yellow-billed cuckoo (*Coccyzus americanus*), is potentially present in northeastern Washington (USFWS, n.d.c; WDFW, 2023b). However, the yellow-billed cuckoo does not occur in the Terrestrial Study Area (Cornell Lab of Ornithology, 2023; Wiles and Kalasz, 2017). Thus, there are no federally or state-listed bird species in the Terrestrial Study Area.

There are no amphibian or reptile species that are currently federally or state-listed as threatened or endangered in the Terrestrial Study Area.

2.4 COPCs and COIs

Soil COPCs and COIs were identified during the SLERAs (Section 1.4) and are summarized in Table 2-7. The screening-level risk assessment identified a number of organic COIs for the site (Section 1.4). However, COPCs and COIs evaluated in the Upland BERA are limited to target analyte list (TAL) metals selected at the direction of EPA (TAI, 2014a). Additionally, organic chemical analytes were not analyzed for the 2014 UCR Upland Soil Study chemistry data set, given the unlikelihood that these analytes originated from deposition of historical smelter aerial emissions (TAI, 2014a).

Most of the TAL metals identified as COIs in the COPC refinement (TAI, 2020c) were carried forward into the Upland BERA as COIs because screening benchmarks were not available in the COPC refinement. Many of these have ecological toxicity data available for them (Section 4.2). Thus, all COIs from the COPC refinement for which the available toxicity data allow for derivation of benchmarks and/or TRVs are quantitatively evaluated in the Upland BERA; these quantitatively evaluated COIs are

[15] Michigan moonwort is found in northern Washington but does not occur within the Terrestrial Study Area (WNHP, 2021). Columbia crazyweed occurs solely along gravel bars or stony shores along the Columbia River, and most populations have been extirpated as a result of habitat destruction by the construction of the Grand Coulee Dam (WDNR, 2022b). Since Columbia crazyweed exposure occurs in the relict floodplains of the river, this species will be evaluated in the Aquatic BERA.

referred to as COPCs hereafter. COIs for which toxicity data are not available are evaluated qualitatively and continue to be referred to as COIs.

2.5 Potential Sources

This section provides an overview of known and potential sources of COPCs to the Terrestrial Study Area. The information presented is intended to be a high-level summary of sources for the Terrestrial Study Area. Further discussion and refinement of the baseline physical/chemical CSM for the Terrestrial Study Area will be provided in the Upland RI report.

The Upland BERA evaluates risk from metals (that is, COPCs) that have been introduced by one or more of the sources noted in this section, without attributing any particular source(s) to COPCs. The source activities all present varying degrees of potential stress on natural habitats in the Terrestrial Study Area, and in many locations there may be multiple stressors (chemical and/or physical).

Broadly, the sources of metals to the Terrestrial Study Area can be classified into the following categories by order of magnitude and importance: (1) historical smelter operations; (2) mining and mining-related operations; (3) other anthropogenic sources; and (4) geology and local mineralization (geogenic sources), and (5) ambient atmospheric constituents. These categories are described further in Sections 2.5.1 through 2.5.4.

2.5.1 Historical Smelter Operations

Metals smelting operations located in Trail, British Columbia and Northport, Washington are acknowledged sources of COPCs to environmental media in the UCR site (EPA, 2003a, 2008, 2016c, 2020a). COPCs associated with aerial deposition from historical smelter emissions are found in soil over much of the Terrestrial Study Area. Other media discharged from smelter operations (for example, slag, liquid effluent) are inputs to the aquatic portion of the site and will be discussed in the Aquatic BERA and Aquatic RI report.

2.5.1.1 Trail Smelter

The smelter in Trail, British Columbia has operated under a variety of names for more than 120 years and is one of the largest fully integrated lead and zinc smelters in the world. The smelter is on the western bank of the Columbia River approximately 10 RMs upstream of the U.S.-Canada border (Map 2-4).

Major production at the facility includes primary smelting of zinc and lead and secondary smelting for production of other metal products (for example, antimony, bismuth, cadmium, cobalt, copper, germanium, gold, indium, mercury, silver, and thallium), arsenic products, granular and crystallized ammonium sulfate fertilizers, sulfur, sulfuric acid, sulfur dioxide, and ferrous granules (that is, granulated slag) (EPA, 2003a). The Trail facility operates under current permits from the British Columbia Ministry of Environment, and these permits require monitoring and reporting of select constituents at the stacks as well as ambient air quality monitoring stations. Arsenic, cadmium, copper, lead, and zinc are characteristic of current and historical aerial emissions of metals and may serve as tracers of aerial emissions from the Trail facility.

The original Trail facilities were built in 1896 to smelt copper and gold ores (EPA, 2008). At that time, roasting technology was crude and limited to the heap method where ore was piled up with cordwood and limestone intermixed and set aflame. With such crude processes, the smelter produced a matte of 50 percent copper which was further refined elsewhere, while the lead, which was prevalent within local ores, could not be extracted (Wilson, 2021). With the development of the Betts electrolytic process in 1902, the Trail facility was able to produce pure lead, silver, and gold. Zinc production began in 1916.

By 1925, the facility consisted of a complex of structures housing a lead plant, an electrolytic zinc plant, a foundry, a machine shop, and a copper-rod mill (Wilson, 2021). Fertilizer plants were built at the Trail facility in 1930, and a heavy water plant was constructed and operated from 1944 to 1955 (MacDonald, 1997; Wilson, 2021). Phosphate fertilizers were produced there until August 1994 (MacDonald, 1997). Teck Metals Ltd. became the operator of the Trail facility in 2001.

Aerial sulfur dioxide emissions were historically a significant component of facility emissions in the early years of operations. In response to complaints about the significant negative impacts these emissions had on the environmental quality of region, sulfur dioxide control measures began to be established at the facility in 1928. However, damages to the vegetation continued into the 1930s. Subsequent development and modernization at the Trail facility led to further reductions in emissions in the 1990s.

Historical emissions from the Trail smelter included metal-enriched particulates and aerosols. The airborne emissions were transported by prevailing winds down the Columbia River Valley, where particulate metals were deposited at varying distances from the smelter and became incorporated into the upper soil horizon and in the sediments of lakes and wetlands, as demonstrated by various studies conducted in the Terrestrial Study Area (Ecology, 2011, 2013; EPA, 2016d; TAI, 2015, 2017). Elevated metals concentrations have also been mapped in surface soils collected by Canadian investigators between Trail, British Columbia and the U.S.-Canada border (Goodarzi et al., 2002, 2006; Sanei et. al., 2006).

Sampling activities in the U.S. were focused in an area where damage to livestock, crops, forests, and other vegetation was documented in the 1920s and 1930s. This damage was attributed to sulfur dioxide emissions from the Trail smelter and was the subject of litigation and international arbitration in the 1930s and early 1940s (IJC, 1938, 1941; Scheffer and Hedgcock, 1955). Maps depicting sulfur dioxide damage in the Columbia River Valley in 1929, 1930, and 1931 were prepared by the U.S. Department of Agriculture (USDA) (USDA, 1936). Differing degrees of injury were observed as far south as Kettle Falls and into major drainages on the eastern and western sides of the valley.

While sulfur dioxide is a gas and thus may be carried further than particulates, the sulfur dioxide injury maps were used as reasonable approximations of the area potentially impacted by Trail smelter emissions. The ADA from the Upland Soil Study (TAI, 2014a) generally corresponds to sulfur dioxide injury information presented on the 1931 USDA sulfur dioxide map, with the southern limit corresponding to that of the 60 to 100 percent injury area and the eastern and western boundaries defined by injury within the major drainages in the northern part of the site (for example, Sheep Creek, Onion Creek, Deep Creek, and Cedar Creek).

2.5.1.2 Le Roi Smelter

The former Le Roi Smelter (also referred to as the Le Roi/Northport Smelter) was in Northport, Washington, on the east side of the Columbia River, approximately 7 RMs downstream of the U.S.-Canada border. Beginning in 1896, the Le Roi Smelter initially processed copper and gold tellurium ores. Operations were suspended in 1909 and then resumed around 1916 to process lead ore from Leadpoint, Washington. Smelter operations ceased permanently in 1921. The smelter site remained inactive until 1953. From 1953 through 2001, the site was used as a lumber mill. No wood treatment or chemical use was reported during this period of operation (EPA, 2004a; Ecology, 2019a).

Historical emissions from the Le Roi Smelter that may have relevance to the Terrestrial Study Area include aerial emissions (stack and fugitive) likely aerially transported from the site and deposited onto upland soils in and around Northport. EPA conducted a removal action at the Le Roi Smelter property and in the town of Northport in 2004 (EPA, 2004a, 2005a). Follow-up removal actions were conducted at 16 properties in 2020 and additional properties in 2022 (EPA, 2020c, 2022c). Additionally, Ecology is conducting an investigation and cleanup of smelter-related metals contamination at the Northport Waterfront (Ecology, 2022).

The presence of smelter-associated metals in the Northport area excluded properties in Northport as candidates for sampling during the 2014 and 2016 residential soil studies (EPA, 2014; TAI, 2016). The town was not considered for upland soil sampling in 2014 because ecological habitat is generally absent. The decision to exclude Northport removal action soil data from the Upland BERA due to difficulty in obtaining all relevant records was made during soil data set discussions between EPA and TAI in 2022 (EPA, 2022d).

2.5.2 Mining and Mining-Related Operations

In addition to the smelters described in Section 2.5.1, the Terrestrial Study Area contains a number of mine and mill sites that may have historically contributed to localized occurrences of COPCs greater than regional natural background concentrations in soils.

Mineral resources mined within the UCR site include lead, zinc, gold, silver, copper, cadmium, and barium (Derkey et al., 1990; Moen, 1964). Mining activities may be associated with a range of release and transport mechanisms for COPCs in environmental media, including dust from mining operations and ore processing; dust from haul roads, ore loading, and ore haulage; sediment transport via stormwater runoff and surficial discharge of mine water; and chemical alteration within mineralized zones, mined material stockpiles, and overburden stockpiles.

As documented by the EPA Preliminary Assessments and Site Inspections Report (EPA, 2002a), specific facilities that represent potential sources of COPCs to the UCR site include the Young America Mine and Mill, Van Stone Mine/Mill, Anderson Calhoun Mine/Mill, and Last Chance Mine/Mill sites (Map 2-4). However, subsequent sampling and/or removal actions at these sites showed only localized impacts to environmental media, with limited evidence that COPCs were released to environmental media beyond the site boundaries (EPA, 2007i, 2012b; TechLaw, 2012; Ecology, 2017).

2.5.3 Other Anthropogenic Sources

Other anthropogenic sources or activities within the Terrestrial Study Area, besides those described in Sections 2.5.1 and 2.5.2, that may contribute metals to upland soils include the following:

- Runoff from roads and urban areas
- Wood products manufacturing and/or runoff or discharges from lumber milling
- Mercury use in historical mining activities, including placer mining
- Historical railroad spurs, rail and road transport, and loading/transfer points.

There is uncertainty about the degree to which anthropogenic stressors may have altered habitat in the Terrestrial Study Area, either in conjunction with or independent of COPC concentrations.

2.5.4 Geology and Natural Background

The COPC and COI metals considered in this Upland BERA naturally occur in the bedrock, sediments, and soils of northeastern Washington, and the region is known to have bedrock formations enriched with metal ores and placer deposits in the Columbia River and its tributaries. In the Upland BERA, "background" refers to both "natural" and "area" background.[16] 1.3Consistent with the EPA guidance (EPA, 2007a), background warrants careful consideration when assessing risks from metals (Section 1.3). Methods for evaluating the contributions of background to risks to EAEs are described in Section 5.3.1.

Natural chemical and physical weathering of geological materials that contain metallic minerals and metal ore can mobilize metals in the environment and contribute to metals concentrations found in soil, surface water, and sediments. The effects of weathering and the degree to which some portion of metals in these media may be derived from geogenic materials are expected to vary geographically and are dependent upon local geology, terrain, and other conditions. Additionally, ambient atmospheric constituents are transported to and deposited at the site from global or regional atmospheric sources that are not connected to a specific point source (EPA, 2008) and contribute to natural background of COPC concentrations at the site.

TAI conducted an assessment, under EPA oversight, for the area in and around the site (TAI, 2020a) to ascertain natural background soil concentrations as defined by EPA (EPA, 2002b) and by the State of Washington in the Model Toxics Control Act (MTCA) (WAC 173-340-200). Under MTCA, "natural background" means "the concentration of hazardous substance consistently present in the environment that has not been influenced by localized human activities." This includes metals that "naturally occur in the bedrock, sediments, and soils of Washington State due solely to the geologic processes that formed these materials."

The soil background assessment was completed in accordance with the EPA-approved data quality objectives (DQOs) (TAI, 2018a; EPA, 2018) and EPA's level of effort memorandum regarding the assessment and estimation of background elemental concentrations in upland soils from the UCR site (EPA, 2016c). The soil background assessment resulted in three sets of natural background threshold values (BTVs) for metals,[17] depending on the type of soil sample analysis: one for partial-digestion results, one for total-digestion results, and one for mercury results (TAI, 2020a). For the Upland BERA, partial-digestion and mercury BTVs are used because soil concentrations in the Upland BERA data set were determined using partial digestion and mercury methods.

BTVs were calculated as one-sided 95 percent upper confidence limit of the mean (95 UCL) on the 95th percentiles, or 95/95 upper tolerance limits, of the final data sets using ProUCL 5.1 (EPA, 2015a, 2015b), as specified in the DQOs (TAI, 2018a). Quantile-quantile plots were used to identify anomalous high concentrations that would overly influence the BTVs in conjunction with formal outlier tests. Outliers, if identified, were excluded if the resulting BTV was more conservative (that is, the BTV with outliers excluded was lower than the BTV with outliers included). Table 2-8 summarizes the BTVs derived for each metal for the partial digestion and mercury methods.

2.6 Ecological Exposure CSM

As shown in the ecological exposure CSM (Figure 2-1), the exposure media in the Terrestrial Study Area are air, soil, and biota. Ecological receptors that encounter these exposure media are exposed to chemical constituents in these media if there is a complete exposure pathway. These terrestrial exposure media are the focus of this Upland BERA. Aquatic exposure media (for example, sediment and water) in upland

[16] Natural background concentrations are the result of geogenic sources. Under WAC 173-340-200, "Area background means the concentrations of hazardous substances that are consistently present in the environment in the vicinity of a site, which are the result of human activities unrelated to releases from that site."

[17] BTVs were calculated for the following metals and metalloids as specified in the DQOs: aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, molybdenum, nickel, selenium, silver, thallium, vanadium, and zinc.

lakes and wetlands will be evaluated in the Upland Lakes and Wetlands Addendum to this Upland BERA and aquatic exposure media associated with the UCR will be evaluated in the Aquatic BERA.

2.6.1 COPC Sources

Sources of COPCs in the Terrestrial Study Area are divided into primary, secondary, and tertiary sources. Known and potential primary sources of COPCs released into Terrestrial Study Area soils are described in Section 2.4 and include principally historical aerial emissions from smelter operations, mining and mining-related activities, other anthropogenic sources, naturally occurring metals in area soils (natural background), and ambient atmospheric constituents.

COPCs released from primary sources can undergo a variety of chemical and physical transport and fate mechanisms. These mechanisms result in the distribution of chemicals to environmental media, which then become secondary or tertiary sources. Environmental media considered to be secondary or tertiary sources of COPCs relevant to the Terrestrial Study Area are air, dust, and soils, groundwater, and surface water (Figure 2-1).

Although groundwater and surface water may act as secondary and tertiary sources, they are considered insignificant pathways for terrestrial receptors in the Upland Terrestrial Study Area. Risk to aquatic-dependent receptors in the Upland Terrestrial Study Area via surface water pathways will be addressed in an addendum to this BERA after the Upland Lakes Sediment Study is completed.

2.6.2 Fate and Transport Mechanisms

A variety of physical, chemical, and biological transport and fate mechanisms influence the distribution of metals from their sources to locations throughout the Terrestrial Study Area. Metals may be transported by atmospheric suspension and deposition of metal-bearing particulates through wind dispersion (that is, sorbed to airborne soil particles or other particulate matter) or in biological matrices (that is, bioaccumulated in organisms). Consistent with Principle 4 1.3of the Framework for Metals Risk Assessment (EPA, 2007a; Section 1.3), the metal forms (species) and phases[18] in which they occur influence their transport, fate, and bioavailability. The fate and bioavailability of one form may not be representative of that of a different form of the same metal. Each metal’s form depends on its properties as well as local environmental conditions (for example, temperature, CEC, pH, and total organic carbon [TOC]) and the time since deposition.

The transfer of metals occurs via primary, secondary, and tertiary release mechanisms described as follows and shown on Figure 2-1. The primary release mechanism of metals is the direct discharge of metals from primary sources to environmental media. The potential sources of COPCs to upland soils within the Terrestrial Study Area include current/historical smelter operations, other point and nonpoint anthropogenic sources (for example, industrial, agricultural, logging, municipal), and mining operations. Examples of primary release mechanisms from these sources include fugitive and stack emission of metals and solid waste or tailings disposal (Figure 2-1). Once released to the environment, metals present in environmental media (that is, secondary and tertiary sources: air, dust, and soil) are distributed through secondary and tertiary release mechanisms.

[18] Forms (species) refer to the specific compound (for example, copper ion [Cu^{2+}], copper carbonate [$CuCO_3$]), whereas the phase refers to how it occurs in the environment (for example, dissolved versus particulate, including colloidal and bound to ligands, like humic acids).

The secondary and tertiary mechanisms as they relate to air, dust, and soil are described as follows:

- **Air and dust.** Metals in the air and sorbed to dust particulates can be transported via wind dispersion or aerial deposition. Wind dispersion is the process by which metals are transported locally, regionally, or globally via wind currents to different locations. Aerial deposition is the settling of metals from air or dust to soil, sediment, surface water, or biota via wet or dry deposition. Finally, metal-bearing particulates may be lifted and resuspended in the air from surface soils and may be transported in air currents (wind dispersion) to other locations and redeposited (aerial deposition).
- **Soil.** Metals in soil are subject to physical, chemical, and biological processes. Physical transport processes occur through wind dispersion of soil or particulate matter. Chemical processes include adsorption/desorption (that is, attachment or detachment of metals to soil particles) and physical processes like decrepitation/weathering/erosion (that is, the wasting or breaking up of particles resulting in metal releases). In addition, biological uptake of metals from soil (for example, via incidental ingestion) may lead to the bioaccumulation of metals in tissues of ecological receptors. Partitioning of metals in the soil matrix is dependent upon a number of environmental conditions that can enhance or retard adsorption and desorption of metals to soil particles (for example, temperature, pH, redox conditions, CEC, particle size, moisture content, and organic matter).

2.6.3 Exposure Media

Through fate and transport processes, chemicals are distributed to environmental media present at the site, some or all of which may provide exposures to terrestrial organisms. As shown on Figure 2-1, the exposure media in the Terrestrial Study Area are air, biota, and soil. Ecological receptors that encounter these exposure media are potentially exposed to their chemical constituents. Exposure to COPCs in surface water and sediment from upland lakes and other waterbodies in Upland Study Area will be addressed in an addendum to this BERA after the Upland Lakes Sediment Study is completed.

2.6.4 Exposure Pathways

Pathways through which ecological receptors may be exposed to COPCs in exposure media in the Terrestrial Study Area, are as follows:

- Air—inhalation and foliar uptake
- Biota—dietary consumption
- Soil—incidental ingestion and direct contact (dermal exposure for both invertebrates and wildlife, uptake through plant roots)

Each of these pathways, as they relate to specific ecological receptors relevant to the Upland BERA, are shown on Figure 2-1. Exposure pathways are considered “potentially complete and significant,” “potentially complete but not significant,” or “incomplete or not applicable” for each environmental medium and ecological receptor (as shown on Figure 2-1). The potentially complete and significant exposure pathways are further delineated into pathways that are quantitatively evaluated in this Upland BERA and pathways that cannot be reliably quantitatively evaluated due to lack of data. These exposure pathways and the degree to which they are complete and significant are discussed for each receptor group in the section following.

2.6.5 Ecological Exposure CSM for Each Receptor Group

Receptors are ecological resources potentially affected by COPCs and COIs in soil. This Upland BERA uses the term “receptor group,” which was used in the planning documents to describe a broad collection of organisms that occupy related taxonomic groups and/or share similar biological characteristics (that is, soil invertebrates, mammals, birds, etc.). Based on the habitats, plants, and animals observed within the

Terrestrial Study Area (Section 2.3) and the criteria considered in the BERA Work Plan (TAI, 2011), the following terrestrial receptor groups were identified as receptor groups of concern: terrestrial plants, soil invertebrates, birds, and mammals (TAI, 2011, 2012). Exposure pathways are identified for each receptor group on Figure 2-1. Threatened and endangered species are expected to have the same exposure pathways as their overall receptor group.

The components of the ecological exposure CSM described in the preceding sections are generalized to describe contaminant sources, fate and transport mechanisms, exposure media, and pathways applicable to the COPCs and ecological receptors in the Terrestrial Study Area. However, exposure pathways and media can vary by receptor group. Thus, the ecological exposure CSM described in this section is further detailed to focus specifically on exposure media and pathways relevant to each receptor group evaluated in this Upland BERA, as presented in the following paragraphs.

Most COPC exposure occurring to terrestrial plants in the Terrestrial Study Area is through the root system. Foliar uptake also may occur but is considered a minor exposure route compared to root exposure (TAI, 2011, 2012). Thus, the potentially complete and significant exposure pathway quantitatively evaluated for plants is direct contact with soil (Figure 2-1).

The soil invertebrate community includes soft-bodied animals, such as earthworms (*Opisthopora*) and potworms (*Enchytraeida*), and hard-bodied animals, such as beetles (*Coleoptera*), ants (*Hymenoptera*), and springtails (*Collembola*). Soil invertebrates are exposed to metals in soil through direct contact and incidental soil ingestion and through plants, detritus, and other invertebrates upon which they feed. Some soil invertebrates are soft-bodied and may absorb soil chemicals through the dermis, whereas others have hard, chitinous exoskeletons that limit dermal chemical absorption. Both groups may incidentally ingest soil through their diet (Suter, 2007). The potentially complete and significant exposure pathways to soil invertebrates is direct contact with soil and ingestion of soil (Figure 2-1); of these, direct contact with soil can be quantitatively evaluated. Potentially complete but insignificant exposure pathways include ingestion of biota and inhalation.

Terrestrial bird and mammal species in the five feeding guilds (that is, herbivores, invertivores, aerial insectivores, omnivores, and carnivores) share similar exposure pathways, but concentrations of COPCs differ in the biota they consume. Additionally, bird and mammal species may be exposed to COPCs through the incidental ingestion of soil. In the Upland BERA, the term “dietary exposure pathway” is used to refer to exposures from both ingestion of biota and incidental soil ingestion. The amount of soil ingested varies with the species and how they forage. The following are potentially complete and significant exposure pathways to terrestrial birds and mammals (Figure 2-1):

- Ingestion of biota
- Incidental ingestion of soil

While it is recognized that birds and mammals might also be exposed to COPCs via direct contact with soil and inhalation, these pathways are insignificant relative to the ingestion pathway and not evaluated in this Upland BERA. As described in the BERA Work Plan (TAI, 2011), inhalation and dermal exposures may be minor routes of COPC uptake for birds and mammals. However, methods for calculating exposure to wildlife via these routes are not well developed (EPA, 1993), and it is likely that fur, feathers, and scales minimize direct dermal contact; therefore, inhalation and dermal exposure are not used for estimating risk to terrestrial wildlife. COPC exposure via direct consumption from UCR surface waters is expected to pose an insignificant contribution to total dietary exposure (TAI, 2011). These assumptions are consistent with approaches used for other major Superfund ERAs conducted regionally (for example, for the Bunker Hill site in Idaho and the Hanford site in Washington).

2.7 Ecological Assessment Endpoints, Risk Questions, and Measures of Exposure and Effects

As part of the problem formulation process, EAEs, risk questions and measures of exposure and effect have been identified. These elements of the problem formulation frame the risk assessment by identifying the site-specific ecological resources to be addressed, the questions regarding interactions between COPCs and ecological resources to be answered, and the specific approaches used to answer the questions.

2.7.1 Ecological Assessment Endpoints

EAEs describe both the ecological entity to be protected, such as a species, a functional group of species (for example, insectivores), or a community (for example, soil invertebrates), and the characteristics of the entity to be protected (for example, reproductive success) (EPA, 1998). They are developed based on known information concerning the chemicals present, the ecosystems, communities, and species that occur, and the risk management goals. Development of assessment endpoints aids in clearly defining the goal of the risk assessment and helps to focus associated risk analyses. Consistent with EPA guidance and site management goals (EPA, 1997a, 1998, 1999, 2008, 2016a), EAEs for this Upland BERA are focused on managing ecological risks from COPCs to levels that will not prevent the recovery and maintenance of healthy local populations and communities of biota, in accordance with applicable or relevant and appropriate requirements (ARARs). EAEs are assigned to each receptor group (terrestrial plants, terrestrial invertebrates, birds, and mammals) (Table 2-9), along with representative species within a receptor group (as appropriate) as described in the bullets. The assessment endpoints for all receptor groups are the survival, growth, and reproduction of the subject receptor group. These generic assessment endpoints are consistent with the guidelines discussed here, and therefore provide both practical and specific direction required for the BERA. Other than the endangered gray wolf, which is managed at the individual level, risk managers at the site ultimately are concerned with the potential for COPCs to affect levels of organization higher than the individual organism, such as populations and communities. As such, the ecological entities of the EAEs are either the individual, the population, or the community depending on the specific receptor group and receptor:

- Gray wolf, as a state endangered species is evaluated at the individual level.
- Birds and mammals are evaluated at the local population level. EPA (1997a, 1998) define a population as, “an aggregate of individuals of a species within a specified location in space and time.”
- Lower trophic-level receptors including terrestrial plants and invertebrates are evaluated at the local community level. A community is defined as, “...an assemblage of populations of different species within a specified location in space or time” (EPA, 1997a, 1998).

For the purposes of this BERA, local populations and local communities for all EAEs are defined as the component of the population or community that uses the Terrestrial Study Area or distinct areas within the Terrestrial Study Area.

In two cases, the entire receptor group (soil invertebrates and terrestrial plants) contains a single EAE. For the rest, species and/or populations within a receptor group are differentiated from one another by the habitats where they forage and breed, their position within the food web (that is, trophic level), or by being state-listed as endangered. In these instances, the receptor group is subdivided into separate EAEs based on shared ecological characteristics or composed of an endangered species (for mammals). Birds and mammals are subdivided into EAEs based on feeding guilds[19] (for example, herbivores, insectivores,

[19] Feeding guilds share similar foraging and dietary habits.

carnivores). However, the term "receptor group" (Section 2.6.5) is retained to provide an umbrella term under which similar EAEs are evaluated.

Even with the segregation of receptor groups into separate EAEs, the quantitative assessment of all species within a given population-level EAE would not be practical. Therefore, the BERA Work Plan established the need to identify representative species that act as surrogates for all species within a given feeding guild-based EAE. Criteria used to select representative species for population EAEs based on feeding guilds may vary by receptor groups, but in general, the following were used:

- Suitable representative of the feeding guild based on diet
- Suitable habitat (which is provided by the Terrestrial Study Area), particularly for breeding
- Availability of toxicity data derived from the species
- Human or cultural significance, including status as a federally or state-protected species
- Small body weight relative to other species in the guild (a lower concentration of metals in food is needed to exceed the threshold dose because of a higher bodyweight-normalized food ingestion rate)

The EAEs for each receptor group are identified in Table 2-9 and discussed in the following section.

2.7.2 EAEs for Each Receptor Group

The EAEs specified for each receptor group are summarized in Table 2-9 and described in more detail in the following paragraphs.

2.7.2.1 EAE for Terrestrial Plants

There is one EAE, which is survival, growth, and reproduction of terrestrial plants. Unlike birds and mammals, selection of a representative species is neither necessary for establishing exposure assumptions nor appropriate for the EAE's biological level (community); therefore, no representative species were identified for the terrestrial plant community.

2.7.2.2 EAE for Terrestrial Invertebrates

There is one EAE for terrestrial invertebrates, which is survival, growth, and reproduction of soil invertebrates. As with the terrestrial plant community, a representative species for the terrestrial invertebrate community was not selected.

Flying and foliar-dwelling invertebrates are not evaluated directly in this Upland BERA because direct contact with soil is not a significant pathway for invertebrate species that spend most of their lives on vegetation and flying (for example, butterflies) (Section 2.3.3). Thus, it is assumed that risk evaluations focused on soil-dwelling invertebrates are protective of flying and foliar-dwelling insects since soil dwellers experience a higher degree of exposure to metals in soil.

2.7.2.3 EAE for Birds

Five avian feeding guilds (herbivores, invertivores, aerial insectivores, omnivores, and carnivores) were identified based on those described in the approved BERA Work Plan (TAI, 2011) and expanded problem formation (TAI, 2012), with each guild representing an EAE. By occupying various levels of the food web, these EAEs have value in their existence and play an important role in energy flow through the terrestrial food web by both providing a prey base for other wildlife and preying upon plants, invertebrates, and other wildlife. Each EAE is assessed through survival, growth, and reproduction of a representative species.

Five avian feeding guilds were selected as EAEs for the bird receptor group, and representative species were selected following the considerations described in Section 2.7.1. One species per feeding guild was selected as the representative species. Table 2-3 provides the guild identification for each of the species in the Terrestrial Study Area. Table 2-10 summarizes information on the representative species for each feeding guild EAE, including other bird species represented by the surrogate, composition of the diet, and whether they are present in the Terrestrial Study Area habitat when breeding. The following sections present additional information about each representative species that is relevant to understanding its potential for exposure to COPCs in habitats of the Terrestrial Study Area.

Herbivores

The herbivore bird guild includes species in the order Galliformes, including grouse, pheasant, quail, and turkey, which feed primarily on plants. It also includes smaller birds, such as hummingbirds (*Trochilidae*), rock pigeon (*Columba livia*), American goldfinch (*Carduelis tristis*), crossbills (*Loxia* spp.), and mourning dove (*Zenaida macroura*). California quail (*Callipepla californica*), a smaller species of its guild and a medium-sized quail, was chosen as the representative species of this guild as identified in the BERA Work Plan (TAI, 2011) and the expanded problem formulation (TAI, 2012). This species was chosen because it is present in the Terrestrial Study Area during the breeding season, wildlife exposure factors (WEFs) data are readily available, it has a relatively small home range and body mass, and quail species are used in toxicity experiments. Other herbivorous species represented by the California quail are listed in Table 2-3 and summarized in Table 2-10.

The California quail is a generalist species that is often found near permanent surface waters. Preferred breeding habitats of the California quail that occur in the Terrestrial Study Area include forest edge after a disturbance, open riparian and foothill woodlands, brushy foothills, sagebrush shrubland and steppe, stream valleys, cultivated lands, and urban areas (Calkins et al., 2020; EPA, 1993). This species nests on the ground in a shallow depression lined with vegetation, though it has been noted to nest in trees (BCDCC, 1994). They require nearby perennial cover during foraging, such as areas with broken brush. Chicks and juveniles are vulnerable to predation and forage close to cover (within about 3 feet) during their first few weeks of life (Calkins et al., 2020). Adults forage less than or equal to 300 feet from cover in absence of aerial predators and less than or equal to 50 feet when raptors are present. Their home range can be up to 49 acres. The Terrestrial Study Area of the UCR offers suitable breeding and foraging habitat for the California quail throughout the study area, particularly closer to the river valley and in the shrubby foothills. California quail are not migratory species and may breed in the area. One historical observation has been recorded in the Terrestrial Study Area in 2019 (eBird, 2020).

Feeding techniques of California quail include scratching for seeds; jumping for flowers and buds; pecking at the ground; and shelling acorns and sunflower seeds. Their diet consists largely of seeds and leaves of broad-leafed plants but also includes catkins, flowers, grain, berries, and acorns. California quail will also consume small amounts of soil insects (approximately 1 to 6 percent of their diet). These insects include caterpillar, cricket, beetle, and snail species. In the fall and winter, large groups called coveys (an average of 75 individuals) will forage together (Leopold, 1977).

Clutches of California quail average 12 to 16 eggs (Calkins et al., 2020). Chicks that are less than 3 weeks old consume primarily invertebrate material (Leopold, 1977). At 13 to 16 weeks old, the diet of juveniles becomes the same as that of adults. Males eat less during the breeding season and often have empty stomachs during the day (Calkins et al., 2020).

Invertivores

This invertivore guild comprises birds that feed primarily on invertebrates, such as insects and worms, either by probing at the soil or by foraging on plants, leaf litter, or debris. The BERA Work Plan

(TAI, 2011) and the expanded problem formulation identified the American robin as the representative species for invertivorous birds (TAI, 2012).[20] Western meadowlark, American redstart (*Setophaga ruticilla*), eastern kingbird (*Tyrannus tyrannus*), western kingbird (*Tyrannus verticalis*), warblers (*Parulidae*), and wrens (*Troglodytidae*) are also included in this guild. The American robin is similar in size to the meadowlark and is the largest thrush species in North America. The American robin was chosen for this guild because it is present in the Terrestrial Study Area during the breeding season; it is a commonly used species in ERAs, with readily available WEF data; it has a relatively small home range and body mass; and a component of its diet is earthworms, which may accumulate metals more than other invertebrates depending on the metal and soil properties. Other invertivorous species represented by the American robin are presented in Table 2-3 and summarized in Table 2-10.

American robins primarily nest in open woodlands; moist forests; near streams and rivers; and in urban areas, such as parks, fields, and lawns (Vanderhoff et al., 2020; EPA, 1993). Preferred foraging habitat includes open areas such as meadows, pastures, fields, and edge habitat. They forage on the ground most of the time in these areas and sometimes in shrubs or lower parts of trees (EPA, 1993). A suitable nesting and foraging habitat is found within most of the Terrestrial Study Area. Individual robins have been historically observed in numerous locations in the study area as early as 1993 and yearly since 2006 (eBird, 2020).

The American robin is a resident breeder. Some individuals of this species occur in the Terrestrial Study Area year-round. If individuals do migrate from their breeding area, they return before most other species, arriving back as early as March. American robins will roost and forage together in large flocks outside of the breeding season. They build nests from mud and dead vegetation in trees, shrubs, or on the ground in an area protected from rain (Vanderhoff et al., 2020). Clutch size is three to five eggs, on average (Vanderhoff et al., 2020; Foote et al., 2020). Individuals tend to return to the same territories each year (Howell, 1942).

The foraging home range of the American robin is approximately 5 acres (EPA, 1993). They forage on the ground by probing at the soil for worms, insects, snails, and fruit. Their diet is dependent on season and time of year. Beyer and Sample (2017) concluded that the American robin diet during the breeding season (spring and summer) is 40 percent earthworms, 50 percent other ground-dwelling invertebrates, and 10 percent fruits. This species also eats other invertebrates, such as grasshoppers, flies, crickets, beetles, caterpillars, moths, spiders, millipedes, and some snails. Nestlings are fed mainly caterpillars, and earthworms an average of 34 to 40 times a day (Canadian Wildlife Federation, 2020; Howell, 1942).

During fall and winter, the American robin’s diet changes to mainly fruits and seeds. Commonly consumed fruits include berries from shrubs of the genera *Prunus* (for example, chokecherry), *Rubus* (for example, raspberry) and *Juniperus* (for example, common juniper) (Howell, 1942; Wheelwright, 1986). The proportion of fruit by volume is much higher during this period (greater than 90 percent fruit, less than 10 percent invertebrates) compared to the spring (less than 10 percent fruit, greater than 90 percent invertebrates) (Wheelwright, 1986). Over a period of 1 to 2 months in the summer, the transition between diets occurs, which can result in an equally mixed diet (50 percent fruit, 50 percent invertebrates). American robin habitat has also been shown to affect diet. Individuals in forested areas consume more fruit in their diet than those in open areas, such as agricultural fields, open woodland, and shrubland (Vanderhoff et al., 2020).

[20] Note that although the diet of American robins is greater than 90 percent fruit during fall and winter (Vanderhoff et al., 2020), the diet in spring, during reproduction, is greater than 90 percent invertebrates. This species is therefore adequately representative of insectivorous birds.

Aerial-feeding Insectivores

The aerial insectivore guild comprises birds that forage by flying and feeding primarily on flying insects. This guild includes members of the swallow, flycatcher (*Tyrannidae*), and swift (*Apodidae*) families as well as the common nighthawk (*Chordeiles minor*). The tree swallow (*Tachycineta bicolor*), a medium-sized species in its guild, was selected as a representative species in the BERA Work Plan (TAI, 2011) and the expanded problem formulation (TAI, 2012). This species is present in the Terrestrial Study Area during the breeding season; it is a commonly used species in ERA with readily available WEF data; and it has a small body mass. Other aerial-feeding insectivorous species represented by the tree swallow are presented in Table 2-3 and summarized in Table 2-10.

Tree swallows nest close to bodies of water and open foraging areas that are sheltered from the wind (Winkler et al., 2020). Potential breeding habitats within the Terrestrial Study Area include fields, meadows, marshes, shorelines, and wet wooded areas. Nesting and foraging habitat occurs throughout the study area. Tree swallows have been recorded as early as 1993 in the Terrestrial Study Area, with few records until 2017. Multiple observations were recorded each year from 2017 to 2020 (eBird, 2020).

The tree swallow is a secondary cavity nester; it uses previously excavated holes by primary cavity nesters, such as woodpeckers. Cavity nests can be found in downed trees, broken tree limbs, hollow stumps, cracks in rocks, and occasionally in infrastructure cavities (for example, bridges) (Aitken and Martin, 2008). The female tree swallow collects vegetation from the ground to use as nest material; this normally occurs within 100 feet of the nest site. Clutch size averages four to seven eggs (Winkler et al., 2020). When feeding young, adults forage within 650 feet of the nest (Winkler et al., 2020). The average home range for tree swallows is 3 miles from their nest.

The tree swallow is a migratory species. It breeds in North America and migrates to its wintering grounds in the southern U.S., Mexico, Caribbean, or Central America (Butler, 1988). Fall migration peaks in late September and spring arrival to breeding grounds in the Terrestrial Study Area occurs in early April; this is much earlier than most other migrating birds (Winkler et al., 2020).

The tree swallow diet consists primarily of flying insects, and they often forage over waterbodies. For the purposes of this Upland BERA, however, it is assumed that exposure to metals comes from flying insects with an entirely terrestrial life cycle. Evaluation of exposures from emergent aquatic flying insects in the Terrestrial Study Area will be conducted in an addendum to this Upland BERA after the upland lakes/wetlands sampling and analysis program is completed. The adult and nestling tree swallow diet mostly consists of true flies, dragonflies, mayflies, and caddisflies (Winkler et al., 2020). It may also include ants, bees, beetles, bugs, butterflies, mollusks, moths, spiders, vertebrates, wasps, and, to a lesser extent, roundworms. Prey items vary in size, but items less than 10 millimeters (mm) long are preyed on most often (Winkler et al., 2020). In the nonbreeding season, they may occasionally eat berries and other vegetation. During an average 45-day nesting period, each adult tree swallow will consume approximately 2,000 insects per day (Smith, 2018).

Omnivore

The omnivore guild includes birds that depend and forage on both plant and animal matter. This guild includes families of songbirds, such as chickadees (*Paridae*), nuthatches (*Sittidae*), sparrows (*Passeridae*), thrushes (*Turdidae*), woodpeckers (*Picidae*), jays (*Corvidae*), and vireos (*Vireonidae*). The black-capped chickadee (*Poecile atricapillus*), a smaller species in its guild, was chosen as the representative species for this guild, as identified in the BERA Work Plan (TAI, 2011) and the expanded problem formulation (TAI, 2012). This species was chosen because it is present in the Terrestrial Study Area during the breeding season, WEF data are readily available, and it has a small home range and body mass. Other omnivorous species represented by the black-capped chickadee are presented in Table 2-3 and summarized in Table 2-10.

Both nesting and foraging habitats are found throughout the Terrestrial Study Area. Black-capped chickadees prefer birch and alder (*Alnus* spp.) trees in deciduous forests that provide optimal foraging and nesting habitat (Foote et al., 2020). Clutch size ranges from 1 to 13 eggs (Foote et al., 2020). The black-capped chickadee has a home range of approximately 7 acres (Stefanski, 1967). They have been recorded as early as 1993 in the Terrestrial Study Area and every year since 2006 throughout all seasons (eBird, 2020). This species is a year-round resident and does not migrate during the nonbreeding season. Large movements (irruptions) occur irregularly every 2 years or more, although these events are not deemed as true migration and are likely caused by lack of food or potentially caused by habitat destruction (Bock and Lepthien, 1974).

Black-capped chickadees primarily forage in trees and may also catch insects while flying (Moreno, 1990). Adults feed on a combination of terrestrial invertebrates, fruits, and seeds. During the breeding season, approximately 80 to 90 percent of their diet comprises soil invertebrates, and the remaining 10 to 20 percent are seeds and fruits (Foote et al., 2020). Most of their diet is caterpillars but also includes spiders, snails, slugs, and centipedes. Fruits consumed include honeysuckle (*Lonicera* spp.) and blueberries. In the winter, their diet is equally mixed, with roughly 50 percent soil invertebrates and 50 percent plants (seeds and fruits). This diet consists primarily of insects, spiders, seeds from conifers and herbaceous species, berries, when available, and occasionally dead animal fat (for example, deer, skunk, and fish) (Foote et al., 2020). Black-capped chickadees will cache their food and can recover it up to 28 days later. Nestlings are mainly fed caterpillars and some spiders, larvae, termites, butterflies, flies, and pupae (Foote et al., 2020).

Carnivores

The carnivore guild comprises birds that forage on animal matter, including raptors, such as hawks and eagles (*Accipitridae*), falcons (*Falconidae*), and owls (*Strigidae*), as well as the turkey vulture (*Cathartes aura*). The American kestrel (*Falco sparverius*) was chosen as the representative species of this guild as identified in the BERA Work Plan (TAI, 2011) and the expanded problem formulation (TAI, 2012). It is the smallest of the raptors and is a year-round resident in the Terrestrial Study Area (Smallwood and Bird, 2020). This species is present in the Terrestrial Study Area during the breeding season, it is a commonly used species in ERAs with readily available WEF data, and it has a relatively small body mass among raptors. Other carnivorous species represented by the American kestrel are presented in Table 2-3 and summarized in Table 2-10.

Suitable nesting and foraging habitat for the American kestrel is present in the Terrestrial Study Area and includes semi-open habitat with short ground vegetation and taller woody vegetation interspersed throughout (Smallwood and Bird, 2020). This encompasses meadows, grassland, open parkland, fields, urban, and suburban areas. Single dead trees, multiple interspersed trees, or nearby forest patches are required for nesting and perching. Large (greater than 60 acres) open areas with short vegetation are required for foraging. Most of the suitable habitat in the Terrestrial Study Area is close to the river valley. American kestrels have been recorded as early as 1993 in the Terrestrial Study Area and yearly since 2006 (eBird, 2020).

The American kestrel is a secondary cavity nester; they use previously excavated woodpecker cavities or naturally created cavities. They can also nest on buildings or in stream banks (EPA, 1993). Favored nesting locations include large deciduous snags prone to decay, such as cottonwoods (*Populus* spp.) (Rohrbaugh and Yahner, 1993). The same nest site is occasionally used in subsequent breeding seasons (Smallwood and Bird, 2020). Breeding population densities are low in the U.S., around 0.010 nest per acre (EPA, 1993). The average clutch size is four to five eggs (Smallwood and Bird, 2020). The

American kestrel home range is less than 1,200 acres and can be as small as 500 acres in a productive foraging area (Craighead and Craighead, 1956).

Both adults and nestlings feed on a combination of terrestrial arthropods and small vertebrates, such as worms, spiders, scorpions, beetles, other large insects, amphibians, reptiles, and a wide variety of small to medium-sized birds and mammals. Diet composition in the summer is approximately 33 percent invertebrates, 33 percent mammals, 31 percent birds, and 3 percent reptiles (EPA, 1993). They favor terrestrial arthropods such as grasshoppers, and, in their absence, small terrestrial mammals such as rodents and bats (EPA, 1993). During the winter, mammals and birds comprise most of their diet. When hunting, American kestrels scan for their prey from a high perch or in the air before diving to capture it on the ground. They occasionally forage in the air for insects. American kestrels will also cache their prey to be used during periods of unfavorable weather or to meet the needs of a growing brood. Depending on the time of year and amount of energy expended while hunting, individuals can consume approximately four to eight voles a day (RSPB, 2020). A group of five nestlings are fed about 2 or 3 items per hour, up to 40 items per day, with a total biomass of 165 grams per day (Smith et al., 1972).

2.7.2.4 EAE for Mammals

Mammalian feeding guild-based EAEs are the same as those identified for birds: herbivores, aerial insectivores, invertivores, omnivores, and carnivores. In addition to the five feeding guild EAEs, one EAE for threatened and endangered mammalian species is evaluated. This EAE focuses on the value of the federally and state-listed endangered gray wolf as a regulator of ungulate population sizes and contributor to mammalian species biodiversity in the Terrestrial Study Area.

Each mammal EAE is also assessed through survival, growth, and reproduction of a representative species (Table 2-9). Representative species for each feeding guild were selected following the considerations described in Section 2.7. One species per feeding guild was selected as the representative species, with the exception of carnivorous mammals, where the gray wolf was added because it has status as a state-listed endangered species (Section 2.3.7; Table 2-6). Table 2-4 provides the guild identification for each of the species in the Terrestrial Study Area. Table 2-11 summarizes information on the representative species for each feeding guild EAE, including other mammal species represented by the surrogate, composition of the diet, size of home range, and whether they are present in the Terrestrial Study Area when breeding. The following sections present additional information about each representative species that is relevant to understanding its potential for exposure to COPCs and COIs in habitats of the Terrestrial Study Area.

Herbivores

The herbivorous guild comprises species that feed on vegetation and includes lagomorphs (that is, hares, rabbits, and pikas), rodents (porcupine [*Erethizon dorsata*], hoary marmot [*Marmota caligata*], voles, and gophers), beaver, and ungulates such as deer, moose, and elk. The meadow vole (*Microtus pennsylvanicus*) was selected as the representative species for herbivorous mammals, as identified in the BERA Work Plan (TAI, 2011) and the expanded problem formulation (TAI, 2012). This species was chosen because it is present in the Terrestrial Study Area, it is a common species used in ERAs with WEF data readily available, and it has a small home range and body mass. Other herbivorous mammals represented by the meadow vole are presented in Table 2-4 and summarized in Table 2-11.

The meadow vole is found throughout the northern half of the U.S. and is the most widely distributed small herbivore in North America (EPA, 1993). It requires habitats with minimal woody plants and lots of grass cover for runways. It primarily moves through corridors in matted grasses to forage and is also adapted to underground and semiaquatic habitats (Johnson and Johnson, 1982). The meadow vole requires loose organic soils to burrow just beneath the surface of the ground or above the surface in grasses or other herbaceous vegetation. Suitable habitat includes grasslands and wet areas, such as marshes, wet meadows, and grassy areas along rivers and lakes (Getz, 1961) and is found throughout the Terrestrial Study Area.

The home range of the meadow vole can be up to about 2.5 acres; the actual size depends on a variety of factors, such as season, habitat, population density, age, and sex. Summer ranges tend to be larger than their winter ranges, and ranges of individuals inhabiting marshes tend to be larger than those that inhabit meadows (Getz, 1961). Population density decreases cause home range size to increase. Territories of males overlap those of other males and females and are approximately three times larger than female territories (EPA, 1993). Females defend territories against each other and do not overlap territories of the same sex.

Like many small mammals, density-dependent factors influence their population dynamics. Meadow vole populations fluctuate annually, with peak densities at 2- to 5-year intervals (Sullivan, 1996). A number of different causal factors may influence population density, including food quality, predation, climate events, density-related stress, and genetically determined behavioral variants (Sullivan, 1996).

Meadow voles do not hibernate or experience torpor like other mammals, and they are active year-round (EPA, 1993). They breed multiple times throughout the year, with a peak breeding period in spring and fall. Number of litters varies and increases with decreasing latitude (Johnson and Johnson, 1982). They become more active during the day in the winter months, while they are more active during the night in the summer (or when temperatures rise above 20°C) to help with temperature regulation.

The diet of the meadow vole consists of 98 percent terrestrial vegetation and 2 percent soil invertebrates (Lindroth and Batzli, 1984, referenced in EPA, 1993). This includes fresh grasses, sedges, herbs, seeds, grains, fungi, roots, bark from small shrubs/trees, and sometimes arthropods or insects (Johnson and Johnson, 1982). Diet varies by season and habitat, but the meadow vole eats primarily vegetation and shows a preference for young, tender vegetation (EPA, 1993).

The meadow vole is prey for hawks, owls, kestrels, and mammalian predators, such as short-tailed shrews (*Blarina brevicauda*), badgers (*Taxidea taxus*), and foxes.

Invertivores

The invertivore guild comprises mammals that feed primarily on invertebrates, such as insects and worms, either by probing at the soil or by foraging on plants, leaf litter, or debris. This guild includes shrew species (*Sorex* spp.). The masked shrew (*Sorex cinereus*) was chosen as the representative species for this guild because it is present in the Terrestrial Study Area, WEF data are readily available, it has a small home range, and it has a small body length and mass. This species also has a very high food and incidental soil ingestion rate for its body size compared to other small mammals, and because it consumes soil, invertebrates in soil, and leaf litter, it has a high degree of exposure to metals in soil. Thus, it is a conservative surrogate for other invertivorous mammals. Other invertivorous species represented by masked shrew are presented in Table 2-4 and summarized in Table 2-11.

The masked shrew is the most widely distributed shrew in North America. It occupies numerous habitat types, and suitable habitat is present in the Terrestrial Study Area. It prefers drier coniferous forests but can also inhabit open meadows, woodlands, avalanche slopes, riverbanks, lakeshores, bogs, and willow thickets (Junge and Hoffmann, 1981). Habitat suitability depends on the availability of water and the highest population densities can be found in moist environments. The masked shrew also thrives in disturbed habitats, such as those disturbed by fire or logging (MFWP, 2020b). It hunts primarily on the ground but may also climb into low vegetation and shrubs or dig in loose substrate (van Zyll de Jong, 1983).

Shrews have high metabolic rates and can eat approximately their body weight in food each day (EPA, 1993). In general, the masked shrew consumes a variety of invertebrates, including insect larvae, ants, beetles, crickets, grasshoppers, spiders, harvestmen, centipedes, slugs, and snails. Seeds and fungi are also consumed (Nagorsen, 1996). The masked shrew is also an important predator of forest insect pests, such as jack pine budworm (*Choristoneura pinus*) and larch sawfly (*Pristiphora erichsonii*). Reported predators of the masked shrew include garter snakes (*Thamnophis* spp.), domestic cats (*Felis catus*), and hawks.

Masked shrew young are born in summer (June through September) and overwinter before reproducing in the spring of the following year. Most overwintered adults breed and die before autumn, with few individuals living beyond 16 months. The average litter size is 6.5 (standard deviation of 1.67), with an average of two litters per adult female (Whitaker, 2004).

Masked shrews, like most shrew species, are territorial (Whitaker, 2004). The average home range of the masked shrew is 1.5 acres (Nagorsen, 1996). Shrews will aggressively defend their territories, and territory size is positively correlated with habitat quality (Lima et al., 2002; Wang and Grimm, 2007; Whitaker, 2004). For the masked shrew, home range size decreased as density increased, consistent with the highly territorial behavior seen in this species (Whitaker, 2004). In an evaluation of shrew abundance and density in the genus *Sorex*, Smallwood and Smith (2001) found that shrew densities ranged from 0 to 17,667 shrews per km^2, with an average density of 1,344 shrews per km^2.

Aerial Insectivores

The aerial insectivore guild comprises bat species that forage by flying and feeding primarily on flying invertebrates such as insects. The BERA Work Plan (TAI, 2011) and the expanded problem formulation (TAI, 2012) identified the little brown bat (*Myotis lucifugus*), a small bat species, as the representative species for mammalian aerial insectivores. This species was chosen because it is present in the Terrestrial Study Area, WEF data are readily available, and it has a small body mass. Other aerial insectivorous species represented by the little brown bat are presented in Table 2-4 and summarized in Table 2-11.

Little brown bats can be found over a wide latitudinal and elevation range. Bat species require different types of habitat for foraging, roosting (day, night, and maternal), and hibernating (hibernacula). Roosts and hibernacula are chosen based on stable ambient temperatures (Havens, 2006). Roosting habitat is occupied during the spring, summer, and fall months, and can include building interiors, roofs, and attics; tree cavities; the underside of rocks; and piles of wood. Roosts are selected to be in proximity to foraging habitat. Suitable roosting and foraging habitats exist within the Terrestrial Study Area. This includes forested areas and urban structures near riparian woodlands or rivers.

Day roosts have very little or no light, provide good shelter, and typically have southwestern exposures to provide heat for arousal from daily torpor (Havens, 2006). The little brown bat travels between 1 to 14 kilometers from day roosts to nightly foraging sites (Nowak, 1994; WDFW, 2023). Night roosts are used by little brown bats as temporary stops when foraging to rest and digest their prey. These roosts are selected for their confined spaces, where large concentrations of bats can cluster to increase the temperature in the roost. These roosts are primarily occupied when temperatures are below 15°C. Night roosts are separate from day roosts, which may diminish the accumulation of feces at day roosts and avoid signaling predators (MFWP, 2020c). Maternal roosts are similar to day roosts but are warmer than ambient temperature. They are occupied by females and their offspring and are used every year. Females have one young per year, born in late June or July. The little brown bat is not territorial, and females are found in maternal colonies of 12 to 1,000 individuals or more (MFWP, 2020c).

Hibernacula are used during the winter months and include abandoned mines or caves where the temperature is always above freezing and humidity is high. The little brown bat enters hibernation in

September to November and will emerge in March to May. This species does not make very long migrations during the change of seasons (Wilson and Ruff, 1999), although individuals can travel up to 100 miles. There are multiple inactive mines and caves within the Terrestrial Study Area that could potentially act as hibernacula.

Little brown bats forage on insects over water or open habitat, feeding on swarms of insects while flying. Evaluation of exposures from emergent aquatic flying insects in the Terrestrial Study Area will be assessed in an addendum to this Upland BERA that will be prepared following completion of the Upland Lakes Sediment Study. For the purposes of this Upland BERA, bats are assumed to be exposed to metals in flying insects of terrestrial origin. They primarily consume midges but will also feed on beetles, caddisflies, mayflies, moths, lacewings, and occasionally mosquitoes (Havens, 2006). Individuals return to areas where they have had prior feeding success. This species consumes insects with length ranges from 3 to 10 mm. It eats half of its body weight per night (when active), and lactating females eat approximately 110 percent of their body weight per night (Edythe et al., 1977). Females are larger in size than males. Food digestion is fast; it takes 35 to 54 minutes to pass through the digestive system (Havens, 2006). Predators include marten (*Martes americana*), fisher, hawks, owls, mice, weasels (*Mustela* spp.), snakes, raccoons, and domestic cats.

Omnivores

The omnivore guild includes the striped skunk (*Mephitis mephitis*), squirrels, chipmunks, raccoons, bears, and rodents (mice and rats) that depend and forage on both plant and animal matter. The deer mouse (*Peromyscus maniculatus*), a small species in its guild, was selected as the representative species of this guild, as identified in the BERA Work Plan (TAI, 2011) and the expanded problem formulation (TAI, 2012). This species was chosen because it is present year-round and breeds in the Terrestrial Study Area, it is a common species used in ERAs and WEF data are readily available, and it has a small home range and body mass. Other omnivorous species represented by the deer mouse are presented in Table 2-4 and summarized in Table 2-11.

The deer mouse is the most widespread and geographically variable rodent species in North America (Millar, 1989). It is a year-round nocturnal resident and is common in most dry land habitat that is within its range. Suitable habitat in the Terrestrial Study Area includes dry mesic habitats, such as upland mixed and cedar forests; deciduous forests; ponderosa pine forests; other coniferous forests; alpine habitats; meadows; and grasslands (EPA, 1993). This species favors both heavy ground- and mid-story cover in addition to open areas with sparser vegetation for foraging (Vickery, 1981; Kaufman and Kaufman, 1989). The deer mouse occupies multiple nest sites that are frequently constructed just below ground level or near the ground in stumps, logs, brush piles, tree cavities, reconstructed bird nests, tree bark, or even cottages or outbuildings. Nests are made of rounded masses of vegetable matter, as much as 100 mm in diameter (Baker, 1983).

The deer mouse home range varies from 0.05 to 0.74 acre, based on population density and sex (EPA, 1993). Females become increasingly territorial of their foraging and nesting ranges when population density increases (Wolff, 1989). Male home ranges are larger than those of females and overlap other individuals' ranges. This species is preyed on by owls, hawks, snakes, and carnivorous mammals.

Deer mice eat primarily seeds, arthropods, some green terrestrial vegetation, roots, fruits, and fungi as available. In a year, the deer mouse eats equal amounts of terrestrial plants and soil invertebrates (EPA, 1993). During the spring, summer, and fall, butterfly and moth larvae are a large portion of their diet (Whitaker, 1966). In the winter, deer mice mainly consume seeds. During both spring and winter, miscellaneous vegetation accounts for a larger percentage of their diet as well. The non-seed plant materials provide most of their daily water requirements (MacMillen and Garland, 1989). The deer mouse will also cache food, such as invertebrates, seeds, and fruit, during the fall and winter.

Females have 1 to 11 offspring; they breed every 3 to 4 weeks in warmer months and less often in winter. Food digestibility and assimilation for most of its diet is estimated to be high (88 percent) (EPA, 1993). Like voles and shrews, population regulation in deer mice is complicated and likely driven by strong density-dependent factors. For example, in British Columbia, deer mouse populations were found to be related to pulses in food supplies and high abundance was related to high early juvenile productivity (Sullivan et al., 2022).

Carnivores

The carnivore guild includes badger, bobcat (*Lynx rufus*), coyote (*Canis latrans*), fisher, red fox (*Vulpes vulpes*), gray wolf, Canada lynx, marten, mink, cougar (*Puma concolor*), weasels, and wolverine. The short-tailed weasel was chosen as the representative species for this guild, as identified in the BERA Work Plan (TAI, 2011) and the expanded problem formulation (TAI, 2012). The short-tailed weasel is one of the smallest mammals in the guild. It was chosen because it is present in the Terrestrial Study Area, it has readily available WEF data, and it has a small body mass compared to other carnivores in the study area. Other carnivorous species represented by the short-tailed weasel are presented in Table 2-4 and summarized in Table 2-11. Additionally, the gray wolf is evaluated separately in this Upland BERA because it is a federally and state-listed endangered species (WDFW, 2023b).

Short-Tailed Weasel

The short-tailed weasel is a solitary nocturnal mammal that inhabits open woody areas near water (EPA, 1993). Preferred habitats within the Terrestrial Study Area include riparian woodlands, marshes, and open areas adjacent to shrub rows or forest borders that are in proximity to bodies of water. This species is primarily ground-dwelling but can also climb trees and swim. Dens are constructed approximately 12 inches underground in existing rodent burrows, under stone walls, tree roots, or hollow logs (Loso, 1999). The short-tailed weasel forages in underground tunnels. The home range for this species is no more than 49.5 acres (Loso, 1999). Females have a smaller home range and remain close to their birthplace throughout their lives, whereas the males disperse and attain large territories that overlap each other. As prey abundance increases, population density also increases, which results in smaller home ranges (Wilson and Ruff, 1999). The short-tailed weasel is present in the Terrestrial Study Area year-round. It breeds once a year and has 3 to 18 offspring.

The diet of the short-tailed weasel comprises small (rabbit-size or smaller), warm-blooded terrestrial vertebrates (Eder, 2002). These can include voles, shrews, rats, chipmunks, nestlings, and rabbits. When these prey items are scarce, their diet can also include eggs, frogs, fish, and insects. In the winter, short-tailed weasels may feed entirely on small rodents (King, 1983). Females weigh less than males and tend to eat the smaller mammals. Side cavities of burrows are used as latrines and food caches to stock up for the high energy and heat production demands of this species (Wilson and Ruff, 1999). Large carnivores are predators of the short-tailed weasel; this includes red fox, martens, fishers, badgers, raptors, and occasionally domestic cats.

Gray Wolf

The gray wolf is one of the most wide-ranging land mammals in North America, occupying a wide variety of habitats that include a sufficient year-round presence of ungulates and alternate prey; suitable and somewhat secluded denning and rendezvous sites; and sufficient space with minimal exposure to humans or disturbances (Mech, 1999; USFWS, 1987). Preferred habitats are found in a large portion of the Terrestrial Study Area; this includes all forested, woodland, river valley, meadows, or alpine habitats that are distant from towns and roads. In the winter, territories are established at lower elevations, wherever prey are most abundant (MFWP, n.d.a).

The gray wolf is a highly social animal that lives and hunts in packs, ranging from 2 to 36 individuals but typically containing 5 to 9 individuals (Mech, 1999). Their summer home range is smaller than their winter range. The approximate home range average is 54,000 acres, but there can be a large degree of variability (USFWS, 2002). Gray wolves are a year-round resident of the Terrestrial Study Area that does not migrate; however, they may move seasonally to follow migrating ungulates within their territory (MFWP, n.d.a). Males can move, on average, 280,000 acres from their natal territory, whereas females disperse approximately 19,000 acres and may establish a new territory or join another pack. Dispersal peaks twice a year during January/February and May/June (Boyd and Pletscher, 1999).

The diet of the gray wolf consists of mammals. Large ungulates, such as elk, deer, and moose, make up 90 percent of their diet (USFWS, 1987; Wiles et al., 2011; Stahler et al., 2006). Wolves will occasionally hunt smaller prey, such as beavers, rodents, and rabbits; hunt livestock; and scavenge on carrion. The gray wolf can consume up to 9 kilograms of meat in one meal, feeding on the prey's entire carcass, including some hair and bones (Smith, 2002). Few animals prey on gray wolves. They are highly territorial animals, and occasionally wolves from other packs or coyotes will prey on lone or young individuals.

2.7.3 Risk Questions and Measures of Exposure and Effects

A risk question is an operational statement of an investigator's research assumption made to evaluate logical or empirical consequences (EPA, 1997a, 1998). Risk question(s) frame the risk assessment conducted for each EAE, with the overall goal of meeting the National Contingency Plan criterion of maintaining protection of the environment. Risk questions provide working hypotheses about the relationship between the assessment endpoints and the responses of receptors when exposed to chemicals at the site.

Measures of exposure and measures of effects provide the specific basis for evaluating risk to EAEs associated with exposure to COPCs via pathways identified in the CSM (EPA, 1997a).

Candidate measures of exposure and effects identified in the expanded problem formulation (TAI, 2012) have been refined in this Upland BERA to conform to the specific effects data used to assess risk. The risk question and associated measures for each EAE are presented in Table 2-12.

Two receptor groups, terrestrial plants and soil invertebrates, have two risk questions each, whereas birds and mammals have one risk question each. Each risk question and associated measures constitute a line of evidence (LOE) used to characterize risk. As described in Section 4, for terrestrial plants and soil invertebrates LOE 2 (COPC concentrations in soil compared to bioavailability-adjusted benchmarks) provides refinements to screening-level measures of exposure and effect employed in LOE 1 (COPC concentrations in soil compared to bulk soil screening-level benchmarks) that result in more accurate predictions of site-specific risks.

3. Upland BERA Data Set

The site-specific data sets used for the Upland BERA are derived from soil chemistry and plant tissue chemistry data collected in support of the broader UCR RI/FS; soil analyses by Washington State Department of Ecology (Ecology); and ecosystem classification data developed by NatureServe, the U.S. Geological Survey (USGS), and WDNR. These data sets were compiled and reviewed in data usability assessments (DUAs) within the context of evaluating ecological risk in the Terrestrial Study Area. The environmental chemistry DUA is presented in Appendix A. This section summarizes the DUAs and describes how data are managed and treated for use in the Upland BERA.

In addition to the soil chemistry data sets used to assess risk in the Terrestrial Study Area, a soil background evaluation was conducted to establish BTVs used in the Upland BERA. The data set used to derive the BTVs was a data set requested by EPA and that underwent a separate DUA in the soil background assessment (TAI, 2020a). The background data set and its DUA are described in detail in the background assessment technical memorandum (TAI, 2020a). Uncertainties associated with the background data set are discussed in the receptor-specific risk characterization sections, where relevant to the ecological risk analyses.

3.1 Data Usability Assessment

Data used to characterize risk in the Terrestrial Study Area of the site were identified through a four-step DUA performed consistent with EPA guidance (EPA, 1992), summarized as follows and detailed in Appendix A:

- **Step 1—Data inventory.** Identify all studies with data potentially relevant for use in the Upland BERA. For environmental chemistry data, a preliminary screening was performed before bringing applicable data sets into this inventory.
- **Step 2—Data quality assessment.** Assess whether data identified in Step 1 are of acceptable quality for use in the Upland BERA.
- **Step 3—Data suitability assessment.** Assess relevance of sampling locations and types of measurements and the reliability of sampling and analytical methods relative to the study objectives.
- **Step 4—Data comparability assessment.** Determine whether data collected from different studies or using different methods can be combined for specific evaluations.

The data inventory and data quality assessment (Steps 1 and 2) determine whether relevant data are available and whether data are of acceptable quality to support risk assessment applications. Data found to be of acceptable quality were carried forward to the suitability evaluation.

For Step 3, the suitability evaluation, the relevance of the data is considered relative to the risk questions (Section 2.7.3), particularly the piece of evidence (POE)-specific analysis questions (Table 2-12). The specific suitability of data (for example, suitability based on sample locations) for each type of analysis in the Upland BERA are further discussed, if necessary, prior to the analysis (for example, for HQs in Section 0, for bioavailability factors in Appendix E [Attachment E3], and for bioaccumulation models in Appendix C).

In Step 4, the data comparability assessment, methods for sample collection, handling, and analyses are evaluated to determine whether data sets from different studies are similar enough in study design and analytical method to be combined for a specific analysis, and whether such combinations are appropriate given the particular analysis being conducted.

Sections 3.1.1 through 3.1.3 summarize the results of this four-step DUA for the three broad types of site-specific data used in the Upland BERA (provided in detail in Appendix A): soil chemistry data, in vitro bioaccessibility (IVBA) assay data, and bioaccumulation data. Table 3-1 summarizes the DUA results for the environmental chemistry data, including soil chemistry data, IVBA data, and bioaccumulation data.

3.1.1 Soil Chemistry Data

Step 1—Soil chemistry data sets included in the data inventory were restricted to data sets that included samples that met the preliminary screening criteria, which were samples sieved to less than 2-mm grain size; surface samples collected from depths within the 0- to 12-inch bgs interval, in accordance with EPA guidance on the soil biotic zone (EPA, 2015c); and soil samples collected above the Columbia River's pre-1973 maximum flood extent topographic elevations, as defined for the 2014 UCR Upland Soil Study Quality Assurance Project Plan (QAPP) (TAI, 2014a).[21] Other soil data sets collected from within the Terrestrial Study Area that did not meet these criteria were excluded, as documented in Appendix A. The following three relevant soil studies were identified during the data inventory step:

- **2014 UCR Upland Soil Study (TAI, 2015).** In 2014, TAI collected 215 soil samples[22] by incremental composite (IC) soil sampling within 171 DUs in the Terrestrial Study Area; of these, 142 decision units (DUs) were evaluated in the Upland BERA and the remainder will be evaluated in the Aquatic BERA. DUs ranged from 7 to 25 acres in size, with most being approximately 25 acres.
- **2012 Ecology Upland Soil Study, Stevens County, Washington (Ecology, 2013).** In 2012, Ecology collected 131 fourpoint composite samples from 106 locations within 13 subareas of the site. Each composite sample was obtained from an approximately 0.025-acre area. Spatially, samples were located within the area covered by the 2014 UCR Upland Soil Study (TAI, 2015) near the U.S. Canada border. Vertical profile core samples were also taken from 13 locations, co-located with the fourpoint composite samples.
- **2015 Bossburg Study (TAI, 2016.)** In 2015, TAI collected eight IC samples within six soil DUs at Bossburg Flat Beach or near the former cable ferry landing. Soil DUs ranged from 1 to 3 acres in size. Discrete soil core samples were also obtained from 18 locations within the 6 soil DUs. As part of the same study, TAI also sampled sediments from 10 DUs on or adjacent to the Bossburg Flat Beach and Evans Campground Beach using IC sampling. Chemistry data for these samples will be evaluated in the Aquatic BERA.

Step 2—Data quality documentation was available for all three soil studies and the data were deemed acceptable for use in the Upland BERA.

Step 3—Chemistry data from all surface soil IC or composite samples sieved to less than 2 mm prior to analysis from the 2014 UCR Upland Soil Study, the 2015 Bossburg Study, and the 2012 Ecology Upland Soil Study were considered suitable for addressing the risk questions (Section 2.7.3). These three data sets contain samples collected and analyzed using standard methods appropriate for environmental risk assessment (TAI, 2015; Ecology, 2013; TAI, 2016). Sample locations were in the Terrestrial Study Area near the U.S.-Canada border or near Bossburg Flat Beach; both areas are considered potential habitat for terrestrial plants, soil invertebrates, birds, mammals, and herpetofauna. Samples from each of the three data sets are suitable for answering the risk questions presented in Section 2.7.3. Vertical profile core samples that were co-located with the IC or composite samples from the 2015 Bossburg Study and the 2012 Ecology Upland Soil Study were collected. While some of these vertical soil profile samples

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[21] Sediment samples and soil samples collected below the pre-1973 maximum flood extent of the river will be evaluated in the Aquatic BERA. This includes soil samples collected for the Northport waterfront RI/FS (Ecology, 2019a).

[22] EPA directed TAI to use the IC soil sampling approach for collecting soil samples in the Upland Soil Study.

were collected at depths meeting the criteria (within 0 to 12 inches), those individual core locations are within the DUs and the chemistry within the DUs is represented by the IC samples. Therefore, these vertical profile samples are not suitable for use in the Upland BERA because they are considered geographically redundant.

Step 4—The 2014 UCR Upland Soil Study, 2012 Ecology Upland Soil Study, and 2015 Bossburg Study sampled soil to similar depths, used similar analytical methods, and collected soil samples within 3 years of each other. However, collection methods differed between studies. While each study collected composite samples, the area over which composites were taken as well as the total number of samples composited differed substantially. The 2014 UCR Upland Soil Study used an incremental sampling method that composited approximately 30 samples from an approximately 25-acre area; the 2015 Bossburg Study used incremental sampling method to composite approximately 30 samples from 1- to 3-acre areas; and the 2012 Ecology Upland Soil Study collected soil samples from four points within an approximately 0.025-acre area to create the four-point composite samples used in that study. In addition, the 2012 Ecology Upland Soil Study sample analyses did not include some of the physiochemical data needed to calculate bioavailability-adjusted benchmarks (Section 4.2.1). Each of these data sets provide a different visualization of exposure and, in combination, provide a more robust representation of risk than any single data set does alone. In terms of spatial coverage of sample locations, while the 2014 Upland Soil Study data have the broadest spatial extent, they also have the lowest density of sample locations. The 2012 Ecology Upland Soil Study data provide an increased density of samples in the northern portion of the ADA immediately adjacent to the border where the highest metals concentrations were identified. Similarly, the 2015 Bossburg Study data provide a higher resolution representation of contamination in the vicinity of another potential source, transfers of ore and concentrate at the Bossburg ferry landing. All three data sets are therefore used for estimating exposure and calculating HQs.

Summary — Based on the updated DUA for this Upland BERA, it is concluded that the soil chemistry data sets (2014 UCR Upland Soil Study, 2012 Ecology Upland Soil Study, and 2015 Bossburg Study) are fully acceptable for evaluating the risk questions identified in Section 2.7.3 for the Terrestrial Study Area.

3.1.2 In Vitro Bioaccessibility Data

Step 1—Similar to the soil chemistry data in the previous section, a preliminary screening was applied before the data inventory for IVBA data. Data were included in the IVBA data inventory if at least one of the following criteria were met: (1) there were available IVBA data for soil sample locations included in the soil chemistry data sets in the previous section, or (2) there were co-located IVBA data and either pH or TOC data that could be used in lead-specific or zinc-specific regression equations (Appendix E). Following the preliminary screening, the following three studies were included in the IVBA data inventory:

- **2009–2011 Beach Sediment Study (TAI, 2014b).** This study included the collection of co-located IVBA (lead and arsenic), TOC, and pH data from beach sediment samples, which were used in the regression analysis for lead and pH.[23]
- **2014 UCR Upland Soil Study (TAI, 2015).** This study included the collection of both sample-specific IVBA data from upland soil samples and co-located IVBA lead/pH and IVBA zinc/TOC data from soil samples used in the regression analysis.

[23] Although the focus of this BERA is on upland soils, beach sediment data were included in the analyses of the relationship between bioaccessibility and soil/sediment pH (Appendix E). Inclusion of sediment data broadens the pH range (UCR sediment pH is generally greater than that in UCR soils; Appendix E, Figure 6) considered and improves predictive utility of the resulting regression model.

- **2015 Bossburg Study (TAI, 2016).** This study included the collection of both sample-specific IVBA data from Bossburg Flat soil samples and co-located IVBA lead/pH data from Bossburg Flat soil and beach sediment samples used in the regression analysis.

Step 2—Each of the three studies in the IVBA data inventory were deemed acceptable for use in the Upland BERA.

Step 3—All IVBA data deemed acceptable for use are suitable for use in one or more ways: (1) generation of sample-specific bioavailability estimates for soil samples included in Upland BERA EPCs, and/or (2) development of regression relationships for IVBA lead/pH and IVBA zinc/TOC data that are applied across EPCs with associated pH or TOC values, with the exception of the core samples from the 2015 Bossburg Study. These core samples are geographically redundant with the IC samples and, due to their increased depth, are less relevant to wildlife exposures.

Step 4—For use as sample-specific bioavailability estimates, IVBA data are not combined across data sets because this use is limited to sample-specific measurements represented by a single study. For IVBA regression relationships, data were considered comparable and combinable, as described in the evaluation conducted in Appendix E.

3.1.3 Bioaccumulation Data

Step 1—Studies were included in the bioaccumulation data inventory if site-specific, co-located soil and biota tissue were collected. One study, the Upper Columbia River Final Plant Tissue Study Data Summary Report (referred to herein as the "2018 Plant Tissue Study") (TAI, 2019a), was identified for the inventory. This study collected co-located soil sieved to less than 150 micrometers (µm) and plant tissue from 12 sampling areas within Tribal allotments in the Terrestrial Study Area. The objective of this study was to characterize metal concentrations in wild plants for use in the human health risk assessment.

Step 2—The 2018 Plant Tissue Study data are deemed acceptable for use, with no significant analytical issues or missing data quality documentation items.

Step 3—The study design focused on the human health risk assessment; therefore, the 2018 Plant Tissue Study (TAI, 2019a) data may lack relevance to ERA due to soil sieve sizes. However, tissues evaluated include foliage, roots, and fruit (all tissues consumed by wildlife) and the plant species collected were those consumed by people; these plant species are also often consumed by wildlife. Due to the relevance of the species and tissues sampled, the UCR plant tissue data are more relevant for risk evaluation than most field bioaccumulation data typically used in risk assessments. Because the 2018 Plant Tissue Study focused on a relatively condensed spatial distribution of sample areas within the Terrestrial Study Area relating to human exposure considerations, these data are not considered suitable for direct use because plant tissue EPCs for wildlife dietary estimates. However, these data are considered suitable for developing soil-to-plant bioaccumulation models that may be used to estimate soil sample-specific wildlife dietary uptake. Uncertainties associated with this approach are evaluated in the relevant sections of this Upland BERA (detailed discussion in Appendix C and summaries in Sections 8.2.1.1 and 9.2.1.1).

Step 4—The acceptable and suitable bioaccumulation data are from a single study (2018 Plant Tissue Study) that used consistent analytical methods; thus, the data comparability evaluation is not applicable to the bioaccumulation data.

3.2 Summary of Upland BERA Data Sets

This section provides an overview of the studies identified in the DUA as usable for the Upland BERA.

3.2.1 Soil Chemistry Data

The soil chemistry data set used in the Upland BERA comprises samples from the three studies described in the following subsections. The data set is presented in full in Appendix B.

3.2.1.1 2014 UCR Upland Soil Study

Soil sampling was conducted by TAI in September and October 2014 within three subareas (ADAs, relict floodplain deposition areas [RFDAs], and windblown sediment deposition areas [WSDAs]), using IC soil samples (a detailed description can be found in the data summary report [DSR] for the 2014 UCR Upland Soil Study [TAI, 2015]). In total, 215 IC soil samples were collected from 171 DUs (142 DUs in the ADA,[24] 16 DUs in the RFDAs, and 13 DUs in the WSDAs). At 22 of these DUs, triplicate soil samples were taken (171 DUs + 44 additional samples from triplicates = 215 total IC samples). On average, DUs were approximately 25 acres in size. This size was selected because it is considered large enough to capture a reasonable estimate for the home ranges of most small mammals. The purpose of this study was to collect soil data to evaluate upland soils in areas potentially affected by point sources (for example, aerial deposition of smelter particulates), historical fluvial deposition of sediment on relict floodplains, and redeposition of windblown sediment. The data obtained were intended for use in assessing risk to ecological and human receptors from exposure to metals in soils (the less than 149-µm fraction for health and human risk assessment and the less than 2-mm fraction for ERA). ADA DUs are evaluated in this Upland BERA, whereas DUs from ADA-140, RFDAs, and WSDAs will be evaluated in the Aquatic BERA (Section 1). ADA DU locations from the 2014 UCR Upland Soil Study are shown on Map 2-1, and summary statistics of the chemistry data are presented in Table 3-2.

Up to 30 surface soil samples (that is, 0 to 2.95 inches bgs) were composited for each incremental sample using a handheld soil probe. Prior to sieving to less than 2-mm and less than 149-µm particle fractions, an aliquot was taken from each sample for the analysis of grain size, total solids, and pH to inform contaminant bioavailability. A subset of less than 149 µm samples were analyzed by IVBA assay to evaluate metal bioavailability in soil (Appendix A). The less than 2-mm fraction was analyzed for total solids, effective cation exchange capacity (eCEC), TOC, and EPA's TAL metals (aluminum, antimony, arsenic, barium, beryllium, cadmium, calcium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, potassium, selenium, silver, sodium, thallium, vanadium, and zinc), plus molybdenum. Triplicate samples were collected to assess the precision of the sampling process in accordance with the 2014 UCR Upland Soil Study QAPP (TAI, 2014a). Out of the 142 ADA DUs, 16 were sampled in triplicate (approximately 11 percent). Soil concentrations were consistent among triplicate samples. Almost all analytes met the QAPP relative standard deviation (RSD) control limit of 35 percent, and an RSD above the control limit never occurred more than once for an analyte in any deposition area; therefore, heterogeneity was found to be acceptable. Triplicate RSDs are presented in Tables 5-7 to 5-9 of the data summary report for the 2014 UCR Upland Soil Study (TAI, 2015).

The discussion that follows provides an overview of sampling activities within the ADA, which is evaluated as part of the Terrestrial Study Area. Additional detail regarding the DU sampling is presented in the data summary report for the 2014 UCR Upland Soil Study (TAI, 2015).

24 Although originally assigned to the ADA, DU ADA-140, located in the northern part of the site, adjacent to the western shoreline of the river , has approximately 80 percent of its spatial extent below the pre-1973 maximum flood extent elevation, so will be evaluated in the Aquatic BERA.

The ADA was sampled using a distribution of DU polygons (Map 2-1). Criteria used for identifying DU locations within the ADA included the following (TAI, 2014a):

- Areas below the high-pool elevation of Lake Roosevelt and the Columbia River were excluded.
- Areas within the designated relict floodplain sampling areas were excluded.
- Areas within 50 meters of roads and railroads were excluded.
- A 500-meter no-sample buffer was established around all mine sites within the ADA that were sampled as part of the assessment detailed in the EPA Start2 Preliminary Assessments and Site Inspections Report (EPA, 2002a). A 100-meter no-sample buffer was established around all other known mine sites in the ADA.
- Locations with a slope greater than 30 degrees were excluded.
- Locations greater than 550 meters from any road were excluded to minimize travel time for field personnel reaching sampling locations.

Most DUs were approximately 25 acres and a minimum of 319 meters from any other DU boundary and did not straddle disturbed and undisturbed land. Additional detail regarding the ADA DU sampling design is presented in Section B.1.2 of the 2014 UCR Upland Soil Study QAPP (TAI, 2014a).

Samples were collected from a total of 142 DUs within the ADA during the 2014 UCR Upland Soil Study. The ADA DUs were mostly on forested hillslopes throughout the river valley and uphill from the Columbia River. DUs in the lowlands near the Columbia River or tributaries contained sparse forests and/or grassland. Several DUs were located proximate to roadways (but no closer than 50 meters) or contained visual indicators of logging or new growth forests. Few ADA DUs were adjacent to the river (for example, ADA-140) or contained visual indicators of structures.

3.2.1.2 2012 Ecology Upland Soil Study

Soil sampling was conducted by Ecology in October 2012. The two primary objectives of this study were to collect representative surface and shallow subsurface soil samples for analysis of smelter-related heavy metals and to evaluate potential spatial patterns and statistical variability of smelter-related metals concentrations in the study area. Sampling was conducted at 106 sampling locations within 13 sample subareas using a four-point composite sampling method. At 13 locations, a vertical profile (core) sample was also obtained. At 13 locations, replicate composite samples were collected. In total, 118 composite samples and 51 vertical profile samples were obtained from 106 sampling locations.

Samples consisting of four surface soil grabs (that is, 0 to 3 inches bgs) were composited using a stainless-steel spoon and represent a 0.025-acre sampling area. These sample area sizes are much smaller than most small mammal home ranges. Core samples for vertical profiling were collected from boreholes excavated with an auger, shovel, or trowel, with samples collected from depth intervals of 0 to 3 inches, 3 to 6 inches, 6 to 12 inches, and 12 to 24 inches bgs. As described in Section 3.1.1, the composite samples are suitable for use in the Upland BERA, and the geographically redundant core samples are not used. All soil samples were sieved to less than 2 mm and analyzed for EPA’s TAL metals, total mercury, TOC, pH, and total solids.

3.2.1.3 2015 Bossburg Study

In April and May 2015, TAI conducted soil sampling within six DUs on Bossburg Flat Beach or near the former cable landing area. Triplicate IC soil samples were obtained from one DU for a total of eight soil IC samples. Soil DUs ranged from 1 to 3 acres in size, with an average of 2.3 acres. The purpose of the Bossburg Flat Beach refined sediment and soil study was to further define exposure estimates and inform risk evaluations of human health and ecological receptors associated with nearshore sediment and soil adjacent to and downstream of the Young America Mine site and at Evans Campground Beach. Sediment and soil sampling was focused on a relatively small area of the site, specifically areas surrounding mining and mill operations, former cable ferry landings, and along two public beaches adjacent to the river (Bossburg Flat Beach and Evans Campground Beach). Detailed descriptions of sampling efforts for the 2015 Bossburg Study are provided in TAI (2016). Sediment results from the Bossburg Study will be evaluated in the Aquatic BERA.

For IC samples, surface soil samples (that is, 0 to 6 inches bgs) were collected from 30 predetermined locations within a DU. Incremental samples were obtained with a 4-centimeter-diameter coring tool, temporarily placed in separate plastic bags and composited into buckets. Bulk soil was examined for grain size and pH. The discrete core samples (0 to 6.9 inches, 6.9 to 12 inches, and 12 to 18 inches) collected using a coring tool are geographically redundant and not used in the Upland BERA. Soil samples sieved to less than 2 mm were analyzed for EPA's TAL metals, total mercury, TOC, and total solids.

3.2.2 IVBA Data

IVBA data were collected from the 2009–2011 Beach Sediment Study, the 2014 UCR Upland Soil Study, and the 2015 Bossburg Study. These studies met criteria for inclusion (Appendix A and Section 3.1.2) by containing co-located IVBA data for samples included in the soil chemistry data inventory (regardless of size fraction) or any sediment or soil samples with co-located IVBA lead and pH data, or co-located IVBA zinc and TOC data (regardless of sample location or size fraction). These data are used to assess the bioavailability of COPCs in soils for wildlife (Appendix E).

3.2.2.1 2009–2011 Beach Sediment Study

TAI sampled beach sediments along the Columbia River in September 2009, April 2010, and April to May 2011. Samples were collected from two beaches in 2009, three beaches in 2010, and 26 beaches in 2011, for a total of 33 beaches. Sediment was obtained from 0 to 6 inches deep from discrete locations in each beach using stainless-steel tools or an electric impact corer and randomly assigned to composite samples. Five composite samples from 12 locations (that is, 5 x 12) were obtained per beach except for two beaches in 2010 (3 x 12) and three beaches in 2011 (5 x 7). IVBA data were analyzed from one composite sample per beach. Sediment samples sieved to less than 2 mm were used to measure pH and TOC. Four size fractions (that is, less than 63 µm, 63 to 125 µm, 125 to 250 µm, 250 µm to 2 mm) were analyzed for total and IVBA concentrations of arsenic and lead. Note that these beach sediment data were employed to develop a broader pH range to support a more robust IVBA-pH regression (Appendix E, Figure 6).

3.2.2.2 2014 UCR Upland Soil Study

The 2014 UCR Upland Soil Study is broadly summarized in Section 3.2.1; the following discussion is specific to IVBA data collected as part of this study. A subset of 25 DUs were sampled for IVBA data, including 23 DUs in the ADA and 2 DUs in the RFDA. Unsieved soil was examined for pH, and soil sieved to less than 2 mm was analyzed for TOC. Total metal and IVBA concentrations of TAL metals and molybdenum were quantified from soil sieved to less than 149 µm.

3.2.2.3 2015 Bossburg Study

The 2015 Bossburg Study is presented in Section 3.2.3; details specific to IVBA data collection are discussed here. TAI sampled sediments from 10 DUs on or adjacent to the Bossburg Flat Beach and Evans Campground Beach from April to May 2015 using IC sampling in addition to the soil DU sampling discussed previously. Four sediment DUs were sampled in triplicate for a total of 18 IC sediment samples. The IC sampling design (for example, depth, compositing) was the same for soils and sediments. Sediment pH and TOC were measured from the same size fractions as IC soil samples. IC sediment samples sieved to less than 250 µm and IC soil samples sieved to less than 150 µm were analyzed for total and IVBA concentrations of arsenic and lead.

3.2.3 Bioaccumulation Data

Site-specific bioaccumulation data are available from the 2018 Plant Tissue Study. In April, May, June, and August of 2018, TAI collected plant tissue and co-located soil samples from 12 sampling areas within Tribal allotments in the Terrestrial Study Area (TAI, 2019a). Fourteen plant species were sampled: black tree lichen, camas, kinnikinnick, lomatium, spring beauty (Indian potato), willow, huckleberry, wild rose, chokecherry, hazelnut, ponderosa pine, sarvisberry, tule, and wild mint. The co-located soil samples were targeted for each plant species and tissue type sample; co-located soil samples were collected next to small plants or below the crown of larger bushes and trees. For individual plant samples, one co-located soil sample was collected. For composite plant samples, a co-located soil sample was collected for each individual plant sampled, and soil was composited in the field proportionally to the weight of the plant tissue from each plant in the composite.[25] Plants included in the composite sample were sampled within a 3-meter radius of the original plant when possible. Soil samples were collected from 0 to 3 inches bgs using a decontaminated auger, coring, or spade tool, and then air-dried and sieved to less than 150 µm at the analytical laboratory, as specified in the QAPP (TAI, 2018b). This particle size fraction is intended to represent the fraction expected to adhere to human skin via dermal contact (TAI, 2019a).

A total of 174 plant tissue and co-located soil samples were analyzed for TAL metals (except calcium, magnesium, potassium, and sodium) and total solids. Total mercury was analyzed in select plant tissue species (kinnikinnick leaves, wild rose leaves and stems, wild mint leaves, willow branches, and tule) with a total of 63 plant tissue and co-located soil samples. Willow samples were collected from relict floodplains (one sampling area on Deadman's Eddy Island and one on Barnaby Island).

3.3 Environmental Chemistry Data Management and Treatment

Data management and data treatment of the Upland BERA environmental chemistry data set are handled as described within the Final Data Management Plan (TAI, 2019c) and as summarized in the following subsections.

3.3.1 Sample Types and Usage

Several different types of soil chemistry samples were collected in the three studies comprising the Upland BERA soil chemistry data set. These types and their use is as follows:

- **Split samples.** Split samples are two or more parts of a homogeneous sample divided for analysis of the same parameters by different laboratories. Split samples are used to check analytical techniques. The parent sample is used in the analyses in this Upland BERA.
- **Laboratory replicate samples.** Laboratory replicates are used to assess analytical error. The parent sample results are used in this Upland BERA.

[25] For black tree lichen, a composite plant sample was collected from trees within 20-meter-diameter plots. The co-located soil sample was taken from the center of the 20-meter plots.

- **IC samples (TAI 2015, 2016).** The 2014 UCR Upland Soil Study and the 2015 Bossburg Study data sets include replicate samples collected from a subset of DUs. The term "replicate" is used to refer to the second and third IC samples collected from a DU where triplicate IC samples were collected. IC field replicates require completely separate incremental locations within a DU. Primary, duplicate, and triplicate IC samples should be retained as individual sample results rather than being reduced to a single sample result, according to TAI (2019c). Therefore, risk calculations are conducted for each of the triplicate results, and the highest (most conservative) result is presented in the text, tables, figures, and maps of the Upland BERA. Where applicable, results from replicate samples are also discussed to evaluate the variability in concentrations from a given DU, and its implications on risk conclusions.
- **Composite samples (Ecology, 2013).** The 2012 Ecology Upland Soil Study also included replicates in its sample design, although one replicate was collected from a subset of sample locations rather than two replicates. The same approach used for replicate handling for IC samples is also applied to 2012 Ecology Upland Soil Study data.

3.3.2 Qualified and Censored Data Handling

Data qualifiers relevant to the Upland BERA are described as follows. Censored data are handled using different approaches depending on the calculation being applied in the Upland BERA. Specific handling approaches are described as part of the calculations and are presented here and in later sections:

- **Nondetected values.** Nondetected values are identified as "U," "U*," "UJ," "JU*". The U flag indicates the analyte was not detected below the method reporting limit (MRL). The U* flag indicates the analyte should be considered not detected because it was detected in an associated blank at a similar level. The UJ or JU* flag indicates the analyte was not detected below the method detection limit (MDL). The studies included in this Upland BERA reported nondetected values either at the MRL or MDL as provided by each study and reported in the UCR project database. Values reported to the MRL are more conservative than those reported to the MDL when used as concentration data in calculations. IVBA data (that is, IVBA metal concentration, total metal concentration, or percent IVBA concentration) with a U, U*, UJ, or JU* flag were considered invalid and not retained for IVBA analyses. Similarly, data flagged with a U, U*, or UJ within the bioaccumulation data set were not retained for bioaccumulation modeling; affected soil-plant tissue data pairs were excluded from the analysis. Unless otherwise noted in later sections for specific applications, censored data (data flagged with U, U*, UJ, or JU*) are handled using the full MRL or MDL for calculations.
- **Estimated or biased data.** Data qualified as "J" (estimated), "J+" (biased high), or "J-" (biased low) are acceptable for use in risk assessments. No modifications are to be made to these data to "correct" them. Data flagged with these qualifiers may require careful consideration when interpreting risk evaluations and results.
- **Rejected data.** Data qualified as "R" or "UR" for rejected data were data that failed quality assurance (QA) checks. These data are excluded from the Upland BERA. All IVBA data (that is, IVBA metal concentration, total metal concentration, or percent IVBA concentration) for a sample were excluded when either the IVBA or total metal concentration were flagged as R or UR.
- **Blank contamination.** Data qualified as "B" for blank contamination were treated as nondetected results (for example, the same as data flagged as U).

3.3.3 Significant Digits and Rounding for Analytical Data Set

To limit the amount of uncertainty in the data analysis, results from calculations are rounded to the number of significant digits in the least exact factor for multiplication and division or the quantity with the least number of significant decimal places for addition and subtraction. When rounding, the final digit is rounded up 1 if the value to its right is 5 or greater; otherwise, the last significant digit is retained without changing its value. Rounding occurs as the last step of calculations, not following each stage of the calculation. Where appropriate, unrounded calculation results are shown to ensure that results are transparent, reproducible, and consistent. For HQ calculations, an additional rounding step is conducted for final reporting purposes (Section 5.1.1).

4. Approach to Exposure and Effects Assessments and Risk Analysis

This section provides a description of the methods used to characterize exposure and effects and analyze risk to Terrestrial Study Area EAEs posed by exposure of ecological receptors to COPCs in soil. The approaches and methods used were developed in accordance with the BERA Work Plan (TAI, 2011) and the expanded problem formulation plan (TAI, 2012). These approaches evaluate data and information available for the Terrestrial Study Area of the UCR site. The approaches described are specific to the Terrestrial Study Area of the site as defined in Section 1 of this report; methods and approaches to be used in the aquatic portions of the site will be discussed in the Aquatic BERA.

4.1 Exposure Assessment Approach

The exposure assessment evaluates the frequency and magnitude of the co-occurrence of COPCs and ecological receptors (EPA, 1998). In application, the exposure assessment describes the exposure media, exposure units, exposure pathways, and associated exposure point concentrations (EPCs) relevant to each EAE within a given receptor group. The approaches for conducting the exposure assessment are consistent for all EAEs and receptor groups evaluated in the Upland BERA and are presented in the following subsections.

4.1.1 Exposure Media

Three soil chemistry data sets were used to develop EPCs in the Upland BERA: the 2014 UCR Upland Soil Study (TAI, 2015), the 2012 Ecology Upland Soil Study (Ecology, 2013), and the 2015 Bossburg Study (TAI,2016) (Section 3.2 and Map 2-1). Each of these data sets was identified as acceptable in the DUA; thus, each data set is sufficiently reliable for the Upland BERA, as described in Section 3 of this report.

4.1.2 Exposure Units

Each sample location from each of the three studies described in Section 3.2.1 was considered an exposure unit. A discussion of the uncertainty in risk estimates for individual receptors related to exposure unit size is presented in the subsequent risk characterization sections.

4.1.3 Exposure Point Concentrations

One COPC concentration at each sample location from each data set is used as the EPC in this Upland BERA. All data are presented in Appendix B, Table B-1, and the data are summarized in Table 3-1. As described in Section 4.5.3, alternative EPCs are considered in the uncertainty analyses.

4.1.3.1 Measures of Exposure

Two general measures of exposure for terrestrial receptors are discussed in the BERA Work Plan (TAI, 2011) and used in the Upland BERA as follows:

- **Direct contact with media for terrestrial plants and for soil invertebrates**. Includes dermal (soil invertebrates) and root contact.
- **Dietary exposure for wildlife**. Includes consumption of metals in biota (plant and animal tissue) and abiotic media (soil) through diet and incidental ingestion.

Direct contact exposures are evaluated using measured COPC concentrations in the applicable media. Dietary exposures are evaluated on a daily dose basis for wildlife (that is, body-weight-normalized daily amount of COPC ingested, expressed as milligrams of metal per kilogram of body weight per day [mg/kg bw/day]).

Dose-based dietary exposures for wildlife are calculated using Equation 4-1:

Equation 4-1: Dose-Based Dietary Exposure Model

$$Daily\ Dose = \left(\left(\left(FIR \times \sum\left(f_1 \times C_{food\ 1} \times RBA_{food\ 1} + f_n \times C_{food\ n} \times RBA_{food\ n}\right)\right) + \left(SIR \times C_{soil} \times RBA_{soil}\right)\right) \times AUF\right)/BW$$

Where:

Daily Dose = COPCs ingested per day via food and soil (mg/kg bw/day)
FIR = food ingestion rate (kilogram(s) of food [dry weight] per day [kg food dw/day])
$f_{1 \ldots n}$ = fraction of biota items 1 through n in the overall diet (unitless), based on mass, the sum of which does not exceed 1
$C_{food\ 1 \ldots n}$ = concentration of the metal in the biota items 1 through n (mg/kg-dry weight [dw])
$RBA_{food\ 1 \ldots n}$ = Bioavailable fraction absorbed from ingested biota items 1 through n (unitless)
SIR = Soil ingestion rate (kg soil dw/day)
C_{soil} = Concentration in soil (mg/kg-dw)
RBA_{soil} = Bioavailable fraction absorbed from ingested soil (unitless)
AUF = Area use factor (unitless); fraction of time that a receptor group spends foraging in a given exposure unit, relative to the entire home range.
BW = body weight (kg)

Daily dietary dose modeling relies on WEFs as defined in Equation 4-1. Concentrations in food are estimated from soil EPCs using bioaccumulation models. The bioavailability of metals in soil and food items controls what fraction of the total metal concentration in consumed soil and food is available to cause toxicity. These parameters are described further in the next subsections.

4.1.4 Wildlife Exposure Factors

WEFs used in the dose-based dietary exposure model (for example, food ingestion rates, body weights shown in Equation 4-1) are evaluated for each receptor group (that is, using representative species) based on data compiled from literature sources. WEFs and dietary composition fractions are presented in Table 4-1 and Table 4-2.

Area use is assumed to be 100 percent for all receptors by applying an AUF value of 1 (Refer to Equation 4-1 for how the AUF is incorporated into dose-based dietary exposure). This assumes that a receptor is exposed entirely within a given DU (2014 UCR Upland Soil Study and 2015 Bossburg Study) or sample location (2012 Ecology Upland Soil Study) at all times. This creates uncertainty in COPC exposure to receptors that forage over areas larger than an exposure unit (or when exposure units are small in size, as is the case for the Ecology and Bossburg data sets), populations that encompass multiple locations, and/or receptors that migrate so that they would be present in a given location only a portion of the year (for example, seasonally). Aggregating adjacent sample locations addresses much of this uncertainty. Uncertainty associated with this approach is discussed in the uncertainty section of the risk characterization sections.

4.1.5 Bioavailability of Metals

Consistent with Principle 4 of EPA (2007a) (Section 1.3), understanding the bioavailability of metals in the Terrestrial Study Area is critical for increased accuracy in the risk characterization. A metal's bioavailability to an organism varies and is dependent upon the metal form (species), complexation with ligands, and organism characteristics (for example, gut pH). For metal mixtures, the complexity is magnified by individual metal bioavailabilities, composition of the mixture, proportion of individual metals, and concentration of the mixture (EPA, 2007a). Thus, comparability across metal mixtures is difficult to predict, and bioavailability adjustments in this Upland BERA are developed for individual metals only.

For terrestrial plants and soil invertebrates, bioavailability adjustments are performed in the effects assessment, as part of the benchmark development (Section 4.2.1). For wildlife, bioavailability adjustments are performed within the exposure assessment, as described in more detail in the paragraphs that follow.

Relative bioavailability (RBA) values for soil are included in the calculations of wildlife dietary dose (Equation 4-1). The RBAs account for the difference between the absolute bioavailability of COPCs in site soil or food and the bioavailability of the COPC in the TRV study to which the dietary dose is being compared. The calculated RBAs are sample-specific, COPC-specific, and at times, inferred, as discussed following.

It is conservative to assume complete bioavailability because only a fraction of COPC concentrations in food or soil are available for absorption in an organism's gut. IVBA data for Upland soil were collected as part of several soil studies used to support the human health risk assessment (EPA, 2021). IVBA data deemed acceptable and suitable for use in the ERA (Section 3.1.2) were used to estimate RBA values for wildlife. IVBA analyses simulate the conditions in a human child's gut and provide an estimate of the fraction available for absorption from the gastrointestinal tract (EPA, 2017a). The percent bioaccessible is calculated using two different extractions from the same soil sample; the measured COPC concentration in the IVBA extraction divided by the measured COPC concentration in the total metals extraction yields the percent bioaccessible within that sample, on a per-COPC basis (Appendix E). IVBA extractions were performed on a subset of soil samples within the Upland BERA data set. Thus, the measured percent bioaccessible was used where available, and percent bioaccessible was estimated for the remaining samples using either regression equations using measured pH (lead) or TOC (zinc) as the predictive variable, or mean measured percent bioaccessible (all COPCs except lead and zinc) for applicable sample areas. Further details on this approach are provided in Attachment E3 of Appendix E. An additional refinement to RBA values was conducted for lead, for which in vivo validation studies in wildlife species are available. These in vivo validation studies were used to estimate the relationship between the percent bioaccessibility of lead derived from IVBA analysis and the RBA of lead in soil to wildlife. For all other COPCs, RBA was assumed equivalent to the measured or estimated percent bioaccessible as calculated from IVBA data on a per-COPC basis. Sample-specific and COPC-specific RBA values used for wildlife dose modeling are reported in Attachment E3 of Appendix E. These RBA values are relevant to ingested soil and are thus applied to the estimated soil dose in the wildlife dose modeling; these RBA values are not applied to estimated doses from food.

The applicability of using IVBA data collected for the human health risk assessment to support the Upland BERA's wildlife dose modeling[26] was evaluated. Three technical issues associated with applying IVBA data to wildlife exposure estimation were assessed: (1) considering the influence of digestive system characteristics, which differ across species; (2) evaluating validation studies comparing IVBA and

[26] Birds and small mammals, excluding ruminants. Ruminants are not monogastric; thus, IVBA data are not applicable.

RBA data in animals; and (3) verifying the applicability of IVBA data to the specific TRVs selected for use in the Upland BERA. A detailed discussion of the use of bioavailability in the Upland BERA exposure assessment, and applicability of IVBA to wildlife, is provided in Appendix E. Uncertainties associated with these assumptions are discussed in the uncertainty evaluations for birds and mammals.

Because of a lack of site-specific data, COPCs in food (prey and plant tissue) are conservatively assumed to have an RBA value of 1 in the Upland BERA (Appendix E).[27] This likely overestimates the bioavailability of COPCs in plant and animal tissue consumed by wildlife. Uncertainty associated with this approach is discussed in the uncertainty evaluations for birds and mammals.

4.1.6 Bioaccumulation Models

Bioaccumulation models are used to estimate uptake of COPCs from soil into biota (for example, terrestrial plants, invertebrates of terrestrial origin [earthworms, terrestrial arthropods, and flying insects], and small mammals) for receptors that are evaluated for the dietary exposure pathway. For the Upland BERA, site-specific bioaccumulation models were developed where data were available. As described in Sections 3.1.3 and 3.2.3, site-specific co-located soil and plant tissue data are available from the 2018 Plant Tissue Study (TAI, 2019a). For all other dietary items (earthworms, terrestrial arthropods, flying insects, and small mammals), bioaccumulation models were sourced from the literature and thus are not site specific. Literature containing bioaccumulation models commonly used in Superfund site risk assessments are the primary sources for the literature-based bioaccumulation models used in the Upland BERA. An additional literature search was conducted for COPC-specific soil-to-biota pairs without an available model from said sources; in particular, data were lacking for molybdenum. Thus, sources were ranked by considering site specificity, well respected- sources commonly used in ERAs, availability of models, and availability of raw data that could be used to develop models. Bioaccumulation model source ranking for the Upland BERA is as follows (with rank 1 as the most preferred and rank 4 the least):

1) Site-specific data from the 2018 Plant Tissue Study. The DUA identified these data as acceptable and suitable for use for bioaccumulation modeling from soil to plant tissue (Section 3.1.3).
2) Models from Oak Ridge National Laboratory (Sample et al., 1998a, 1998b). These models are reliable (well documented, appropriate methodology, robust data sets for many metals) and relevant (broad applicability to soil across Europe and North America).
3) Raw data from U.S. Army Center for Health Promotion and Preventative Medicine (USACHPPM) used to develop models (USACHPPM, 2204) (Appendix C). This database is reliable (well documented, appropriate methodology) and relevant (broad applicability to soil across Europe and North America).
4) Models from peer-reviewed literature (for example, Hargreaves et al., 2011). Depending on the specific source, reliability and relevance of the models may vary.

Model source availability on a per-COPC, per-biota type basis is presented in Table 4-3. Models and/or data were selected in order from the ranked sources until a model was available for each of the required COPC-biota type pairings for the Upland BERA.

Additionally, model types were ranked according to the criteria outlined in EPA's ecological soil screening level (Eco-SSL) guidance (EPA, 2007b); regression models were prioritized over median bioaccumulation factors (BAFs) when the regression model was significant (slope p less than 0.05) and met minimum correlation requirements (coefficient of determination (R^2) greater than or equal to 0.2).

[27] An RBA of 1 assumes metal bioavailability in the medium is the same as the metal's bioavailability in the medium used in the TRV study.

The regression models are better at accounting for variable uptake over differing concentrations in soil than BAFs, which are a static ratio. Model selection for each COPC was done first by ranking the sources (rank 1 to 4) as previously noted, then by ranking the model type (that is, regression or BAF). For example, a BAF from model source 1 would be selected over a regression from model source 2.

The USACHPPM (2004) source (source 3) reports a raw database and median BAFs for terrestrial arthropods and for terrestrial plants but did not conduct regression analyses. In addition, the reported BAFs were aggregated by either invertebrate taxonomic order or plant part, neither of which is a useful aggregation for use in the Upland BERA. Therefore, modeling was performed on the raw data provided in USACHPPM (2004), which is presented in full in Appendix C.

Bioaccumulation models used in the Upland BERA are presented in Table 4-4. Additional discussion around model selection for specific biota is presented in the following subsections.

4.1.6.1 Terrestrial Plants

Bioaccumulation models for soil to terrestrial plants are used in dietary pathway evaluations for herbivores and omnivores. The co-located soil and plant tissue data from the 2018 Plant Tissue Study (TAI, 2019a) were used to develop site-specific bioaccumulation models, as is presented in full in Appendix C. Briefly, models were developed as follows:

- **Data handling.** Soil-plant data pairs where both samples had detected concentrations were included for analysis. Willow inner bark samples were excluded because their sampling location (relict floodplains) was not relevant to the Terrestrial Study Area (Section 3.2.3). Black tree lichen samples were also excluded from analysis because this species lacks direct connection with soil, and is an unlikely source of food for most Upland BERA receptors (Tables 2-10 and 2-11). The soil samples in the 2018 Plant Tissue Study were sieved to less than 150 µm; thus, to increase relevance to plant exposures and EPCs, a less than 150 µm to less than 2 mm correction was applied, as described in full in Appendix C. Finally, plant parts were categorized as either being above ground (fruits, seeds, nuts, stems, and leaves) or below ground (roots, bulbs, and corms). Soil and plant concentrations were log transformed before regression modeling.
- **Data analysis.** Linear regression modeling was attempted for situations with more than 10 data points. If the regression model met the acceptance criteria in the previous paragraph, it was selected for use. Otherwise, the median BAF was selected for use.

There are several uncertainties associated with this approach, including the relevance of plant tissue sampling areas to locations across the entire Terrestrial Study Area, the relevance of sampled plant parts to receptor diets across seasons, the use of sieve size correction factors, and poor regression model performance. These uncertainties are discussed in detail in Appendix C and summarized in the relevant wildlife uncertainty discussions.

4.1.6.2 Earthworm Prey

Bioaccumulation models for soil to earthworms are taken from Sample et al. (1998b). Median BAFs for aluminum, iron, molybdenum, and vanadium are obtained from the validation data set summary statistics provided in an appendix to that report. The sample size for these four metals is lower (ranging from 4 to 20) than for other metals (up to 245) in the report. For thallium, no bioaccumulation data were reported in Sample et al. (1998b). A literature search was conducted for thallium bioaccumulation into earthworm tissue, but no appropriate sources were identified. Thus, the default BAF of 1 is used for thallium.

4.1.6.3 Terrestrial Arthropods and Flying Insect Prey

Terrestrial arthropods, including spiders, centipedes, millipedes, and insects, are an important food item for many receptors. Flying insects, a subset of terrestrial arthropods, are the dominant food for aerial insectivorous receptors (bat and swallow). Bioaccumulation of metals to flying insects may be different from that of flightless arthropods given the differences in exposure to soil and life history characteristics. Therefore, separate models were generated for terrestrial arthropods (including both flightless and flying arthropods) and flying insects (only adult flying insects). Bioaccumulation models were developed for both terrestrial arthropods and flying insects from the USACHPPM (2004) data set (Appendix C).

The USACHPPM (2004) data set did not have available data for molybdenum; thus, a literature search was conducted for studies that evaluated the uptake of molybdenum from soil to terrestrial arthropods and/or flying insects. One study was identified (Hargreaves et al., 2011). This source has sufficient reliability (peer-reviewed, N = 20, appropriate methodology) and sufficient relevance (field collection of soil invertebrates [spiders, beetles, flies, wasps, amphipods, worms, true bugs] in pitfall traps in the Canadian Arctic) for use in the Upland BERA. A median BAF was not reported from this source; a single mean BAF for all arthropods (including flying insects) was reported. Therefore, this mean BAF was selected for use for both terrestrial arthropods and flying insects for molybdenum.

4.1.6.4 Small Mammals and Ungulate Prey

Bioaccumulation models for small mammals were selected from Sample et al. (1998a), which used whole-body concentration data of shrews, moles, mice, voles, rats, and squirrels. This source reports models for different trophic levels of small mammals; models were selected consistent with those chosen for EPA's Eco-SSLs (EPA, 2007b), which are either herbivore or general trophic group models. Sample et al. (1998a) does not report data for molybdenum; therefore, a literature search was conducted for studies that evaluated the uptake of molybdenum from soil by small mammals. No appropriate studies on the uptake of molybdenum into whole-body of small mammals were identified. Due to the lack of available data, a default BAF of 1 is used for molybdenum.

A literature search was also conducted for studies that evaluated the uptake of metals to whole-body ungulates for predatory receptors (for example, wolves) expected to consume ungulates such as deer. No studies on whole-body ungulates were identified; the available studies evaluated uptake into muscle tissue (for example, Baes et al., 1984). The available ungulate muscle models yield lower tissue concentrations than the above models for whole-body small mammals for seven (all but mercury and zinc) out of the nine COPCs (aluminum, cadmium, chromium, copper, lead, mercury, selenium, vanadium, and zinc) with available data for comparison (refer to discussion in Section 9.2.1.1); thus, whole-body small mammal models are more conservative for most metals. Given the lack of relevant data for whole-body ungulates, the whole-body models for small mammals were selected to estimate COPC concentrations in ungulate prey. Uncertainty associated with this approach is discussed in the uncertainty section for the relevant POEs (Section 9.2.1.1).

4.2 Effects Assessment Approach

The effects assessment identifies the COPC concentration thresholds (referred to as benchmarks[28] and TRVs[29]) used to evaluate potential toxic effects on ecological receptors. Benchmarks and TRVs represent metal concentrations below which adverse effects on exposed organisms are unlikely to occur and/or are low in severity (for example, a 20 percent reduction as compared to control groups) and above which such

[28] For the purposes of this Upland BERA, the term "benchmark" is used to describe any combination of toxicity values for a measurement endpoint.
[29] TRVs are a form of benchmark specific to ingested doses and/or tissue concentrations.

effects may occur at some level of severity (approximately 20 to 100 percent reduction as compared to control groups) and frequency. “Benchmark” is also used in a general context to describe any combination of toxicity values used to estimate potential adverse effects when not exclusively describing TRVs. These values are derived from toxicity data reported in the scientific literature.

4.2.1 Soil Benchmarks for Plants and Invertebrates

Of the TAL metals list plus molybdenum, 20 metals were identified as either COPCs or COIs for plants and/or invertebrates in the COPC refinement (TAI, 2020b). Metals that did not screen in as COPCs or were not retained as COIs in the COPC refinement document were eliminated from further evaluation in this Upland BERA. The COPCs and COIs retained for evaluation in the Upland BERA for plants and invertebrates are listed in Table 2-7; for simplicity, COIs that can be evaluated quantitatively in the Upland BERA (green highlight in Table 2-7) are referred to as COPCs. Two benchmark types are used to evaluate the potential for adverse effects (toxicity) to plants and invertebrates: soil screening levels (SSLs) (including Eco-SSLs) and bioavailability-adjusted benchmarks (BABs), as shown in Table 4-5. These two different benchmark types formulate the two different POEs that comprise the LOE for soil chemistry compared to benchmarks. An overview of the two benchmark types is provided in Sections 4.2.1.1 and 4.2.1.2. Table 4-6 presents a review of reliability and relevance qualities for the plant and invertebrate soil benchmarks; this information is used to support the weight of evidence (WOE) in the plant and invertebrate risk characterization. The derivation approach for soil benchmarks for plants and invertebrates is presented in full in Appendix D.

4.2.1.1 Soil Screening-Level Benchmarks (Eco-SSLs and SSLs)

When available, EPA’s Eco-SSLs are used for the SSL benchmark type for evaluating the potential for adverse effects from COPCs to plant and invertebrate receptors. COPCs for which a plant and/or invertebrate Eco-SSL are not available are evaluated using SSLs. The SSLs were developed for the site (Appendix D) following similar methods used by EPA to derive the Eco-SSLs (EPA, 2005b). Plant and invertebrate Eco-SSL/SSL benchmarks are presented in Table 4-5 and documented in full in Appendix D. For aluminum and iron, potential toxicity is dependent on soil pH rather than metal concentration in soil (EPA, 2005b); thus, pH thresholds for these two COPCs are used as per their respective Eco-SSLs.

Because SSLs were derived using the same methodology used by EPA to derive Eco-SSLs, SSLs have similar advantages and shortcomings as Eco-SSLs. Advantages include routine use in ERAs for Superfund and state-regulated contaminated sites. Eco-SSLs were derived using a robust, peer-reviewed process that produced benchmarks considered protective of a broad range of plant and invertebrate species. However, as screening-level benchmarks, HQs $\geq$ 1 are intended to identify locations with COPC concentrations that warrant additional evaluation. Exposure unit concentrations above an Eco-SSL do not provide a definitive indication of adverse effects or unacceptable risk. As noted in the Eco-SSL guidance,

> *These values [Eco-SSLs] can be used to identify those COPCs in soils requiring further evaluation in a BERA. The Eco-SSLs should be used during Step 2 of the Superfund ERA process, the screening-level risk calculation. The Eco-SSLs are not designed to be used as cleanup levels, and EPA emphasizes that it is inappropriate to adopt or modify these Eco-SSLs as cleanup standards. (EPA 2005b)*

Eco-SSLs were used in the COPC refinement (TAI, 2020b) consistent with the SSL guidance. Retaining these benchmarks for use in the Upland BERA is a conservative means of assessing the potential for adverse effects and identifying sample locations to be evaluated further. Uncertainties associated with this conservatism are considered in the plant and invertebrate uncertainty evaluations.

4.2.1.2 Bioavailability-Adjusted Benchmarks

BABs provide an approach that accounts for bioavailability in the prediction of potential metals toxicity to plant and soil invertebrate receptors. Bioavailability-adjusted toxicity effects levels were calculated for six COPCs (cobalt, copper, lead, molybdenum, nickel, and zinc) where data were available. The Threshold Calculator (ARCHE, 2020) was used to generate the sample-specific BAB values for COPCs using sample-specific measured bioavailability parameters (organic carbon [OC] content, pH, clay content, eCEC, and for zinc, background metals concentrations). For those plant and invertebrate species tested over a wide range of soil bioavailability conditions, empirical regression relationships between metal toxicity and one or more soil parameters were developed by the Threshold Calculator (ARCHE, 2020). A detailed description of BAB derivations can be found in Appendix D. Each calculated BAB for each sample, COPC, and receptor group is presented in Appendix F. BABs and associated values are rounded to three significant digits, consistent with the use of three significant digits in the input parameters for soil COPC concentration, pH, TOC, percent clay, and eCEC.

One advantage of using BABs is that they incorporate measured soil properties that have an important influence on the fraction of metal in soil that can be taken up by a receptor, consistent with Principle 4 of EPA (2007a) (Section 1.3). BABs from the Threshold Calculator also have regulatory acceptance and are derived in the European Union Registration, Evaluation, Authorization, and Restriction of Chemicals dossiers (Regulation EC No. 1907/2006). A third important advantage of BABs stems from their use of species sensitivity distributions (SSDs) and concentration-response information. With each sample-specific benchmark calculation, the potentially affected fraction (PAF) is also provided for that sample's specific COPC concentration and bioavailability condition. The PAF estimates the proportion of species that are likely to be affected (at a 20 percent effect level in the current assessment) under the sample-specific conditions of COPC concentration and bioavailability. The BABs are calculated as the median hazardous concentration for 5 percent of the species (HC5) derived as the 5th percentile of the SSD. This approach is analogous to EPA's approach to calculating hardness-based ambient water quality criteria (AWQC), which entails the application of a hardness-based bioavailability model to adjust toxicity data in the SSD to a water hardness of interest; the 5th percentile of the SSD is then calculated to derive the criterion. For COPCs for which bioavailability benchmarks are available, sample-specific PAF calculations are used to inform the frequency of adverse effects (20 percent effect level) on plant and invertebrate species within the terrestrial plant and invertebrate community EAEs. Uncertainties associated with use of BABs are discussed in Section 4.5.

One shortcoming of BABs is that they are not available for all COPCs evaluated in this Upland BERA (Table 4-5). Another is the need for measured soil physiochemical properties beyond just COPC concentrations, including OC content, pH, clay content, eCEC, and for zinc, the background zinc concentration. The 2012 Ecology Upland Soil Study sample data do not include eCEC or clay content; thus, BABs cannot be calculated for those sample locations using only the study-specific data. Therefore, the minimum percent clay and eCEC from the 2014 UCR Upland Study DUs overlapping each 2012 Ecology subarea were used to calculate BABs for the 2012 Ecology samples in the respective subareas. For these analyses in locations where duplicates had been collected, the minimum pH and TOC and maximum COPC concentration was used. These analyses are described briefly in Section 6.2 and detailed in Appendix F.

4.2.2 Wildlife Toxicity Reference Values

The TRVs used in the Upland BERA are derived from literature sources using approaches developed by TAI, which have been reviewed and approved by EPA (TAI, 2019d). TRVs are expressed as daily COPC intake rates by body weight of the organism. Three separate TRVs were derived for each COPC when data were available, for growth, reproduction, and survival endpoints measured at the organism level. Wildlife TRVs for four COPCs (cadmium, copper, lead, zinc) are from the EPA-approved wildlife TRV technical memorandum (TAI, 2019d). The other wildlife TRVs used in this Upland BERA are developed in accordance with the approach presented in TAI (2019d). A comprehensive description of the approach and derivation of all TRVs used in this Upland BERA is presented in Appendix E. Dietary dose TRVs were established using a paradigm targeting a 20 percent effect level as compared to controls, where supported by the toxicity data. When possible, ED20s (effective doses with a 20 percent reduction in the response relative to the control) calculated from dose-response models for three separate endpoints (growth, reproduction, and survival) were selected as the TRVs when possible. As discussed in Appendix E, the ED20 was selected as the appropriate effective dose (ED) for the Upland BERA based on precedence and EPA's guidance for a variety of receptors, including wildlife. If dose-response models were not available or appropriate for use, TRVs were identified based on lowest observed adverse effect level (LOAEL) with ≥ 20 percent reduction in the response relative to the control. If the ED20 or LOAEL ≥ 20 derived for a particular metal/receptor was less than the TRV derived by EPA for calculating the Eco-SSL (often set at a no observed adverse effect level [NOAEL]), then EPA's Eco-SSL TRV was used (for some COPCs, toxicity data were not sufficient for deriving a TRV). TRVs were rounded to two significant digits, reflective of the number of significant digits obtained from the literature sources. All dietary TRVs are presented in Table 4-7.

In addition to the use of static TRVs, dose-response models were incorporated into the risk characterization for those COPCs with HQs ≥ 1, as requested by EPA. Where dose-response relationships can be developed from the available toxicity data, EDs (EDx; the effective doses with an x percent reduction in the response relative to the control based on the dose-response model) were also estimated. The EDx values provide an estimate of the possible magnitude of the adverse effect and better connect the possible effect at the level of the organism to potential population-level impacts, which cannot be evaluated by simply reviewing the magnitude of an HQ. Dose-response models were sourced from the TRV derivation process (Appendix E, Annex D). TRVs derived from a LOAEL ≥ 20 or Eco-SSL do not have associated dose-response models. In the specific case of bird reproduction and lead, dose-response information could not be developed from the TRV study but has been previously identified in the literature (Sample et al., 2019, as recommended by EPA). This study by Sample et al. (2019) identified high variability that confounded results for a particular endpoint, Japanese quail egg production, and calculated dose-response curves using more reliable data for chicken egg production. Thus, the ED20 and ED50 as reported by Sample et al. (2019) are used in the Upland BERA as the dose-response information for lead and bird reproduction. Uncertainty associated with these toxicity data is discussed in the bird uncertainty section.

Parameters for dose-response models, along with calculated ED20, ED50, and ED80 values, are presented in Table 4-8. While the calculated ED20, ED50, and ED80 values are rounded to two significant digits to align with the underlying dose and effect data, consistent with the modeling software, the dose-response model parameters output from the model are not rounded.

Uncertainties associated with development of, or the absence of, a TRV (lack of available toxicity data) and/or dose-response models are assessed in the uncertainty evaluations for the relevant receptor group. To support this uncertainty evaluation and the WOE, a review of reliability and relevance qualities is provided for birds and mammals in Tables 4-9 and 4-10, respectively.

4.3 Risk Estimation Approaches for Each Receptor Group

Building on the generalized approach described in Sections 4.1 and 4.2, the analysis approaches specific to each receptor group are described in Sections 4.3.1 through 4.3.3.

4.3.1 Risk Estimation for Terrestrial Plants

The exposure assessment and effects assessment specific to terrestrial plants is presented as follows:

- **Exposure Assessment.** Plants are exposed to metals through direct contact with soil (Section 2.6.5). Exposure estimates are evaluated for each exposure unit (each 2014 UCR Upland Soil Study DU, each 2012 Ecology Upland Soil Study sample location, and each Bossburg location; Sections 4.1.2 and 4.1.3).
- **Effects Assessment.** Soil concentrations are compared to two different types of plant toxicity benchmarks, SSL benchmarks (Eco-SSLs, when available, and SSLs developed for metals without Eco-SSLs) (TAI 2020a), and BABs derived from the Threshold Calculator for metals in soil (ARCHE, 2020), where available. These soil benchmarks are presented in Table 4-5, with details of benchmark derivation provided in Appendix D.
- **Risk Estimation.** The analysis approach is the calculation of HQs (Section 5.1.1) and the bioavailability-adjusted, concentration-response-dependent PAFs (Section 4.2.1.2).
- For samples with triplicate IC soil samples (2014 UCR Upland Soil Study DU and 2015 Bossburg Study) or duplicate soil samples (2012 Ecology Upland Soil Study) the HQ was calculated for all replicates, and the greatest resulting HQ was retained and used in the risk estimation. This approach was used for calculating HQs for both the SSL and BAB benchmarks.

4.3.2 Risk Estimation for Soil Invertebrates

The analysis for soil invertebrates includes an exposure assessment and effects assessment consistent with the approaches described in Section 0 and 4.2, and summarized as follows:

- **Exposure Assessment.** Similar to plants, soil-dwelling invertebrates are exposed to metals through direct contact with soil (Section 2.6.5). The exposure assessment for soil invertebrates is the same as that for terrestrial plants.
- **Effects Assessment.** Use of direct contact soil benchmarks for invertebrates follows the same approach used for plants, which includes SSL benchmarks and BABs (Table 4-5; Section 4.2.1; Appendix D).
- **Risk Estimation.** The analysis method for the soil invertebrate LOE is the HQ, as described in Section 5.1.1. and the bioavailability-adjusted, concentration-response-dependent PAFs (Section 4.2.1.2).
- For samples with triplicate IC soil samples (2014 UCR Upland Soil Study DU and 2015 Bossburg Study) or duplicate soil samples (2012 Ecology Upland Soil Study) the HQ was calculated for all replicates, and the greatest resulting HQ was retained and used in the risk estimation. This approach was used for calculating HQs for both the SSL and BAB benchmarks.

4.3.3 Risk Estimation for Wildlife

The analysis for birds and mammals is described as follows:

- **Exposure Assessment.** The LOE for the bird and mammal EAEs evaluates exposure through diet based on a daily dose. The daily dose approach models a dose to each EAE's representative bird or mammal species as a body-weight-normalized daily amount of metal ingested, expressed as mg/kg bw/day (Equation 4-1 in Section 4.1.3.1). Dose calculations provide a dietary exposure estimate based on a soil concentration. Inputs to the model include COPC and COI EPCs (Section 4.1.3), soil-to-biota bioaccumulation model parameters for food items (terrestrial plants, soil arthropods, flying insects, earthworms, and small mammals[30]) (Section 4.1.6; Tables 4-3 and 4-4), and species-specific WEFs and dietary composition (Section 4.1.5; Tables 4-1 and 4-2). Bioavailability of metals in ingested soil is estimated using sample-specific or inferred RBAs (Section 4.1.5). Metal bioavailability in biota, as well as AUFs, are assumed to be 100 percent, which are conservative assumptions that likely overestimate exposure. The conservatism associated with these exposure assumptions is evaluated in the bird and mammal uncertainty sections. Soil data from DUs of the 2014 UCR Upland Soil Study, the 2012 Ecology Upland Soil Study, and the 2015 Bossburg Study are evaluated on a point-by-point basis. For those receptors with home ranges larger than the areas sampled in each soil study, if a COPC results in sample-specific exceedances, additional analyses are conducted in the uncertainty analysis to determine if aggregations of samples throughout a realistic foraging area would result in TRV exceedances.
- **Effects Assessment.** Toxicity of COPCs to birds and mammals from dietary exposure is evaluated using daily dose-based TRVs and dose-response models for growth, reproduction, and survival endpoints. These TRVs are presented in Table 4-8, and supporting dose-response models are presented in Table 4-9. As discussed in Section 4.2.2, TRVs were developed in accordance with the BERA Work Plan (TAI, 2011) and the methodology in the approved final wildlife TRV technical memorandum (TAI, 2019d).
- **Risk Estimation.** The analysis method for wildlife LOEs is the HQ, as described in Section 5.1.1.

4.4 Background Soil Comparison

Following the risk estimation for the EAEs, EPCs are compared to background concentrations to determine if any locations with HQ ≥ 1 are less than background. Section 2.4.2 describes the process that was used to calculate BTVs. The BTVs used for the comparisons are listed in Table 2-8.

4.5 Uncertainty Analysis

Specific sources of uncertainty in the risk estimation were identified and evaluated quantitatively.

4.5.1 Uncertainty of the Bioavailability-adjusted Benchmarks

The BABs derived from the Threshold Calculator (ARCHE, 2020) and used in the plant and invertebrate risk estimates represent the median 5th percentile of the possible BAB distribution based on the data for each sample. In the plant and invertebrate uncertainty sections, a quantitative assessment was performed to evaluate the uncertainty associated with using the central tendency estimate of the BAB rather than using a lower confidence limit on the BAB. The evaluation assesses the potential to underestimate risk if the true value of the BAB is less than that of the central tendency estimate. An HQ (Section 5.1.1) was calculated for each COPC with a BAB in each location in each data set using the same soil EPC as was

[30] Small mammal tissue concentrations also serve as a proxy for large mammal (ungulates) and nonmammalian vertebrate prey (for example, birds, reptiles, and amphibians) consumed by predators.

used in the risk estimate as the numerator and using the 95% lower confidence limit (LCL) on the BAB provided by the Threshold Calculator as the denominator.

Equation 4-2: HQ Calculation for the LCL on the BAB

$$HQ_{LCL95BAB} = \frac{(Soil\ EPC)}{(LCL\ of\ \text{BAB})}$$

The number of HQs ≥ 1 for each COPC in each study was tallied and compared for the central tendency BAB HQ and the LCL BAB HQ. These analyses are presented in the uncertainty sections in the plant and invertebrate risk characterizations (Sections 6 and 7, respectively).

4.5.2 Uncertainty of the Mean EPC in Each DU

For the risk estimates for the 2015 Upland Soil and 2015 Bossburg studies, EPCs are based on COPC concentrations from either single IC soil composite sample results from DUs without replication, or the maximum HQ among triplicate IC soil composite sample results at a sample location. While these EPCs represent an estimate of the mean for each DU, they do not incorporate the variation on the estimated mean that results from spatial variation within each DU. The Interstate Technology Regulatory Council (ITRC) guidance on incremental sampling (ITRC, 2020) states that triplicate samples should be used to estimate an upper confidence limit on the mean and advises that it is preferred (but not required) to collect and analyze triplicates samples for all DUs and use the 95 UCL of each as the EPC. Alternatively, if triplicates are analyzed for a subset of DUs, ITRC (2020) recommends that variance from triplicates should be applied to the means of DUs without triplicates to extrapolate an upper confidence limit.

The uncertainty and potential to underestimate HQs ≥ 1 for each COPC in the 2014 Upland Soils and 2015 Bossburg studies due to not incorporating variance of the triplicates was evaluated as follows:

- For each set of triplicates, the 95 UCL (Student's t) was calculated as the upper bound EPC estimate for each DU in the 2014 Upland Soils and 2015 Bossburg studies.
- For each set of triplicates, the ratio of the 95 UCL EPC to the arithmetic mean EPC was then calculated for each COPC.
- The arithmetic mean of the ratio (95 UCL EPC/mean EPC) for all COPCs was 1.15 (Figure 4-1), and this value was used as a multiplier to adjust the EPC for DUs without replicates to approximate a 95 UCL EPC.
- HQs were calculated with the 95 UCL EPC as the numerator, and the number of HQs ≥ 1 for each COPC in each study was tallied and compared to that of the HQs calculated for the risk estimate (that is, using the laboratory reported single IC soil composite sample results from DUs without replication and for locations with triplicates, the maximum HQ among triplicates IC soil composite sample results).

These analyses are presented in the uncertainty analysis for each risk characterization section (Sections 6 through 9).

5. Methods Used to Characterize Risk to EAEs

The BERA Work Plan (TAI, 2011) established an analysis framework that uses LOEs to characterize risk. The risk characterization approach integrates these LOEs with analyses of COPC concentrations in background soils and an evaluation of uncertainty underlying the risk analyses to describe risk to EAEs. 5.34.369Each EAE-specific risk characterization section (Sections 6 through 9) is composed of five parts:

- **Risk Estimation**. This section presents the results of the risk analysis. The measures of exposure are integrated with the measures of effects for each LOE to describe the extent and magnitude of COPCs exceeding toxicological effects thresholds.
- **Comparison to Background Soil Concentrations**. This section distinguishes those COPCs for which toxicological threshold exceedances may be due to elevated COPC concentrations in background soils from those elevated due to site sources.
- **Uncertainty Analysis**. This section reviews and summarizes major areas of uncertainty (as well as their direction), provides additional analyses to address key uncertainties when possible, and discusses the strength, reliability, and relevance of the evidence.
- **Risk Description**. This section integrates the risk estimation, background comparison, and uncertainty analysis, and interprets them for ecological significance by assessing the likely nature and severity of effects and the associated spatial scale.
- **Risk Conclusion**. This section presents the risk conclusion for each EAE, identifies ecological COCs posing unacceptable risk to a given EAE, describes the nature of the risks, and provides a statement describing the overall degree of confidence in the risk estimates.

Each section is described in more detail following.

5.1 Risk Estimation

Two types of risk analyses are used to estimate risk for each EAE, (1) calculation of HQs and (2) comparison of exposure measures to dose-response models as described in Sections 5.1.1 and 5.1.2.

5.1.1 Hazard Quotients

All LOEs use an HQ approach to estimate the potential for adverse effects. An HQ compares the exposure estimate (Section 0) to the toxicity benchmark or TRV (Section 4.2) for each COPC, for a given EAE (Equation 5-1).

Equation 5-1: HQ Calculation

$$HQ = \frac{Exposure\ measure\ (Soil\ EPC\ or\ Ingested\ Dose)}{Effects\ measure\ (Benchmark\ or\ TRV)}$$

Where:

HQ = hazard quotient
Soil EPC = representative COPC concentration in soil calculated for each sample or DU
Ingested dose = Daily COPC intake by body weight of the organism
Benchmark = COPC threshold value for potential adverse effects (concentration in soil)
TRV = COPC toxicity reference value (daily dose by body weight of the organism)

All HQs were calculated deterministically wherein a single value each was used to represent the exposure measure and the effects measure. HQs are presented to two significant figures (e.g., 0.32, 8.0, and 21).

Variability and uncertainty underlying the measures of exposure and effect are discussed in the uncertainty analysis for each EAE. As described in Section 3, the 2014 UCR Upland Soil Study, the 2012 Ecology Upland Soil Study, and the 2015 Bossburg Study data sets provide different sampling density and spatial coverages that are important to consider when interpreting HQs. Thus, HQs for the three data sets are presented and interpreted separately in the risk estimation sections and results are integrated in the uncertainty analyses, risk characterization, and risk conclusions sections, as appropriate. All HQ calculations are presented in full in Appendix F.4.1.1

5.1.2 Dose-Response and Concentration-Response Information

As described in Section 5.1.1, HQs are calculated for every EPC for each COPC that has a benchmark or TRV. The HQ based on the benchmark or TRV is first used as a simple binary metric; either the HQ is greater than or equal to (≥) 1 or it is less than (<) 1. It represents a decision point.

- HQ < 1: The estimated exposure is unlikely to cause an adverse effect leading to unacceptable risk to the EAE. Unacceptable risk can be ruled out at this point in the evaluation, and further analysis is not warranted.
- HQ ≥ 1: Unacceptable risk cannot be ruled out at this point in the evaluation. Additional analysis and evaluation are warranted to quantify the nature and magnitude of potential adverse effects associated with the estimated exposure at each applicable location and to determine whether unacceptable risk is present.

As a second step for receptor-COPC combinations with HQ ≥ 1, EDx values (wildlife) or PAFs (plants and invertebrates) are calculated when exposure-response models are available (Sections 4.2.1.2 and 4.2.2). These EDx values or PAFs are then used to characterize the potential for adverse effects on the EAE. When EDx values or PAFs are not available (i.e., the TRV is a LOAEL and dose-response models are not available, or there is no BAB available), HQs provide the only estimate for characterizing the potential for adverse effects.

In a limited number of situations, a discrepancy occurs between the results from the HQ and that from the calculated EDx or PAF value. For example, an HQ of 1.1 has a calculated EDx value of 19 (when the EDx value would be expected to exceed 20 because the TRV was an ED20). This discrepancy is due to the rounding of the TRV to two significant digits prior to calculating the HQ, but not rounding the parameters used to calculate the EDx value (Section 4.2.2). For these cases, the HQ is used first as the screening step. Further dose-response evaluation for HQs ≥ 1 is then conducted, and calculated EDx values < 20 are considered unlikely to cause an adverse effect leading to unacceptable risk to the EAE.

A similar situation also occurs for the PAF for plants and invertebrates. The HQ is used first as a screening step. Further concentration-response evaluation for HQs ≥ 1 is conducted, and calculated PAFs less than 5 percent (although PAFs would, by definition of the ED5-based TRV, exceed 5 percent for HQs ≥ 1) are considered unlikely to cause adverse effects leading to unacceptable risk to the EAE.

5.2 Comparison to Soil Background

As described in Section 2.5.4, soil BTVs were developed to provide context for the HQs calculated in the Upland BERA (TAI, 2020a) and are presented in Table 2-8. The BTVs are important context for understanding HQ results. Thus, HQs for a given COPC and EPC are presented in both figures and maps in conjunction with whether the sample's soil concentration is either greater than or less than the BTV.5.3

5.3 Uncertainty Analysis

This section reviews and summarizes major areas of uncertainty (as well as their direction), provides additional analyses to address key uncertainties when possible, and discusses the strength, reliability, and relevance of the evidence. Uncertainty analyses include the following:

- Identifying and describing the sources of uncertainty in the problem formulation and risk characterization
- Evaluating various scenarios for exposure and effects in the risk analyses to capture the range of uncertainties in assumptions
- Conducting sensitivity analyses to examine the effect of variability in the exposure and effects parameters on the risk estimates

When multiple LOEs are used to assess risk to an EAE, a WOE framework is needed to reconcile any inconsistencies as well as to determine the reliability of all available LOEs for a given receptor-COPC pair (EPA, 2016b). In 2020, TAI and EPA held a series of collaborative calls on the WOE approach for the UCR BERA. In the Upland BERA, two receptor groups employ two LOEs to assess risk (Table 2-12). However, because the two LOEs used to assess risk are progressive with the second LOE superseding the first, no formal process is needed to weigh the LOEs relative to one another. Therefore, no WOE was needed nor used in this Upland BERA. However, one important concept, related to WOE, a statement of confidence in the overall risk conclusions, is synthesized from the uncertainty analysis and presented in the risk conclusions for each EAE. Confidence is derived from the reliability, relevance, and strength of the data including the collective properties (number, diversity, sufficiency; absence of bias; and coherence) of the LOEs.

5.4 Risk Description

The risk description integrates the risk estimation, comparison to background, and uncertainty analysis subsections to describe the risk to each EAE. The risk description is conducted in accordance with EPA's guidance to interpret whether the potential adverse effects "represent changes that are undesirable because they alter valued structural or functional attributes of the ecological entities under consideration" (EPA, 1998). The risk description is narrative, with a focus on EAEs and COPCs with the potential for adverse effects. Discussions are provided for important concepts in the risk description including spatial scale, nature and severity of effects, and translation of predicted impacts on survival, growth, and reproduction of individual receptors to population- or community-level impacts. The risk description classifies each COPC into one of the following three categories:

- **Negligible risk.** Exposure to COPC is not expected to elicit adverse effects on the receptor population or community and thus unacceptable risk can be ruled out. No further evaluation is warranted.
- **Unacceptable risk.** Exposure to COPC has a sufficient likelihood of eliciting an adverse effect on the receptor population or community that consideration in risk management decisions is needed. The COPC is identified as a COC and is recommended to be addressed in the Upland RI with a discussion regarding nature, extent, fate, and transport.
- **Uncertain risk.** Insufficient information is available to assess risk to the receptor-COPC pair. This category is limited to COPCs for which toxicological effects data are insufficient for a given receptor group to make reliable risk conclusions. The COPC is recommended to be addressed in the Upland RI to determine whether it spatially co-occurs with COCs and in the FS to ensure that remedial measures address any associated risk.

5.5 Risk Conclusions

The risk conclusion summarizes the findings of the risk analysis for each EAE. Ecological COCs posing unacceptable risk to a given EAE are identified along with a description of the nature of the risks, and a statement describing the overall degree of confidence in the risk estimates.

6. Risk Characterization for Terrestrial Plants

This risk characterization section assesses whether exposure of plants to COPCs in soils may pose an unacceptable risk in the Terrestrial Study Area. The problem formulation for terrestrial plants, including an ecological overview, is presented in Section 2. The methods for exposure and effects assessment and analysis are presented in Section 4, and the approach for integrating this information to characterize risks is presented in Section 5.

The risk characterization is structured as follows:

- Risk Estimation: presents the results of the risk analysis
- Comparison to Background Soil Concentrations: compares site soils to BTVs for each COPC
- Uncertainty Analysis: analyzes and discusses the major areas of uncertainty
- Risk Description: describes the likely nature, intensity, and spatial scale of effects
- Risk Conclusions: summarizes the risks and overall degree of confidence in the risk estimates

The terrestrial plant community is the ecological entity for the plant EAE. The EAE-specific risk questions are as follows:

- **SSL benchmarks:** Are the concentrations of COPCs in soils in the Terrestrial Study Area greater than soil screening benchmarks for the survival, growth, and reproduction of terrestrial plants, such that adverse effects to the local community are expected?
- **BABs:** Are the concentrations of COPCs in soils in the Terrestrial Study Area greater than bioavailability-adjusted soil benchmarks for the survival, growth, and reproduction of terrestrial plants, such that adverse effects to the local community are expected?

The two LOEs used for characterizing risk to the terrestrial plant EAE are as follows:

- LOE 1: COPC concentrations in soil compared to bulk SSL benchmarks (Eco-SSLs and SSLs). All COPCs have SSL benchmarks available (Table 4-5).
- LOE 2: COPC concentrations in soil compared to BABs. COPCs for plants that have BABs are cobalt, copper, lead, molybdenum, nickel, and zinc (Table 4-5). For COPCs with BABs, in addition to the HQ, the PAF of plant species is also calculated from the SSD of plant EC20s. The PAF estimates the fraction of the plant community adversely affected at a given sample location.

These two types of soil benchmark comparisons are not independent LOEs. The BABs are a refinement of the SSL benchmarks using additional soil chemistry data to make inferences about bioavailability of the COPCs in each sample. For COPCs and/or sample locations having sufficient data, soil benchmarks are adjusted for bioavailability and the adjusted soil benchmarks and PAFs are given precedence over the SSLs for the risk characterization. For those COPCs that have both BABs and SSLs, HQs are calculated for both benchmarks, but the risk characterization describes risks for each sample location/COPC combination based only on the BAB.

6.1 Risk Estimation

This section presents the risk analyses for the terrestrial plant community EAE and summarizes the results. For each LOE, HQs are calculated for every sample and COPC for which there is a soil benchmark. HQs are interpreted as follows:

- **HQ < 1:** The estimated exposure is unlikely to cause an adverse effect to the EAE. Risk is negligible.
- **HQ ≥ 1:** Unacceptable risk cannot be ruled out. The nature and magnitude of potential adverse effects associated with the estimated exposure at each applicable location is further evaluated and described.

Data sets used for the plant risk characterization are the 2012 Ecology Upland Soil Study, the 2014 UCR Upland Soil Study, and the 2015 Bossburg Study (Ecology, 2013; TAI, 2015, 2016). Samples from each study are compared to the SSL and BAB benchmarks. Because percent clay and eCEC, two soil parameters necessary to calculate BABs, were not measured as part of the 2012 Ecology Upland Soil Study, sample-specific values were assumed to be represented by the minimum percent clay and eCEC from 2014 UCR Upland Soil Study DUs that overlap respective 2012 Ecology Upland Soil Study subareas where samples were collected. The analyses performed to estimate percent clay and eCEC for the 2012 Ecology Upland Soil Study samples are presented in Appendix F and discussed in the uncertainty analysis.

For each data set, the number and percentage of samples with HQs ≥ 1 and ≥ 5 for the BAB and SSL, and the median and maximum PAFs are presented in Table 6-1. For those COPCs evaluated using BABs, Figure series 6-1a through 6-1f shows boxplots and empirical cumulative distribution functions of concentrations relative to PAFs. For those COPCs evaluated using only the Eco-SSL or SSL (not including aluminum and iron), figures showing boxplots and empirical cumulative distribution functions of concentrations relative to SSL benchmarks are provided on Figure series 6-2a through 6-2m. All HQ and PAF calculations are presented in Appendix F. For those COPCs with a BAB, BAB HQs and PAFs are considered in risk estimation and SSL HQs are not discussed further.

For each sample in each data set, the COPCs exceeding the BAB benchmark if available, or the Eco-SSL or SSL if not, are presented in Table 6-2. Table 6-2 also provides HQs, PAFs, and BTV comparisons for each sample as available.

For COPCs with soil concentrations exceeding the BAB if available, or the Eco-SSL or SSL if not, Maps 6-1 through 6-11 show sample locations where exceedances occur as well as those where concentrations result in an HQ ≥ 5 or PAF ≥ 20[31] and whether samples are greater than or less than the BTV. The colors green, yellow, and blue indicate the HQ range categorically at each location. For aluminum and iron, symbols are based on pH range rather than HQ. The closed symbols indicate concentrations > BTV, while open symbols represent concentrations ≤ BTV. The results in Table 6-1 and on Figures 6-1a through 6-1f and 6-2a through 6-2m are briefly described as follows:

- **Lead and zinc** BAB HQs are ≥ 1 in many samples from the 2012 Ecology Upland Soil Study, the 2014 UCR Upland Soil Study, and the 2015 Bossburg Study (Table 6-1).
  - Lead HQs are ≥ 1 in 50 percent of Bossburg Study samples, 40 percent of Ecology samples, and 13 percent of 2014 UCR Upland Soil Study samples. Maximum PAFs are > 50 percent in samples from the 2012 Ecology Upland Soil and 2015 Bossburg studies (Figure 6-1c).
  - Zinc median PAFs are ≥ 12 percent in samples from all three data sets with maximum PAFs ≥ 42 percent in all three data sets (Table 6-1; Figure 6-1f). Most (67 percent or greater) samples in all three data sets exceed the BAB, which is equivalent to a 5 percent PAF.
- **Arsenic, barium, chromium, manganese, and selenium** HQs are ≥ 1 for SSL benchmarks in one or more data sets (Figure series 6-2a through 6-2m). BABs are not available for these COPCs. Nearly all manganese samples (97 percent to 100 percent, depending upon the data set) exceed the SSL (Table 61; Figure 6-2h).

[31] HQ ≥ 5 and PAF ≥ 20 are included as a standardized effect metric that is a consistent indicator of significant potential for adverse effects (that is, the soil concentration is more than 5-fold greater than the HC5 [concentration protective of 95 percent of species] and/or is likely to produce effects in more than 20 percent or exposed species [PAF]).

- **Cobalt and nickel** shared the following conditions:
  - BAB HQs are ≥ 1 in only a few samples from the 2012 Ecology Upland Soil Study (< 3 percent**)**. There are no exceedances of BABs for the 2014 UCR Upland Soil Study or the 2015 Bossburg Study for either cobalt or nickel.
  - Nickel PAFs are > 25 percent in two 2012 Ecology Upland Soil Study samples (Figure 6-1a).
  - Most (> 99 percent) cobalt samples are well below the BAB and the maximum cobalt PAF (5.2 percent) only slightly exceeds the BAB of 5 percent of species affected (Table 6-1, Figure 6-1a).
- **Aluminum and iron** Eco-SSLs are based on pH; for each there are some instances in which pH is below the threshold, suggesting the potential for effects from both metals on plants.
- **Antimony and thallium** concentrations do not exceed SSLs in any sample. BABs are not available.
- **Copper and molybdenum** exceed neither SSLs nor BABs. No molybdenum data are available from the 2012 Ecology Upland Soil Study or the 2015 Bossburg Study.

In general, the greatest number of HQs ≥ 1 and highest HQs are for lead, manganese, and zinc.

6.2 Comparison to Background

This section presents a comparison of soil concentrations measured in each of the three data sets to the BTVs shown in Table 2-8.

- COPCs that exceed background and exceed the soil benchmark associated with the highest LOE evaluated for a given COPC (BAB if available, otherwise the SSL) in at least one sample from one study include arsenic, barium, chromium, cobalt, lead, manganese, nickel, selenium, and zinc (Table 6-3).
- Lead, manganese, and zinc have the greatest number and proportion of exceedances of soil benchmarks and BTVs.
  - All measured lead concentrations in the three studies exceed the BTV (Figure 6-1c). Of the samples from the 2012 Ecology Upland Soil Study, the 2014 UCR Upland Soil Study, and the 2015 Bossburg Study, 40 percent, 13 percent, and 50 percent, respectively of lead concentrations exceed the BTV and the BAB.
  - While most samples from all three data sets exceed the manganese Eco-SSL, more than half of the Eco-SSL exceedances are below the BTV (Figure 6-2h).
  - For zinc, most samples have concentrations above the BTV and the BAB. All but three, four, and two zinc concentrations exceed the BTV in the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg, respectively. Of the samples from the 2012 Ecology Upland Soil Study, the 2014 UCR Upland Soil Study, and the 2015 Bossburg Study, 69 percent, 83 percent, and 50 percent, respectively of zinc concentrations exceed the BTV and the BAB. Of the samples from the 2012 Ecology Upland Soil Study, the 2014 UCR Upland Soil Study, and the 2015 Bossburg Study, four, four, and two samples had zinc concentrations below the BTV.
  - Arsenic and selenium also have multiple exceedances of soil benchmarks and BTVs (Tables 6-1 through 6-3; Figures 6-2b and 6-2k; Maps 6-2 and 6-10). 2014 UCR Upland Soil Study and 2012 Ecology Upland Soil Study samples exceeding selenium SSLs and BTVs are generally dispersed on the east side of the UCR including the easternmost locations sampled.

6.3 Uncertainty Analysis

Uncertainties associated with the risk characterization for plants are presented in the following subsections and in Table 6-4. Key uncertainties associated with each component of the risk assessment are also evaluated.

6.3.1 Problem Formulation

In the BERA Work Plan (TAI, 2011), foliar uptake is identified as a potentially complete but minor exposure pathway (Section 2.6.5). Shahid et al. (2017) performed a review of the available data on foliar uptake of metals and reports that while foliar uptake can impact physiological and metabolic processes, the severity of impact is generally less than from root uptake, with some foliar exposure studies showing no effect (Shahid et al., 2017). The paucity of data on the foliar pathway presents an uncertainty in the risk characterization for plants. However, consistent with Shahid et al. (2017) and because it is identified as a minor exposure pathway in the BERA Work Plan, exclusion of the foliar pathway is unlikely to result in over- or underestimates of risk.

6.3.2 Exposure Assessment

Given the large area of the Terrestrial Study Area and sampling limitations, the soil chemistry data set only partially characterizes the full magnitude and extent of contamination. Despite the availability of the three upland soil data sets, large portions of the Terrestrial Study Area are uncharacterized (Map 2-1). As noted in Section 1, the extent of OU3 is expected to be determined by analyses presented in the Draft Final Upland RI Report. The data sets and data usability assessment for each soil study are described in Section 3. The strengths and uncertainties associated with each of the three soil studies used to characterize exposure to plants are discussed in brief following and key uncertainties are further analyzed to determine their impact on risk conclusions.

6.3.2.1 2012 Ecology Upland Soil Study

EPCs are the reported COPC concentrations for each sample. Each sample is based on a composite of four samples averaged over 0.025 acres. This spatial scale is relevant to exposure of immobile receptors such as plants. Samples from this study, concentrated in the northern portion of the Terrestrial Study Area adjacent to the Canadian border, capture small-scale variation of COPC concentrations with minimal spatial averaging due to both the higher spatial density of samples and the smaller area represented by each sample. Because the area sampled is limited to the northern portion of the Terrestrial Study Area, exposure estimates and risk estimates for localized plant communities at a small spatial scale are limited to this portion of the Terrestrial Study Area.

The use of the measured percent clay and eCEC data from the 2014 UCR Upland Soil Study for 2012 Ecology Upland Soil Study samples imparts some uncertainty to the calculated BAB and PAF values. Because percent clay and eCEC values were selected based on ADA DUs located within each respective Ecology subarea, a degree of the spatial variability in these parameters is retained within BAB and PAF calculations. Analyses presented in full in Appendix F show that use of minimum percent clay and eCEC values results in an intermediate level of COPC bioavailability relative to alternative percent clay and eCEC assumptions. Substitution of percent clay and eCEC data calculated for Ecology subareas into BAB calculations for adjacent 2014 UCR Upland Soil Study samples closely approximated the BABs calculated from sample-specific data, indicating that this approach provides reasonable conservative estimates of the BABs and PAFs.

6.3.2.2 2015 Bossburg Study

EPCs are the reported COPC concentrations for each sample. Each sample is based on a 30-point incremental composite samples (ICS), over a 1- to 3-acre DU. ICS averages soil chemistry throughout each DU so exposure estimates from this study represent somewhat localized plant communities at a small spatial scale. Sampling was limited to the Bossburg Flats area, and this is the only data set representing this portion of the Terrestrial Study Area.

Because ICS samples represent an estimate of the mean concentration within a DU, EPCs may under- or overestimate the true mean COPC concentration due to site variability and sampling error. EPA guidance recommends that EPCs be calculated as the 95 UCL to ensure that exposure is not underestimated (EPA, 1992, 2002c).

A triplicate sample was collected at one ICS location, which is used to calculate the 95 UCL for each COPC following ITRC guidance and EPA risk assessment forum (RAF) recommendations for the UCR human health risk assessment (HHRA) and BERA (ITRC, 2020; EPA, 2021). The ratio of the 95 UCL to the arithmetic mean of the triplicate samples is shown on Figure 4-1. Because of the small number of soil samples, the uncertainty analysis for the Bossburg soil data set was combined with that of the 2014 UCR Upland Soil Study, as described in Section 6.3.2.3.

6.3.2.3 2014 UCR Upland Soil Study

EPCs for the 2014 UCR Upland Soil Study are based on 25-acre DUs, each composed of 30 point-location increments composited into an ICS sample. This approach averages soil chemistry throughout each DU. Due to compositing across increments, this data set does not capture smaller areas of greater exposure that could be important for local plant communities within the DUs. Samples were collected from randomly selected locations throughout the Terrestrial Study Area, excluding steep slopes, areas within 50 meters or greater than 550 meters of roads, and areas within 500 meters of active or abandoned mines. Due to uncertainty in the extent of contamination, the outer boundaries of contamination associated with unacceptable risks in OU3 may not be captured within the area sampled (TAI, 2014a). Additionally, contamination in the unsampled areas between sampling locations is uncertain.

EPCs are calculated as the reported COPC concentrations determined for each DU which may under- or overestimate the true mean COPC concentration in the DU due to site variability and sampling error. An uncertainty analysis was conducted by calculating HQs for each DU in the 2015 Bossburg and 2014 UCR Upland Soil data sets as the 95 UCL divided by the highest tiered soil benchmark (BAB if available, otherwise Eco-SSL or SSL) rather than the ICS estimated mean/soil benchmark. For samples without triplicates, the upper confidence limit (UCL) was estimated as the reported COPC concentration multiplied by the 95 UCL: mean ratio of 1.15 averaged over all COPCs in all triplicates from the 2015 Bossburg and 2014 UCR Upland Soil studies (Table 6-5). When calculated using the 95 UCL, the number of DUs with HQs $\geq$ 1.0 in the 2015 Bossburg Study data set increases only for lead, whereas for the 2014 UCR Upland Soil Study, the number of DUs with HQ $\geq$ 1 increases for arsenic, barium, lead, nickel, selenium, and zinc, and the number of HQ $\geq$ 5 increases for manganese (Table 6-5). These calculations indicate that use of the reported concentration (ICS estimated mean) for each DU as the EPC rather than the 95 UCL may substantially underestimate the spatial extent and magnitude of risk to plants.

6.3.3 Effects Assessment

Uncertainties associated with Eco-SSLs and SSL benchmarks and with BABs are summarized in Table 6-4 and as follows.

6.3.3.1 Ecological Soil Screening Levels and Soil Screening Levels

General uncertainties related to the Eco-SSLs and SSLs include their unknown ability to predict site-specific effects, the relevance of the plant species in the underlying toxicity tests to the plant communities in the Terrestrial Study Area, and the extent to which conditions in the toxicity tests represent conditions in the Terrestrial Study Area (for example, conditions related to bioavailability) (Table 6-4).

In addition to the general uncertainties related to Eco-SSLs and SSLs, there are some uncertainties relevant to the soil benchmarks for specific COPCs. The following COPCs have HQs ≥ 1, in at least one of the three data sets, for the terrestrial plant community EAE (Table 6-1): aluminum, arsenic, barium, chromium, cobalt, iron, lead, manganese, nickel, selenium, and zinc. Additional information on these Eco-SSLs/SSLs is provided in Table 4-6, and the uncertainties, such as their reliance on soil pH as a predictor for possible toxicity (aluminum and iron) and the bioavailability of metals species used in toxicity tests (arsenic, barium, chromium, cobalt, copper, lead, manganese, selenium, and zinc), are discussed in Table 6-4.

In general, uncertainties in the SSLs are likely to overestimate risk; however, uncertainty associated with the arsenic Eco-SSL may underestimate risk. The Eco-SSL of 18 milligrams per kilogram (mg/kg) is calculated as the geomean of three maximum acceptable toxic concentration (MATCs) representing three species tested under high bioavailability conditions (EPA, 2005d). Four acceptable studies reporting toxicity values less than the plant Eco-SSL were excluded from Eco-SSL derivation according to derivation rules. Two barley growth lowest observed effect concentrations (LOECs) (Jiang and Singh, 1994) (LOEC of 2 mg/kg) were excluded because they are unbounded LOECs and a radish and a soybean population MATC both 6 mg/kg (Woolson and Isensee, 1981) were excluded because studies with higher bioavailability scores were available. Additionally, after Eco-SSL publication, New Zealand derived a soil quality guideline protective of plants following methods consistent with BAB, Australian, and EU REACH soil quality guideline derivation methods (Cavanagh and Munir, 2019). SSDs were fit to chronic toxicity EC10s and EC30s including added arsenic data on plant growth or yield for 21 plant species (10 to 2,500 mg/kg). The 5th percentiles of the added arsenic EC10 and EC30 SSDs are 6 and 16 mg/kg, respectively. This analysis indicates that the Eco-SSL (18 mg/kg) may not be protective of low levels of growth impacts in the most sensitive species. Additionally, at concentrations above the Eco-SSL > 5 percent of species may experience substantial reductions in growth.

6.3.3.2 Bioavailability-adjusted Benchmarks

The sources of uncertainty associated with BABs are described in Table 6-4. The BAB is an HC5, which is the concentration expected to cause a 20 percent adverse effect on 5 percent of plant species (Oorts-2020). The BAB is calculated as the median 5th percentile of the fitted log-normal distribution of plant toxicity data. Upper- and lower-95 percent confidence limits (UCLs and LCLs, respectively) on the BAB are also calculated for each COPC (Table 6-6). The ranges of confidence limits are broad because in addition to variability about the benchmark, they encompass a wide range of bioavailability conditions throughout the site. The importance of variability in the toxicity data underlying the BABs is best illustrated by the difference in the number of HQs ≥ 1 when calculated as the EPC/(LCL on BAB) rather than the EPC/BAB. As shown in Table 6-6, many more samples exceeded the LCL on the BAB for lead and zinc indicating that uncertainty in the BABs may underestimate the extent of risk from these COPCs. In contrast, few samples exceed the LCL on the BAB for copper and molybdenum, and to a lesser extent cobalt and nickel, increasing confidence that these chemicals pose low risk to plants. Where exceedances of the LCL on the BAB occur, cobalt PAFs are less than 1 percent and nickel PAFs are less than or equal to 3.3 percent. While confidence limits on the PAFs are not calculated, this indicates that most plant species would likely not be affected by site cobalt or nickel. Though not calculated, a similar reduction in the number of HQs ≥ 1 would result from calculating HQs based on the UCL on the BABs illustrating that uncertainty in the BABs may also lead to overestimates of risk.

A detailed discussion of the uncertainties associated with the BABs is also presented in Appendix D. The soil benchmarks account for bioavailability based on regression-based equations and conservative default leaching/aging factors. The uncertainties with the soil benchmarks include their unknown ability to predict site-specific effects, limited data sets underlying the bioavailability models, the lack of site-specific plant species in the underlying toxicity data, and the unknown extent to which conditions in the toxicity tests represent conditions in the field (Table 6-4).

6.3.4 Risk Characterization

6.3.4.1 Use of Hazard Quotients to Assess Risk

HQs do not give a quantitative prediction of the likelihood or severity of adverse effects, although they are expected to increase as the HQ increases. HQs less than 1, when calculated using conservative assumptions, provide compelling evidence of negligible risk because the COPC concentrations are less than the toxicological effects threshold. When an HQ is ≥ 1, additional consideration must be given to the dose-response data underlying the effects data to predict whether exposure similar to site exposures are associated with serious effects on test organisms. This limitation is minimized through the integration of concentration-response or dose-response data with the exposure estimates. Such information has been incorporated into the Upland BERA where applicable and available, such as through the use of the PAF.

6.3.4.2 **2.5.4Metal Interactions**

There is uncertainty in the approach for assessing risks to terrestrial plants related to possible interactions among metals. As discussed in Section 1.3, this issue is one of the basic principles to be considered in assessing risks posed by metals identified by EPA (EPA, 2007a). The complexities associated with exposure to mixtures of metals contribute significant uncertainty to the evaluation of risks to terrestrial vegetation from exposure to metals.

The use of single-chemical HQs does not account for potential changes in toxicity caused by the presence of chemical mixtures. Interactions among metals occur by competition for binding locations on specific enzymes or on cellular receptors during the processes of absorption, excretion, or sequestration at the target site (EPA, 2007a). For example, one mechanism of action causing metals toxicity is an impact on chlorophyll. Chlorophyll biosynthesis is critical to photosynthesis and zinc toxicity results in impacts to chlorophyll biosynthesis (Chaney, 1993). Plants require iron and manganese for the synthesis of chlorophyll and lead toxicity impairs the uptake of iron and manganese (Singh et al., 2016). Therefore, zinc and lead exposure may interact to cause additive or synergistic toxicity. Other metals known to have toxicological interactions in plants include antimony, arsenic, copper, and nickel. Metal mixtures have shown varying simple and complex toxicological interactions relative to single metals. For example, arsenic and zinc mixtures can have either additive or antagonistic effects on barley root elongation (Guzman-Rangel et al., 2017, 2018). In barley root elongation toxicity tests, copper, nickel, cadmium, and zinc metal mixtures inhibited root elongation up to 50 percent compared with less than 10 percent for single metals (Versieren et al., 2016). In a study of arsenic, copper, and zinc in wheat root elongation, arsenic-copper-zinc mixtures had additive effects, while binary mixtures had antagonistic effects (arsenic-copper, arsenic-zinc) or dose-dependent synergistic effects (copper-zinc) (Gong et al., 2020). Depending on the type of toxicological interaction (for example, additivity, antagonism, potentiation, or synergism) and the respective exposures for the metals, the single-chemical HQ may overestimate or underestimate the potential for adverse effects.

Table 6-2 identifies locations where multiple COCs exceed soil benchmarks protective of plants (BAB if available, otherwise the Eco-SSL or SSL). While the specific toxic effects of these exceedances cannot be predicted without site-specific toxicity testing, locations with exceedances of multiple COPCs generally pose a greater risk to plants than those with fewer exceedances, with higher PAFs and HQs indicating higher risks.

6.3.4.3 Metal Essentiality

The fact that some metals are essential for maintaining proper health of plants also contributes uncertainty in plant risk estimates. As discussed in Section 1.3, this issue is one of the basic principles to be considered in assessing risks posed by metals identified by EPA (EPA, 2007a). Of the COPCs for plants, copper, manganese, molybdenum, nickel, and zinc are essential to plants, while cobalt and selenium are beneficial to plants (EPA, 2007a). Some level of these metals is necessary or beneficial for vital functioning, such as nitrogen fixation (cobalt, copper, molybdenum) and translocation (nickel); valence changes that stimulate chlorophyll production (cobalt, copper, molybdenum); oxygen production (manganese); metabolism (copper, zinc); and sulfur substitution (selenium) (Kabata-Pendias, 2011). Different plant species may have different nutritional requirements or optimal conditions for essential metals, and the extent to which the nutritional requirements of the test plant species are representative of the plant species that occur within the Terrestrial Study Area is uncertain. The implications of this uncertainty on the estimation of risks to terrestrial plants is unclear.

6.3.5 Background Analysis

As discussed in Section 2.5.44, BTVs were calculated to represent natural background soil concentrations as defined by EPA (EPA, 2002b) and by the State of Washington in the MTCA (WAC 173340200). BTVs were estimated from a subset of the available regional soil COPC concentration data within the vicinity of the Terrestrial Study Area and with minimal influence of historical smelter emissions. Because the BTVs are an approximation of regional natural background COPC concentrations, there is uncertainty as to how well they represent the range of concentrations that may be naturally present at finer scales within the study area.

The BTVs are compared against the 90th percentile of natural background concentration range across Northeast Washington (Ecology, 2019b). In general, the BTVs calculated for the Terrestrial Study Area fell within the range of other regional background concentrations (Table 6-7). However, the BTVs for antimony, barium, chromium, and silver are less than the lower range of the 90th percentile natural background concentrations. This suggests that risks from these elements relative to background may be overestimated in the Terrestrial Study Area. Additionally, the BTV for arsenic (23.3 mg/kg) is greater than the upper range of the 90th percentile natural background concentrations (20 mg/kg), suggesting screening against the BTVs may somewhat underestimate risks relative to background for arsenic. Overall, with the previously noted exceptions, the BTVs selected for use in the Terrestrial Study Area are in line with other available regional background values. The BTVs are therefore expected to contribute minimal uncertainty to the risk assessment.

6.4 Risk Description

As described in Section 5.4, the risk description integrates the findings from the risk estimation (Section 6.1), comparison to background (Section 6.2), and uncertainty (Section 6.3) sections to describe the risk to plants resulting from COPCs in the Terrestrial Study Area. The risk description also includes discussion of the spatial scale and the nature and severity of potential adverse effects.

6.4.1 Individual COPCs

Risk descriptions for individual COPCs are as follows:

- Aluminum
  - HQs: In the case of aluminum, HQs are not applicable because the screening threshold for aluminum is based on pH; when pH is less than 5.5, aluminum presents an ecological risk. There are 17, 19, and 1 locations in the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies, respectively with pH ≤ 5.5 (Table 6-1).
  - Nature and severity: Where pH is sufficiently low, excess aluminum exposure may cause adverse effects, such as reduced growth (Table 6-1, Table 4-6), but effects vary considerably among species (EPA, 2003b).
  - Spatial distribution: Locations with pH less than 5.5 occur sporadically through the Terrestrial Study Area but appear less frequent in the southern portion of the study area (Table 6-2, Map 6-1).
  - Background: Aluminum is below the BTV of 40,500 mg/kg in all locations (Table 6-3, Table 2-8).
  - Specific uncertainty: EPA (2003b) notes the nature and severity effects of aluminum vary considerably among species, and aside from bioavailability at pH less than 5.5, a specific threshold concentration for effects was not established.
  - Conclusion: The pH is sufficiently low in between 13 percent and 17 percent of locations in the Terrestrial Study Area that ecological risk due to aluminum could be of concern. However, aluminum concentrations do not exceed the BTV in any locations. Therefore, aluminum poses negligible risk and is not retained as a COC for plants.
- Antimony
  - HQs: HQs for SSLs are less than 1 for all samples from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies.
  - Nature and severity: There are no exceedances of the SSL at any location indicating adverse effects are unlikely.
  - Spatial distribution: There are no exceedances of the SSL at any location.
  - Background: There are 32, 141, and 6 locations from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies, respectively that exceeded background (Table 6-2, Table 2-8).
  - Specific uncertainty: No specific uncertainties affect risk conclusions (Table 4-6, Table 6-4).
  - Conclusion: Because there are no exceedances of the SSL at any locations and uncertainty analyses do not indicate increased risks, antimony poses negligible risk and is not retained as a COC for plants.

- Arsenic
  - HQs: HQs for Eco-SSLs exceeded 1 in 34 and 42 of locations (32 percent and 30 percent) for the 2012 Ecology Upland Soil and 2014 UCR Upland Soil studies, respectively, and no 2015 Bossburg Study locations. BABs are not available for arsenic (Tables 6-1, Figure 6-2b).
  - Nature and severity: The SSL for arsenic is based on reduced plant growth (Table 6-1, Table 4-6). At locations exceeding the SSL (18 mg/kg), most have HQs between 1 and 3 (Table 6-2, Figure 6-2b). SSDs of arsenic toxicity to plants described in the uncertainty analysis indicate that >5 percent of species experience >30 percent adverse effects on plant growth or yield at concentrations >22 mg/kg, which is close to the BTV.
  - Spatial distribution: 2012 Ecology Upland Soil Study and 2014 UCR Upland Soil samples exceeding arsenic SSLs are generally located close to the UCR along the length of the area sampled in each soil study (Map 6-2). Arsenic exceedances co-occur with at least one other COPC at all locations (Table 6-2).
  - Background: The arsenic BTV is 23.3 mg/kg and the SSL is 18 mg/kg. Therefore, 46 percent of the samples with HQs ≥ 1 also exceed background (Table 6-3, Table 2-8; Figure 6-2b).
  - Specific uncertainty: Uncertainty in the EPC may underestimate the number of DUs with HQs ≥ 1 (Table 6-5). Uncertainty in the Eco-SSL indicates that low levels of effects in the most sensitive species may occur at concentrations somewhat lower than the Eco-SSL. The site-specific BTV (23.3 mg/kg) is greater than background concentrations calculated by Ecology across Northeast Washington (20 mg/kg) suggesting screening against the BTVs may somewhat underestimate risks.
  - Conclusion: There is risk of reduced plant growth throughout the Terrestrial Study due to exposure to arsenic concentrations that exceed background. Arsenic poses unacceptable risk and is carried forward into the RI as a COC for plants.
- Barium
  - HQs: HQs for SSLs equaled or exceeded one in only one and two locations (0.9 percent and 1.4 percent) for the 2012 Ecology Upland Soil and 2014 UCR Upland Soil studies, respectively. BABs are not available for barium (Table 6-1; Figure 6-2c).
  - Nature and severity: The SSL of 1,400 mg/kg is based on a single reduced plant growth MATC in one species (Table 6-1, Table 6-4). At locations exceeding or equal to the SSL, the HQs are 1.8, 1.0, and 1.0, indicating that concentrations at these locations are close to or below the lowest concentrations observed to cause adverse effects (Table 6-2, Figure 6-2c).
  - Spatial distribution: There is one isolated 2012 Ecology Upland Soil Study sample and two adjacent 2014 UCR Upland Soil Study locations where barium equals or exceeds the SSL (Map 6-3). In each of these locations, barium is not the only metal with HQ ≥ 1 (Table 6-2).
  - Background: The barium BTV of 395 mg/kg is less than the SSL, so the three values with HQ ≥ 1 also exceeded background (Table 6-3, Table 2-8).
  - Specific uncertainty: Uncertainty in the EPC may slightly underestimate the number of DUs with HQ ≥ 1. The barium SSL was developed by TAI using a method similar to that of EPA Eco-SSL guidance (EPA, 2005e). A single study reporting a MATC was used, thus the SSL has more uncertainty than Eco-SSLs developed for COPCs with significantly more data (such as zinc). The bioavailability of the barium in the soil benchmark study was determined to be low, making the SSL potentially less conservative than SSLs for studies with highly bioavailable metals.

  - Conclusion: Because barium exceeds the SSL in only about 1 percent of locations with HQs less than 2, the benchmark may overestimate the potential for effects, and many other metals also exceed their soil benchmarks in the same locations; barium poses negligible risk and is not retained as a COC for plants.

- Chromium
  - HQs: HQs for SSL soil benchmarks exceed 1 in a single location (0.9 percent of locations) from the 2012 Ecology Upland Soil Study. HQs for all locations in the 2014 UCR Upland Soil and 2015 Bossburg studies are less than 1. BABs are not available for chromium (Tables 6-1 and 6-2).
  - Nature and severity: The SSL soil benchmark developed by TAI (Table 6-1, Table 4-6) is based on plant growth. The single location with the SSL exceedance had an HQ of 2.5 (Table 6-2).
  - Spatial distribution: The exceedance occurred at location SA9-2C, where chromium is 1 of 10 metals exceeding the SSL soil benchmark. This location is close to the UCR near the U.S.-Canada border (Map 6-4).
  - Background: Chromium concentrations exceeded background in multiple locations, including the 2012 Ecology Upland Soil Study location with HQ ≥ 1 (Table 6-3, Table 2-8).
  - Specific uncertainty: The chromium SSL was developed by TAI using a method similar to that of EPA Eco-SSL guidance (EPA, 2005b). Toxicity data are available for a single species; thus, the SSL has more uncertainty than Eco-SSLs developed for COPCs with significantly more data (such as zinc). Bioavailability of chromium in the soil in the studies used to develop the soil benchmark was high (Table 6-4), but bioavailability estimated for the sample in this location was low (1 on a scale of 0 to 4, with 4 being greatest) (Table 6-2).
  - Conclusion: Because chromium exceeds the SSL in only one location, the soil benchmark may overestimate the potential for effects, and many other metals also exceed their soil benchmarks in that location, chromium poses negligible risk and is not retained as a COC for plants.

- Cobalt
  - HQs: HQs for BAB soil benchmarks exceed 1 in a single location (0.9 percent of locations) from the 2012 Ecology Upland Soil Study. HQs for all locations in the 2014 UCR Upland Soil and 2015 Bossburg studies are less than 1.
  - Nature and severity: The BAB soil benchmark developed by ARCHE Consulting (Table 6-1, Table 4-6) is based on plant yield (roots and shoots). The single location with the BAB exceedance had an HQ of 1.1 (Table 6-2).
  - Spatial distribution: The exceedance occurred at location SA7-8C, where cobalt is 1 of 4 metals exceeding the BAB soil benchmark. This location is close to the UCR near the U.S.-Canada border (Map 6-5).
  - Background: There are four locations from the 2012 Ecology Upland Soil Study that exceeded background, including the single location with HQ ≥ 1 (Table 6-3, Table 2-8; Map 6-5). This also indicates that concentrations in these four locations are greater than the concentrations in the other two soil studies.
  - Specific uncertainty: While uncertainty in the BAB could underestimate the number of HQs ≥ 1 (Table 6-6), all locations with HQs ≥ 1 exceed the soil benchmark for at least one additional COPC (Table 6-2). No additional specific uncertainties affect risk conclusions (Table 4-6, Table 6-4).
  - Conclusion: Because cobalt exceeds the BAB in a single location and uncertainty analyses do not indicate increased risks, cobalt poses negligible risk and is not retained as a COC for plants.

- Copper
  - HQs: HQs for BABs are < 1 for all samples from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies.
  - Nature and severity: There are no exceedances of the BAB at any location, indicating adverse effects are unlikely.
  - Spatial distribution: There are no exceedances of the BAB at any location.
  - Background: There are several locations throughout the Terrestrial Study Area that exceed background (Table 6-3, Table 2-8).
  - Specific uncertainty: While uncertainty in the BAB could result in one HQ ≥ 1 (Table 6-6), all locations with HQs ≥ 1 exceed the soil benchmark for at least one additional COPC (Table 6-2). No additional specific uncertainties affect risk conclusions (Table 4-6, Table 6-4).
  - Conclusion: Because there are no exceedances of the BAB at any locations and uncertainty analyses do not indicate increased risks, copper poses negligible risk and is not retained as a COC for plants.
- Iron
  - HQs: As described for aluminum, the screening threshold for iron is based on locations having pH less than a threshold value of 5.0. Only in locations with sufficiently low pH would iron result in the potential for adverse effects on plants. One location in the 2012 Ecology Upland Soil Study and five locations in the 2014 UCR Upland Soil Study had sufficiently low pH for effects from iron to potentially occur.
  - Nature and severity: Where pH is sufficiently low, excess iron exposure may cause adverse effects, such as reduced growth, discoloration, or spotting (Table 6-1, Table 4-6), but effects vary considerably among species.
  - Spatial distribution: Locations with pH less than 5.0 occur sporadically through the Terrestrial Study Area (Map 6-6).
  - Background: Iron is below the BTV in all locations, except eight 2012 Ecology Upland Soil Study samples dispersed throughout the northern portion of the site (Table 6-3, Table 2-8; Map 6-6).
  - Specific uncertainty: EPA (2005j) notes that the nature and severity of effects of iron vary considerably among species.
  - Conclusion: The pH in more than 97 percent of locations is above a pH of 5.0, the threshold below which iron would be of concern. Iron concentrations at locations with pH < 5.0 are all less than the BTV. Risk to plants due to exposure to iron is negligible, and iron is not carried forward as a COC for plants.
- Lead
  - HQs: HQs for BABs exceeded 1 in 42, 18, and 3 locations (40 percent, 13 percent, and 50 percent of locations) from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies, respectively (Tables 6-1; Figure 6-1c).
  - Nature and severity: The BAB is based on toxicity studies measuring growth, photosynthesis, and yield (Table 6-1, Table 4-6). For simplicity of discussion, these endpoints are grouped here as measures of growth. BAB HQs ≥ 1 indicate the potential for > 20 percent reduced plant growth in more than 5 percent of species (PAF > 5 percent) in each location where exceedances occur.

- Spatial distribution: 2012 Ecology Upland Soil Study samples exceeding the BAB occur throughout the area sampled. 2014 UCR Upland Soil Study samples exceeding the BTV and BAB are close to the UCR throughout the length of the Terrestrial Study Area (Map 6-7). In the Bossburg Study data set, all three exceedances occur in a single cluster on the east side of the UCR.
- Background: Lead concentrations for all locations in the three data sets exceed the lead BTV of 27.2 mg/kg (Table 6-3, Table 2-8; Figure 6-1c).
- Specific uncertainty: Uncertainty in the EPC may underestimate the number of DUs with HQs ≥ 1 (Table 6-5) and uncertainty in the BAB could underestimate (Table 6-6) or overestimate the potential for adverse effects. No additional specific uncertainties associated with the lead exposure to plants, or the BAB are identified (Section 6.3; Table 6-4, Table 4-6).
- Conclusion: There is widespread risk of reduced plant growth along the UCR throughout the length of the Terrestrial Study Area due to exposure to lead concentrations that exceed the BAB and background. Lead poses unacceptable risk and is carried forward into the RI as a COC for plants.

- Manganese
  - HQs:
    - HQs for the Eco-SSL exceeded 1 in 97 percent, 100 percent, and 100 percent of locations in the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies, respectively (Tables 6-1 and 6-2; Figure 6-2h). A BAB is not available.
    - HQs for the Eco-SSL exceeded 5 in 47 percent and 34 percent of locations in the 2012 Ecology Upland Soil and 2014 UCR Upland Soil studies, respectively (Tables 6-1 and 6-2; Figure 6-2h).
  - Nature and severity: The manganese Eco-SSL is based on the geometric mean of growth MATCs for three species of plants (Table 4-6). Although HQs are not quantitative indicators of the magnitude of effects, having 12 locations with HQs greater than 10 suggests the potential for a significant reduction in plant growth.
  - Spatial distribution: Exceedances of the Eco-SSL occur throughout the Terrestrial Study Area, but background exceedances are limited and appear to be clustered and are more frequent at locations further from (and/or higher elevation than) the UCR (Map 6-8). At many locations with HQs ≥ 1, manganese is the only COPC exceeding a benchmark (Table 6-2).
  - Background: The manganese BTV of 1,250 mg/kg is more than 5-fold greater than the Eco-SSL benchmark of 220 mg/kg, so while more than 97 percent of locations in the three data sets exceeded the Eco-SSL, only 38 and 32 locations (36 percent and 23 percent) in the 2012 Ecology Upland Soil and 2014 UCR Upland Soil studies exceeded background (Table 6-3, Table 2-8; Map 6-8). No 2015 Bossburg Study samples exceeded the BTV.
  - Specific uncertainty: The Eco-SSL was derived from studies conducted under medium and high bioavailability conditions (bioavailability scores of 2 or 3) (Appendix D, Table D-3). At locations that were greater than the BTV, cationic bioavailability scores are medium or high (2 or 3) in 17 of 38 (45 percent) of 2012 Ecology Upland Soil Study locations, and 2 of 32 (6 percent) of 2014 UCR Upland Soil Study locations (Table 6-2).
  - Conclusion: Manganese exposure presents risk of reduced plant growth throughout the Terrestrial Study Area and exceeds background in about 70 locations in total. Manganese poses unacceptable risk to plants and is retained as a COC for evaluation in the RI.

- Molybdenum
  - HQs: HQs for the BAB are less than 1 for all samples from the 2014 UCR Upland Soil Study. The 2012 Ecology Upland Soil and 2015 Bossburg studies did not report molybdenum data.
  - Nature and severity: There are no exceedances of the BAB at any location, indicating adverse effects are unlikely.
  - Spatial distribution: There are no exceedances of the BAB at any location.
  - Background: There are 37 locations from the 2014 UCR Upland Soil Study that exceeded background (Table 6-2, Table 2-8).
  - Specific uncertainty: While uncertainty in the BAB could result in four HQs ≥ 1 (Table 6-6), all locations with HQs ≥ 1 exceed the soil benchmark for at least one additional COPC (Table 6-2). No specific uncertainties affect risk conclusions (Table 4-6, Table 6-4).
  - Conclusion: Because there are no exceedances of the BAB at any locations, molybdenum poses negligible risk and is not retained as a COC for plants.
- Nickel
  - HQs: HQs for BABs exceeded 1 in 3 of the 2012 Ecology Upland Soil Study locations and no locations from the 2014 UCR Upland Soil or 2015 Bossburg studies (Table 6-1; Figure 6-1c).
  - Nature and severity: The BAB is based on toxicity studies measuring yield of seeds, roots, and shoots in 11 species of plants. BAB HQs ≥ 1 indicate the potential for > 20 percent reduced plant growth in more than 5 percent of species (PAF > 5 percent) in each location where exceedances occur with a maximum PAF of 39 percent of species adversely affected.
  - Spatial distribution: Exceedances occur in three dispersed locations in the 2012 Ecology Upland Soil Study, where nickel is one of four to six metals exceeding screening values (Map 6-9; Table 6-2).
  - Background: The nickel concentration exceeded the BTV of 35 mg/kg in all three 2012 Ecology Upland Soil Study locations with BAB HQs ≥ 1.
  - Specific uncertainty: While uncertainty in the EPC may result in two 2014 UCR Upland Soil Study DUs with HQs ≥ 1 (Table 6-5) and uncertainty in the BAB could result in several locations with HQs ≥ 1 (Table 6-6), all locations with HQ ≥ 1 exceed the soil benchmark for at least one additional COPC (Table 6-2). No additional specific uncertainties associated with the nickel exposure to plants, or the nickel soil benchmark are identified (Section 6.3; Table 6-4, Table 4-6).
  - Conclusions: Although nickel has HQs ≥ 1 in three locations and exceeds background in those locations within the 2012 Ecology Upland Soil Study area, the limited spatial extent of risk from nickel and its co-occurrence with other metals support the conclusion that additional risk from nickel is negligible so nickel is not retained as a COC for plants.
- Selenium
  - HQs: HQs for the Eco-SSL exceeded 1 in 13, 21, and 1 location (12 percent, 15 percent, and 17 percent of samples) in the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies, respectively (Tables 6-1; Figure 6-2k). BABs are not available for selenium.
  - Nature and severity: The Eco-SSL soil benchmark is based on the geometric mean of EC20s and MATCs for measures of growth in six species of plants (Table 6-1, Table 4-6). HQs from the 2012 Ecology Upland Soil Study ranged up to 10. HQs from the 2014 UCR Upland Soil and 2015 Bossburg studies ranged up to 6.4 (Table 6-3; Figure 6-2k).

- Spatial distribution: Most locations with HQs ≥ 1 occur on the east side of the UCR with exceedances along the length of the Terrestrial Study Area (Map 6-10).
- Background: All locations in the three studies exceed the BTV of 0.098 mg/kg (Table 6-3, Table 2-8; Map 6-10).
- Specific uncertainty: In the 2012 Ecology Upland Soil Study, nearly all of the results with HQ = 0.96 are reported as nondetects (Table 6-2), and most results with HQs ≥ 1 are reported as detects. In addition, the distribution of selenium HQs (and hence concentrations) varies among the three soil studies with the 2012 Ecology Upland Soil Study having the greatest magnitude of exceedances, consistent with the small spatial scale of sampling in this study versus the averaging over larger areas implicit in ICS sampling employed in the 2014 UCR Upland Soil Study and the 2015 Bossburg Study (Figure 6-2k). Uncertainty in the EPC may underestimate the number of 2014 UCR Upland Soil Study DUs with HQs ≥ 1 (Table 6-5).
- Conclusions: Selenium HQs exceed 1, presenting a risk of reduced growth to plants in 12 percent to 17 percent of locations. Selenium concentrations exceed background throughout the Terrestrial Study Area. Selenium poses unacceptable risk to plants and is retained as a COC for evaluation in the RI.

▪ Thallium
  - HQs: HQs for SSLs are less than 1 for all samples from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies.
  - Nature and severity: There are no exceedances of the SSL at any location indicating adverse effects are unlikely.
  - Spatial distribution: There are no exceedances of the SSL at any location.
  - Background: There are 16 locations from the 2012 Ecology Upland Soil study that exceeded background (Table 6-2, Table 2-8).
  - Specific uncertainty: No specific uncertainties affect risk conclusions (Table 4-6, Table 6-4).
  - Conclusion: Because there are no exceedances of the SSL at any locations and uncertainty analyses do not indicate increased risks, thallium poses negligible risk and is not retained as a COC for plants.

▪ Zinc
  - HQs: HQs for BABs exceeded 1 in 73, 117, and 4 locations (69 percent, 83 percent, and 67 percent of locations) from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies, respectively (Tables 6-1; Figure 6-1f).
  - Nature and severity: The soil benchmarks for zinc are based on toxicity studies measuring growth, first bloom, and yield of seeds, roots, and shoots (Table 6-1, Table 4-6). As with lead, these endpoints are grouped here as measures of growth. BAB HQs greater than 1 in 117 of the 2014 UCR Upland Soil Study locations indicate the potential for reduced plant growth in more than 5 percent of species (PAF > 5 percent) in each location. Median PAFs for all three soil studies indicate ≥ 12 percent of species may have ≥ 20 percent reduction in growth in more than half of the locations throughout the Terrestrial Study Area (Table 6-1).
  - Spatial distribution: Most locations exceeded soil benchmarks for zinc. The few locations that did not exceed background are scattered through the Terrestrial Study Area (Map 6-11).

- Background: There are only ten locations across the three data sets that did not exceed background (Table 2-8, Table 6-3). The 2014 UCR Upland Soil locations are in the far northeastern portion of the Terrestrial Study Area, the 2012 Ecology Upland Soil locations are in the northern portion of the site, and the 2015 Bossburg studies locations are in the southwestern portion of the site (Map 6-11).
- Specific uncertainty: Uncertainty in the EPC may underestimate the number of 2014 UCR Upland Soil Study DUs with HQs ≥ 1 (Table 6-5). No specific additional uncertainties associated with the zinc exposure to plants or the soil benchmarks are identified (Section 6.3; Table 6-4, Table 4-6).
- Conclusion: There is widespread risk of reduced plant growth throughout the Terrestrial Study Area due to exposure to zinc concentrations that exceed the BAB and background. Zinc poses unacceptable risk and is carried forward into the RI as a COC for plants.

6.4.2 Multiple Metals

The preceding discussion focuses on individual COPCs, with limited discussion of multiple metals with HQs ≥ 1. This section explores locations and portions of the site having the greatest potential for the combined effects of multiple metals. Table 6-2 identifies those sample locations with HQs ≥ 1, listing all of the COPCs exceeding the highest tiered soil benchmarks (BAB if available, otherwise Eco-SSL or SSL) at each location.

Maps 6-12a through 6-12d combine results for the plant COCs, zinc, lead, arsenic, manganese, and selenium, with results for each COC shown in a cluster of five symbols at each location. The HQ and BTV comparisons are indicated with color and open/closed symbols, respectively, as described in Section 6.1 for Maps 6-1 through 6-11. On Maps 6-12a through 6-12d, the five symbols are arranged in two rows, with zinc and lead PAFs represented by hexagons in the top row, and arsenic, manganese, and selenium Eco-SSLs/SSLs represented as circles in the bottom row.

As illustrated in Table 6-2 and on Maps 6-12a through 6-12d, the only COCs that exceed benchmarks in a given location by themselves are selenium and manganese, whereas at most locations where exceedances occur, multiple COCs exceed benchmarks. Zinc exceedances are most widespread and generally of greater severity than other COPCs, with PAFs ≥ 20 percent (> 20 percent of species affected) occurring more frequently near the northern part of the Terrestrial Study Area along the river valley and adjacent valley terraces (Map 6-12). Though lead exceedances are less widespread, the spatial pattern of exceedances is similar. Manganese exceedances appear to be distributed differently across the landscape with concentrations ≤ BTV along the river valley. However, manganese HQs are ≥ 1 at all locations where zinc PAFs are ≥ 20, and at locations where manganese is > BTV. Zinc PAFs range from 21 to 77 indicating manganese risks co-occur with zinc risks (Table 6-2). Similarly, of the 14 locations where the lead PAF is ≥ 20, manganese and zinc both also exceed their respective benchmarks at all of them. Arsenic and selenium also exceed their respective SSLs at many locations (Table 6-2).

As described in Section 6.3.4.1, the simultaneous exposure of plants to elevated concentrations of multiple metals results in complex interactions and the effects are difficult to predict. Additivity of effects can be assumed when the same mode of action is affected, such as impacts on photosynthesis. For the purposes of this risk assessment, it is assumed that locations with multiple COPCs exceeding benchmarks generally pose a greater risk to plants than those locations with fewer exceedances and that risk at a specific location is at least as great as that associated with the COPC with the highest PAF or HQ.

6.5 Terrestrial Plant EAE Risk Conclusions

This Upland BERA evaluates the following risk question related to terrestrial plants within the Terrestrial Study Area:

- Are the concentrations of COPCs in soils in the Terrestrial Study Area greater than soil screening benchmarks for the survival, growth, and reproduction of terrestrial plants such that adverse effects to the local community are expected?
- Are the concentrations of COPCs in soils in the Terrestrial Study Area greater than bioavailability-adjusted soil benchmarks for the survival, growth, and reproduction of terrestrial plants such that adverse effects to the local community are expected?

Based on the analyses discussed in the risk characterization, zinc, lead, and manganese are COCs that present the greatest and most widespread risk to plants in the Terrestrial Study Area, based on widespread exceedances of BABs (zinc and lead) and the SSL (manganese) with the most likely effect being reduced growth. In addition, arsenic and selenium, based on exceedances of SSLs, are COCs that contribute a lesser but non-negligible risk of reduced growth to plants in the Terrestrial Study Area.

Uncertainty in the underlying exposure and effects analyses indicate that risks from these COCs may be somewhat under- or overestimated. Risk estimates for lead and zinc are based on robust toxicity data sets and incorporate site-specific bioavailability information into the risk estimates, which increases the reliability and relevance of the analysis resulting in a moderate degree of confidence in the risk prediction for these COCs. Because there are no site-specific measures of effects, risk predictions remain somewhat uncertain.

Because the SSLs for manganese, arsenic, and selenium do not explicitly evaluate the site-specific bioavailability of these COCs, there is somewhat greater uncertainty in the extent and magnitude of associated risks. The SSLs underlying the risk estimates for these COCs are based on more limited toxicity data and because they are derived for screening purposes are more likely to overestimate than underestimate risks. There is a low- to moderate degree of confidence in these risk estimates.

The remaining COPCs for plants, aluminum, antimony, barium, chromium, cobalt, copper, iron, molybdenum, nickel, and thallium, present negligible risk and are not carried forward as COCs. With the exception of nickel, uncertainty is not likely to underestimate risk and there is a high degree of confidence that these COPCs pose negligible risk. Uncertainty in exposure and effects analyses may underestimate risk from nickel resulting in a moderate degree of confidence that nickel poses negligible risk. Because nickel exceedances occur at locations where COCs exceed soil benchmarks, any potential adverse effects from nickel will likely be addressed by addressing the identified COCs.

7. Risk Characterization for Soil Invertebrates

This risk characterization section assesses whether exposure of soil invertebrates to COPCs in soils may pose an unacceptable risk in the Terrestrial Study Area. The problem formulation for soil invertebrates, including an ecological overview, is presented in Section 2. The methods for exposure and effects assessment and analysis are presented in Section 4, and the approach for integrating this information to characterize risks is presented in Section 5.

The risk characterization is structured as follows:

- Risk Estimation: presents the results of the risk analysis.
- Comparison to Background Soil Concentrations: compares site soils to BTVs for each COPC.
- Uncertainty Analysis: analyzes and discusses the major areas of uncertainty.
- Risk Description: describes the likely nature, intensity, and spatial scale of effects.
- Risk Conclusions: summarizes the risks and overall degree of confidence in the risk estimates.

The soil invertebrate community is the ecological entity for the soil invertebrates EAE. The EAE-specific risk questions are as follows:

- **SSL benchmarks.** Are the concentrations of COPCs in soils in the Terrestrial Study Area greater than soil screening benchmarks for the survival, growth, and reproduction of soil invertebrates, such that adverse effects to the local community are expected?
- **BABs.** Are the concentrations of COPCs in soils in the Terrestrial Study Area greater than bioavailability-adjusted soil benchmarks for the survival, growth, and reproduction of soil invertebrates, such that adverse effects to the local community are expected?

The two LOEs used for characterizing risk to the soil invertebrate EAE are as follows:

- LOE 1: COPC concentrations in soil compared to bulk SSL benchmarks (Eco-SSLs and SSLs). All COPCs have SSL benchmarks available (Table 4-5).
- LOE 2: COPC concentrations in soil compared to BABs. COPCs for soil invertebrates that have BABs are cobalt, copper, molybdenum, and zinc (Table 4-5). For COPCs with BABs, in addition to the HQ, the potentially affected fraction (PAF) of invertebrate species is also calculated from the SSD of soil invertebrate EC20s. The PAF estimates the fraction of the soil invertebrate community adversely affected at a given sample location.

These two types of benchmark comparisons are not independent LOEs; the BABs are a refinement of the SSL benchmarks using additional soil chemistry data to make inferences about the bioavailability of COPCs in each sample. For COPCs and/or sample locations having sufficient data, benchmarks are adjusted for bioavailability and the adjusted benchmarks and PAF are given precedence over the SSLs for the risk characterization. For those COPCs that have both BABs and SSLs, HQs are calculated for both benchmarks, but the risk characterization describes risks for each sample location/COPC combination based only on the BAB.

7.1 Risk Estimation

This section presents the risk analyses for the soil invertebrate community EAE and summarizes the results. COPCs for invertebrates are aluminum, arsenic, barium, chromium, cobalt, copper, iron, manganese, molybdenum, silver, thallium, vanadium, and zinc (Table 2-7). For each LOE, HQs are calculated for every sample and COPC for which there is a benchmark. HQs are interpreted as follows:

- **HQ < 1.** The estimated exposure is unlikely to cause an adverse effect to the EAE at that sample location. Risk is negligible.
- **HQ ≥ 1.** Unacceptable risk cannot be ruled out. The nature and magnitude of potential adverse effects associated with the estimated exposure at each applicable location is further evaluated and described.

Data sets used for the invertebrate risk characterization are the 2012 Ecology Upland Soil Study, the 2014 UCR Upland Soil Study, and the 2015 Bossburg Study. Samples from each study are compared to the SSL and BAB benchmarks. Because percent clay and eCEC, two soil parameters necessary to calculate BABs, were not measured as part of the 2012 Ecology Upland Soil Study, sample-specific values were assumed to be represented by the minimum percent clay and eCEC from 2014 UCR Upland Soil Study DUs that overlap respective 2012 Ecology Upland Soil Study subareas where samples were collected. The analyses performed to estimate percent clay and eCEC for 2012 Ecology Upland Soil Study samples are presented in Appendix F and discussed in the uncertainty analysis.

For each data set, the number and percentage of samples with HQs ≥ 1 and ≥ 5 for the BAB and SSL, and the median and maximum PAFs are presented in Table 7-1. For those COPCs evaluated using BABs, Figures 7-1a through 7-1d shows boxplots and empirical cumulative distribution functions of concentrations relative to PAFs. For those COPCs evaluated using only the Eco-SSL or SSL figures showing boxplots and empirical cumulative distribution functions of concentrations relative to SSL benchmarks are provided on Figures 7-2a through 7-2k. All HQ and PAF calculations are presented in Appendix F. For those COPCs with a BAB, BAB HQs and PAFs are considered in risk estimation and SSL HQs are not discussed further.

For each sample in each data set, the COPCs exceeding the BAB benchmark if available, or the Eco-SSL or SSL if not, are presented in Table 7-2. Table 7-2 also provides HQs, PAFs, and BTV comparisons for each sample as available.

For COPCs with soil concentrations exceeding the BAB if available, or the Eco-SSL or SSL if not, Maps 7-1 through 7-7 show sample locations where exceedances occur, as well as those where concentrations result in an HQ ≥ 5 or PAF ≥ 20 and whether concentrations are greater than or less than the BTV. The colors green, yellow, and blue indicate the HQ range categorically at each location. For aluminum and iron, symbols are based on pH range rather than HQ. The closed symbols indicate concentrations > BTV, while open symbols represent concentrations ≤ BTV.

The results in Table 7-1 and on Figure series 7-1 and 7-2 are as follows:

- **Zinc** BAB HQs are ≥ 1 in many samples from the 2012 Ecology Upland Soil Study, the 2014 UCR Upland Soil Study, and the 2015 Bossburg Study (Table 7-1; Map 7-7). Over the three soil studies, 27 to 44 percent of results exceed the BAB, which is equivalent to a 5 percent PAF, with the maximum PAF in the three data sets ranging from 17 percent (for the 2015 Bossburg Study) to 91 percent (for the 2012 Ecology Upland Soil Study) of species adversely affected.
- **Barium, chromium, and manganese** SSL benchmark HQs are ≥ 1 for numerous samples in the 2012 Ecology Upland Soil Study and 2014 UCR Upland Soil Study data sets; in contrast, none of these three COPCs had SSL benchmark HQs ≥ 1 for any sample from the 2015 Bossburg Study data set (Figures 7-2b, 7-2c, and 7-2f, respectively). BABs are not available for these COPCs. A significant percentage (42 percent) of barium sample results from the 2012 Ecology and 2014 UCR Upland Soil studies exceed the barium SSL (Table 7-1; Figure 72b; Map 7-2). Relatively few (≤ 5.7 percent) chromium results exceed the SSL (Table 7-1; Figure 7-2c; Map 7-3). Nearly all

(≥ 89 percent) manganese results from the 2012 Ecology and 2014 UCR Upland Soil studies exceed the SSL (Table 7-1; Figure 7-2f; Map 7-3).

- **Copper** BAB HQs are ≥ 1 in only two samples, one each from the 2012 Ecology Upland Soil Study and the 2015 Bossburg Study, with PAFs of 5.4 and 10 percent, respectively (Table 7-1; Figure 7-1b; Map 7-4).
- **Aluminum and iron** Eco-SSLs are based on pH; for each COPC, there are some samples in which pH is below the threshold, suggesting the potential for effects from both metals on soil invertebrates (Table 7-1).
- **Arsenic, cobalt, molybdenum, silver, thallium, and vanadium** results do not exceed BABs nor SSLs (Table 7-1). Note that molybdenum data are only available from the 2014 UCR Upland Soil Study; the 2012 Ecology Upland Soil Study and the 2015 Bossburg Study samples were not analyzed for molybdenum.
- Among the 13 COPCs evaluated for soil invertebrates, the greatest number of HQs ≥1 and highest HQs are for manganese and zinc.

7.2 Comparison to Background

This section presents a comparison of soil concentrations measured in each of the three data sets to the BTVs shown in Table 2-8.

COPCs that exceed background and exceed the soil benchmark associated with the highest LOE evaluated (BAB if available, otherwise the SSL) in at least one sample include: barium, chromium, copper, manganese, and zinc (Table 7-3).

Barium, manganese, and zinc have the greatest number and proportion of results with exceedances of soil benchmarks and BTVs.

For barium, 42 percent of the 2012 Ecology and 2014 UCR Upland Soil studies sample results exceed the SSL (Table 7-1), with approximately two-thirds of these SSL exceeding samples also exceeding the BTV (Table 7-3; Figure 7-2b). All Bossburg Study samples had concentrations < SSL, so no BTV comparisons were performed.

For manganese, while ≥ 89 percent of the 2012 Ecology and 2014 UCR Upland soil studies samples had concentrations ≥ Eco-SSL (Table 7-1), more than half of the Eco-SSL exceeding results are below the BTV (Table 7-3; Figure 7-2f). All 2015 Bossburg Study samples had concentrations < SSL, so no BTV comparisons were performed.

For zinc, most samples have concentrations above the BTV and the BAB. Of the samples from the 2012 Ecology Upland Soil, the 2014 UCR Upland Soil, and the 2015 Bossburg studies, 44 percent, 27 percent, and 33 percent, respectively, of zinc concentrations exceed the BTV and the BAB (Table 7-3). Of the samples from the 2012 Ecology Upland Soil Study, the 2014 UCR Upland Soil Study, and the 2015 Bossburg Study, four, four, and two samples had zinc concentrations below the BTV.

The six 2012 Ecology Upland Soil Study samples that exceeded the chromium SSL also exceeded the BTV (Table 7-2). In addition, these six sample locations are not localized but rather are dispersed among four of the Ecology study areas (SA3, SA5, SA7, and SA9; Table 7-2, Map 7-3). The single 2014 UCR Upland Soil Study sample result exceeding the SSL also exceeds the BTV (Figure 7-2c).

The two copper sample results exceeding the BAB also exceed the BTV (Table 7-3; Figures 7-1b).

7.3 Uncertainty Analysis

Uncertainties associated with the risk characterization for soil invertebrates are presented in the following subsections and in Table 7-4. Key uncertainties associated with respective components of the risk assessment are also evaluated.

7.3.1 Exposure Assessment

Given the large area of the Terrestrial Study Area, the soil chemistry data set only partially characterizes the full magnitude and extent of contamination. The strengths and uncertainties associated with each of the three soil studies used to characterize exposure to soil invertebrates are the same as those discussed in Section 6.3.2 for plants.

For the 2014 UCR Upland Soil and 2015 Bossburg studies, EPCs are calculated as the reported COPC concentrations determined for each DU, which may under- or overestimate the true mean COPC concentration in the DU due to site variability and sampling error. Following the same approach described in Section 6.3.2 for plants, an uncertainty analysis for soil invertebrates was conducted by calculating HQs for each DU in the 2015 Bossburg and 2014 UCR Upland Soil Study data sets as the 95 UCL divided by the highest tiered soil benchmark (BAB if available, otherwise Eco-SSL or SSL) rather than the ICS estimated mean/soil benchmark. For samples without triplicates, the UCL was estimated as the reported COPC concentration multiplied by the 95 UCL: mean ratio of 1.15 averaged over all COPCs in all triplicates from the 2015 Bossburg and 2014 UCR Upland Soil studies (Tables 6-5 and 7-5). When calculated using the 95 UCL, the number of DUs with HQs ≥ 1.0 in the 2015 Bossburg Study data set increases for manganese from zero to one and for zinc the number of HQ ≥ 1 increases from two to three. For the 2014 UCR Upland Soil Study, the number of DUs with HQ ≥ 1 increases by 13, 3, and 20 for barium, manganese, and zinc, respectively, and the number of HQs ≥ 5 increases by 3 for manganese (Table 7-5). These calculations indicate that use of the reported concentration (ICS estimated mean) for each DU as the EPC rather than the 95 UCL underestimates the spatial extent and magnitude of risk to soil invertebrates.

7.3.2 Effects Assessment

7.3.2.1 Eco-SSLs and SSLs

General uncertainties related to the Eco-SSLs and SSLs include their unknown ability to predict site-specific effects, the relevance of the invertebrate species in the underlying toxicity tests to the invertebrate communities in the Terrestrial Study Area, and the extent to which conditions in the toxicity tests represent conditions in the Terrestrial Study Area (e.g., conditions related to bioavailability) (Table 7-4).

In addition to the general uncertainties related to Eco-SSLs and SSLs, there are uncertainties relevant to the soil benchmarks for some specific COPCs. The following COPCs have HQs ≥ 1, in at least one of the three data sets, for the soil invertebrate community EAE (Table 7-1): aluminum, barium, chromium, iron, manganese, and zinc. Additional information on these Eco-SSLs/SSLs is provided in Table 4-6, and the uncertainties, such as their reliance on soil pH as a predictor for possible toxicity (aluminum and iron) and the bioavailability conditions of soils used in toxicity tests relative to site soils (barium, chromium, manganese, and zinc) are discussed in Table 7-4.

The chromium SSL of 57 mg/kg used in this BERA may overestimate risk to soil invertebrates. Specifically, New Zealand derived a soil quality guideline (SQG) protective of soil invertebrates following methods consistent with BAB, Australian, and EU REACH SQG derivation methods (Cavanagh and Munir, 2019) as described for arsenic and plants (Section 6.3.3). Soil invertebrate

LOEC/EC30s included in the chromium SSD range from 250 mg/kg to 2,500 mg/kg, representing four species of invertebrates that were tested in aged soils with varying bioavailability conditions. LOEC/EC30 for 11 species of plants and 12 microbial processes were also included in the SSD. The added chromium ED5 for plants, invertebrates, and soil processes calculated from the SSD is 187 mg/kg. Adding this ED5 to the UCR site median background concentration of 14 mg/kg calculated from the data set used to derive the BTVs, results in an estimated site-specific ED5 SQG of 201 mg/kg. Except for one sample, SA9-2C, which has a chromium concentration of 470 mg/kg, which is more than two times higher than the next highest sample (Appendix F), all site samples are below this alternative SQG.

7.3.2.2 Bioavailability-adjusted Benchmarks

The sources of uncertainty associated with BABs are described in Table 7-4. The BAB is an HC5, which is the concentration expected to cause a 20 percent adverse effect on 5 percent of soil invertebrate species (Oorts, 2020). The BAB is calculated as the median 5th percentile of the fitted log-normal distribution of soil invertebrate toxicity data. Upper- and lower-95 percent confidence limits (UCL and LCL, respectively) on the BAB are also calculated for each COPC (Table 7-6). The ranges of confidence limits are broad because, in addition to variability about the benchmark they encompass a wide range of bioavailability conditions throughout the site. The importance of variability in the toxicity data underlying the BABs is best illustrated by the difference in the number of HQs ≥ 1 when calculated as the EPC/(LCL on BAB) rather than the EPC/BAB. As shown in Table 7-6, many more samples exceed the LCL on the BAB for each of the COPCs for which a BAB is available indicating that uncertainty in the BABs may underestimate the extent of risk from these COPCs. Though not calculated, a similar reduction in the number of HQs ≥ 1 would result from calculating HQs based on the UCL on the BABs illustrating that uncertainty in the BABs may also lead to overestimates of risk.

A detailed discussion of the uncertainties associated with the BABs is also presented in Appendix D. The soil benchmarks account for bioavailability based on regression-based equations and conservative default leaching/aging factors. The uncertainties with the soil benchmarks include their unknown ability to predict site-specific effects, limited data sets underlying the bioavailability models, the lack of site-specific invertebrate species in the underlying toxicity data, and the unknown extent to which conditions in the toxicity tests represent conditions in the field (Table 7-4).

7.3.3 Risk Characterization

7.3.3.1 Use of Hazard Quotients to Assess Risk.

Uncertainties in use of HQs for characterization of risk to soil invertebrates are the same as those described in Section 6.3.4 for plants.

7.3.3.2 Metal Interactions

There is uncertainty in the approach for assessing risks to soil invertebrates related to possible interactions among metals. As discussed in Section 1.3, this issue is one of the basic principles to be considered in assessing risks posed by metals (EPA, 2007a). The complexities associated with exposure to mixtures of metals contribute significant uncertainty to the evaluation of risks to soil invertebrates from exposure to metals. Toxicological interactions (e.g., additivity, antagonism, potentiation, or synergism) of metals mixtures affecting soil invertebrates can be influenced by both the composition and concentrations of component metals, as well as chemical reactions involving precipitation, dissolution and adsorption within the soil matrix, in turn influenced by soil properties such as pH, CEC, oxides and organic matter (Renaud et al., 2021). Mixtures of metals can have a large influence on toxicity due to additivity or synergism. Nys et al. (2017) report that in a series of experiments investigating toxicity of metal mixtures (cadmium, copper, lead, nickel, and zinc) to aquatic invertebrates (*Daphnia magna* and *Ceriodaphnia* *dubia*) in different waters and a plant species (*Hordeum vulgare*) in different soils, some mixtures of metals, each individually causing <10 percent toxic effects, yielded much larger effects (up to 66 percent) when dosed in combination. Nys et al. (2017) report that while most interactions were best described by assuming independent action among the metals, this assumption frequently underpredicted the combined toxicity due to observed synergisms. Similarly, Lock and Janssen (2002) found that assuming additivity in toxicity tests of mixtures of zinc, cadmium, copper, and lead to potworm (*Enchytraeus albidus*) typically overpredicted the observed effects of the mixtures, whereas assuming independent action sometimes underpredicted the combined effects so was judged to be underprotective of risks from metals mixtures.

It is apparent from the examples noted, that because single-chemical HQs assume independent action of the mixture of metals present at the site, they may not be protective of potential changes in toxicity caused by the presence of chemical mixtures. Table 7-2 identifies locations where multiple COCs exceed soil benchmarks protective of soil invertebrates (BAB if available, otherwise the Eco-SSL or SSL). While the specific toxic effects of these exceedances cannot be predicted without site-specific toxicity testing, locations with exceedances of multiple COPCs generally pose a greater risk to soil invertebrates than those with fewer exceedances, with higher PAFs and HQs indicating higher risks.

7.3.3.3 Metal Essentiality

The fact that some metals are essential for maintaining proper health of animals also contributes uncertainty in soil invertebrate risk estimates. There is little information available on the levels of trace metals in soil necessary for the health of the soil invertebrate community. The implications of this uncertainty on the estimation of risks to soil invertebrates is unclear.

7.3.4 Background Analysis

Uncertainties in background analyses that may affect risk characterization for soil invertebrates are the same as those described in Section 6.3.5 for plants.

7.4 Risk Description

As described in Section 5.3, the risk description integrates the findings from the risk estimation (Section 7.1), comparison to background (Section 7.2), and uncertainty (Section 7.3) sections to describe the risk to soil invertebrates resulting from COPCs in the Terrestrial Study Area. The risk description also includes discussion of the spatial scale and the nature and severity of potential adverse effects.

7.4.1 Individual COPCs

Risk descriptions for individual COPCs are summarized as follows:

- Aluminum
  - HQs: The screening threshold for aluminum is based on pH; when pH is greater than 5.5, aluminum presents negligible ecological risk. There are 17, 19, and 1 locations, respectively, in the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies, respectively with pH ≤ 5.5 (Table 7-1).
  - Nature and severity: Where pH is sufficiently low, excess aluminum exposure may cause adverse effects, such as reduced growth and reproduction (Table 7-1, Table 4-6) (EPA, 2003b).
  - Spatial distribution: Locations with pH less than 5.5 occur sporadically through the Terrestrial Study Area but appear less frequently in the southern portion of the study area (Table 7-2; Map 7-1).

- Background: Aluminum concentrations are below the BTV of 40,500 mg/kg in all locations (Table 7-3, Table 2-8).
- Specific uncertainty: EPA (2003b) notes the nature and severity of effects of aluminum vary considerably among species, and aside from bioavailability at pH less than 5.5, a specific threshold concentration for effects was not established.
- Conclusion: The pH is sufficiently low in between 10 percent and 20 percent of locations in the Terrestrial Study Area that ecological risk due to aluminum could be of concern. However, aluminum concentrations do not exceed the BTV in any locations. Therefore, aluminum poses negligible risk and is not retained as a COC for soil invertebrates.

▪ Arsenic

- HQs: HQs for the SSLs are < 1 for all samples from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies.
- Nature and severity: There are no exceedances of the SSL at any location indicating adverse effects are unlikely.
- Spatial distribution: There are no exceedances of the SSL at any location.
- Background: Because no arsenic concentrations exceed the SSL, background does not affect risk conclusions.
- Specific uncertainty: No specific uncertainties affect risk conclusions (Table 4-6, Table 7-4).
- Conclusion: Because there are no exceedances of the SSL at any locations and uncertainty analyses do not indicate increased risks, arsenic poses negligible risk and is not retained as a COC for soil invertebrates.

▪ Barium

- HQs for the Eco-SSL are ≥ 1 in 42 percent of locations from both the 2012 Ecology and 2014 UCR Upland Soil studies, and none from the 2015 Bossburg Study (Tables 7-1 and 7-2; Figure 7-2b). A BAB is not available. The HQ is ≥ 5 in one 2012 Ecology Upland Soil Study location where the HQ is 7.8 (Tables 7-1 and 7-2).
- Nature and severity: The Eco-SSL is calculated as the geometric mean of three reproduction EC20s for three species of soil invertebrates. In locations with concentrations ≥ the Eco-SSL, reduced invertebrate reproduction may occur.
- Spatial distribution: Exceedances of the barium Eco-SSL occur throughout most of the Terrestrial Study Area, with a greater frequency of exceedances in the uplands away from the river valley. No barium concentrations from the 2015 Bossburg Study area exceeded the Eco-SSL (Map 7-2). In all locations where barium exceeds the Eco-SSL and BTV, at least one additional COPC exceeds its respective benchmark though not always its BTV (Table 7-2; Maps 7-8a through 7-8d).
- Background: The BTV of 395 mg/kg is close to the Eco-SSL benchmark of 330 mg/kg, so most concentrations that exceed the BTV also exceed the Eco-SSL (Figure 7-2b).
- Specific uncertainty: Uncertainty in the EPC may underestimate the number of DUs with HQ ≥ 1 (Table 7-5). The Eco-SSL was derived from studies conducted under high bioavailability conditions (bioavailability scores of 2); whereas cationic bioavailability scores are 2 or higher occur in 27 of the 103 site locations where exceedances occur (Table 7-2), suggesting lower bioavailability where most exceedances occur.

- Conclusion: Barium exposure presents risk of reduced reproduction to soil invertebrates throughout the Terrestrial Study Area, excluding the 2015 Bossburg Study sampling area. Where barium SSL exceedances occur, they co-occur with benchmark exceedances for other metals (Table 7-2). Although uncertainty in the Eco-SSL indicates barium risk may be overestimated, barium poses unacceptable risk and is retained as a COC.

- Chromium
  - HQs: HQs for the Eco-SSL are ≥ 1 in 5.7 percent of the 2012 Ecology Upland Soil Study, < 1 percent of the 2014 UCR Upland Soil Study, and none of the 2015 Bossburg Study locations (Table 7-1, Figure 7-2c). The HQ is ≥ 5 in one 2012 Ecology Upland Soil Study location where the HQ is 8.2 (Tables 7-1 and 7-2).
  - Nature and severity: The SSL is the lower of two earthworm reproduction MATCs from two studies. At concentrations above the SSL adverse effects on invertebrate reproduction may occur.
  - Spatial distribution: Exceedances occur in six dispersed locations in the 2012 Ecology Upland Soil Study and one 2014 UCR Upland Soil Study location (Map 7-3). Chromium is only one of several metals at these locations with exceedances of their respective soil benchmark values (Table 7-2).
  - Background: The chromium concentration exceeds the BTV of 23.8 mg/kg in all seven locations with Eco-SSL HQs ≥ 1 (Table 7-3; Figure 7-2c).
  - Specific uncertainty: Uncertainty in the soil chromium SSL indicates that the SSL may overestimate the risk to soil invertebrates and suggests that adverse effects are unlikely at concentrations that occur in the Terrestrial Study Area, except possibly at one 2012 Ecology Upland Soil Study location (Section 7.3.2, Table 7-4).
  - Conclusions: Although chromium exceeds the SSL and BTV in seven locations, the limited spatial extent of chromium SSL exceedances and likelihood that the SSL overestimates the potential for adverse effects support the conclusion that risk from chromium is negligible, so chromium is not retained as a COC for soil invertebrates.

- Cobalt
  - HQs: SSL HQs are less than 1 for all samples from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies.
  - Nature and severity: There are no exceedances of the SSL at any location indicating adverse effects are unlikely.
  - Spatial distribution: There are no exceedances of the SSL at any location.
  - Background: Because no cobalt concentrations exceed the SSL, background does not affect risk conclusions.
  - Specific uncertainty: Uncertainty in the BAB may under- or overestimate the number of DUs with HQ ≥ 1 (Table 7-6). No additional specific uncertainties affect risk conclusions (Table 4-6, Table 7-4).
  - Conclusion: Because there are no exceedances of the SSL at any locations and uncertainty analyses do not indicate increased risks, cobalt poses negligible risk and is not retained as a COC for soil invertebrates.

- Copper
  - BAB HQs are ≥ 1 in only two samples, one each from the 2012 Ecology Upland Soil Study and the 2015 Bossburg Study with PAFs of 5.4 and 10 percent, respectively (Table 7-1; Figure 7-1b). No 2014 UCR Upland Soil Study samples exceed the BAB.
  - Nature and severity: The BAB is calculated as the 5th percentile of an SSD of survival, growth, reproduction, and litter breakdown EC20s for 14 species of invertebrates. In locations where exceedances occur, 5 to 10 percent of species may be adversely affected.
  - Spatial distribution: Exceedances of the Eco-SSL occur in only two locations. In both locations where copper exceeds the BAB, at least one additional COPC concentration exceeds its respective benchmark (Table 7-2).
  - Background: In both locations where the copper concentration exceeds the BAB, it also exceeds the BTV.
  - Specific uncertainty: Uncertainty in the BAB may under- or overestimate the number of DUs with HQs ≥ 1 (Table 7-6). No additional specific uncertainties affect risk conclusions (Table 4-6, Table 7-4).
  - Conclusion: Copper concentrations exceed the BAB in only two locations with PAFs of 5.4 and 10. In these locations, COPCs identified as COC concentrations (zinc, and in one case also barium and manganese) also exceed their respective soil benchmarks (Table 7-2). Because of the limited spatial extent of exceedances and likelihood that any risk associated with copper will be addressed by addressing these other COCs, risk from copper is negligible and copper is not retained as a COC for soil invertebrates.
- Iron
  - HQs: As described for aluminum, the screening threshold for iron is based on locations having pH less than a threshold value of 5.0. Only in locations with sufficiently low pH would iron result in the potential for adverse effects on soil invertebrates. One location in the 2012 Ecology Upland Study and five locations in the 2014 UCR Upland Soil Study had sufficiently low pH for effects from iron to potentially occur.
  - Nature and severity: Where pH is sufficiently low, excess iron exposure may cause adverse effects on soil invertebrates, but specific effects are uncertain because the Eco-SSL for soil invertebrates is based on plant toxicity data (Table 7-1, Table 4-6).
  - Spatial distribution: Locations with pH less than 5.0 occur sporadically through the Terrestrial Study Area (Map 7-5).
  - Background: Iron is below the BTV in all locations where SSL exceedances occur (Table 7-2, Map 7-5).
  - Specific uncertainty: EPA (2005j) notes that the nature and severity of effects of iron vary considerably among species. Additionally, the Eco-SSL for soil invertebrates is based on plant toxicity data.
  - Conclusion: The pH in more than 97 percent of locations is above 5.0, the threshold below which iron would be of concern. Iron concentrations are below the BTV in all locations where Eco-SSL exceedances occur. Risk to soil invertebrates due to exposure to iron is negligible, and iron is not carried forward as a COC for soil invertebrates.

- Manganese
  - HQs: HQs ≥1 based on the Eco-SSL were observed for 89 percent and 90 percent of sample locations from the 2012 Ecology and 2014 UCR Upland Soil studies, respectively (Table 7-1, Figure 7-2f). No HQs ≥ 1 were observed among the 2015 Bossburg Study samples. A BAB is not available.
  - HQs for the Eco-SSL were ≥ 5 in 11 percent and 1 percent of locations in the 2012 Ecology and 2014 UCR Upland Soil studies, respectively (Table 7-1).
  - Nature and severity: The manganese Eco-SSL is based on the geometric mean of reproduction EC20s for three species of soil invertebrates (Table 4-6). Although HQs are not quantitative indicators of the magnitude of effects, having 14 locations with HQs ≥ 5 suggests the potential for a significant reduction in soil invertebrate reproduction in some locations.
  - Spatial distribution: Exceedances of the Eco-SSL occur throughout the Terrestrial Study Area, but background exceedances are limited and appear (1) to be clustered and (2) to be more frequent at locations further from (and/or higher elevation than) the river (Map 7-6). At several locations with HQ ≥ 1, manganese is the only COPC exceeding a benchmark (Table 7-2).
  - Background: The manganese BTV of 1,250 mg/kg is nearly 3-fold greater than the Eco-SSL benchmark of 450 mg/kg, so while almost all locations in two of the three data sets exceed the Eco-SSL, only 36 percent and 23 percent in the 2012 Ecology and 2014 UCR Upland Soil studies also exceed background (Table 7-3, Table 2-8; Map 7-6). None of the 2015 Bossburg Study samples exceed the BTV.
  - Specific uncertainty: Uncertainty in the EPC may slightly underestimate the number of DUs with HQs ≥ 1 (Table 7-5). The Eco-SSL was derived from studies conducted under high bioavailability conditions (bioavailability scores of 2); whereas at locations above the BTV, cationic bioavailability scores are high (2 or higher) in 19/70 locations where Eco-SSL exceedances occur (Table 7-2) suggesting lower bioavailability where most exceedances occur.
  - Conclusion: Manganese exposure presents risk of reduced soil invertebrate reproduction throughout the Terrestrial Study Area and exceeds background in about 70 locations in total. Manganese poses unacceptable risk to soil invertebrates and is retained as a COC for evaluation in the RI.
- Molybdenum
  - HQs: HQs for the BAB are < 1 for all samples from the 2014 UCR Upland Soil Study. The 2012 Ecology Upland Soil and 2015 Bossburg studies did not report molybdenum data.
  - Nature and severity: There are no exceedances of the BAB at any location indicating adverse effects are unlikely.
  - Spatial distribution: There are no exceedances of the BAB at any location.
  - Background: Because no molybdenum concentrations exceed the SSL, background does not affect risk conclusions.
  - Specific uncertainty: No specific uncertainties affect risk conclusions (Table 4-6, Table 7-4).
  - Conclusion: Because there are no exceedances of the BAB at any locations and uncertainty analyses do not indicate increased risks, molybdenum poses negligible risk and is not retained as a COC for soil invertebrates.

- Silver
  - HQs: HQs for the SSL are < 1 for all samples from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies.
  - Nature and severity: There are no exceedances of the SSL at any location indicating adverse effects are unlikely.
  - Spatial distribution: There are no exceedances of the SSL at any location.
  - Background: Because no samples exceed the SSL, background does not affect risk conclusions.
  - Specific uncertainty: No specific uncertainties affect risk conclusions (Table 4-6, Table 7-4).
  - Conclusion: Because there are no exceedances of the SSL at any locations and uncertainty analyses do not indicate increased risks, silver poses negligible risk and is not retained as a COC for soil invertebrates.
- Thallium
  - HQs: HQs for the SSL are < 1 for all samples from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies.
  - Nature and severity: There are no exceedances of the SSL at any location indicating adverse effects are unlikely.
  - Spatial distribution: There are no exceedances of the SSL at any location.
  - Background: Because no samples exceed the SSL, background does not affect risk conclusions.
  - Specific uncertainty: No specific uncertainties affect risk conclusions (Table 4-6, Table 7-4).
  - Conclusion: Because there are no exceedances of the SSL at any locations and uncertainty analyses do not indicate increased risks, thallium poses negligible risk and is not retained as a COC for soil invertebrates.
- Vanadium
  - HQs: HQs for the SSL are < 1 for all samples from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies.
  - Nature and severity: There are no exceedances of the SSL at any location indicating adverse effects are unlikely.
  - Spatial distribution: There are no exceedances of the SSL at any location.
  - Background: Because no samples exceed the SSL, background does not affect risk conclusions.
  - Specific uncertainty: No specific uncertainties affect risk conclusions (Table 4-6, Table 7-4).
  - Conclusion: Because there are no exceedances of the SSL at any locations and uncertainty analyses do not indicate increased risks, vanadium poses negligible risk and is not retained as a COC for soil invertebrates.

- Zinc
  - HQs: HQs for BABs are ≥ 1 in 44 percent, 27 percent, and 33 percent of locations from the 2012 Ecology Upland Soil, 2014 UCR Upland Soil, and 2015 Bossburg studies, respectively (Tables 7-1, Figure 7-1d).
  - Nature and severity: The zinc BAB is based on 17 toxicity studies for 9 soil invertebrate species measuring growth and reproduction endpoints (Table 4-6, Table 7-4). PAFs ≥ 20 occur in 26 and 5 of the 2012 Ecology Upland Soil and 2014 UCR Upland Soil Study locations, respectively, indicating ≥ 20 percent of species may experience adverse effects on growth and reproduction in several locations throughout the Terrestrial Study Area (Table 7-2).
  - Spatial distribution: Many locations throughout the Terrestrial Study Area exceeded soil benchmarks for zinc (Map 7-7).
  - Background: There are only ten locations across the three data sets that did not exceed background, all of which are below the BAB (Tables 2-8 and 7-2; Figure 7-1d).
  - Specific uncertainty: Uncertainty in the EPC may underestimate the number of 2014 UCR Upland Soil Study DUs with an HQ ≥ 1 (Table 7-5). No additional specific uncertainties associated with soil invertebrate exposure to zinc, or the zinc BAB affect risk conclusions (Section 7.3; Table 4-6, Table 7-4).
  - Conclusion: There is widespread risk of reduced soil invertebrate growth and reproduction throughout the Terrestrial Study Area due to exposure to zinc. Zinc poses unacceptable risk and is retained as a COC for evaluation in the RI.

7.4.2 Multiple Metals

The preceding discussion focuses on individual COPCs, with limited discussion of multiple metals with HQs ≥ 1. This section explores locations and portions of the site having the greatest potential for the combined effects of multiple metals. Table 7-2 identifies those sample locations with HQs ≥ 1, listing all of the COPCs exceeding the highest tiered soil benchmarks (BAB if available, otherwise Eco-SSL or SSL) at each location.

Maps 7-8a through 7-8d combine results for the soil invertebrate COCs, barium, manganese, and zinc, with results for each COC shown in a cluster of three symbols at each location. The HQ and BTV comparisons are indicated with color and open/closed symbols, respectively, as described in Section 7.1 for Maps 7-1 through 7-11. On Maps 7-8a through 7-8d, the three symbols are arranged in two rows, with zinc PAFs represented by hexagons in the top row and barium and manganese Eco-SSLs represented by circles in the bottom row.

As illustrated in Tables 7-3 and on Maps 7-8a through 7-8d, where exceedances of the benchmark and BTV occur, zinc and manganese exceed their respective soil benchmarks in 7 of 87 and 6 of 70 locations by themselves, respectively, whereas at most locations where exceedances occur, all three COCs exceed benchmarks. Zinc exceedances are most widespread and generally of greater severity than other COCs, with PAFs > 20 percent (> 20 percent of species affected) occurring more frequently near the northern part of the Terrestrial Study Area along the river valley and adjacent valley terraces. Manganese exceedances appear to be distributed differently across the landscape with concentrations ≤ BTV along the river valley. Manganese HQs are ≥ 1 at all but 2 of the 31 locations where zinc PAFs are ≥ 20. Conversely, in the 70 locations where manganese is > BTV and HQs are ≥ 1, zinc PAFs are ≥ 5 in less than half of them (24 locations; Table 7-2). Of the 78 locations where barium is > BTV and HQs are ≥ 1, manganese HQs are ≥ 1 in all 78 locations and zinc HQs are ≥ 1 in 28 of the 78 locations.

As described in Section 7.3.3.1, the simultaneous exposure of soil invertebrates to elevated concentrations of multiple metals results in complex interactions, the effects are difficult to predict, and assumption of independent action may result in underestimates of the combined risks. For the purposes of this risk assessment, it is assumed that locations with multiple COCs exceeding benchmarks pose a greater risk to soil invertebrates than those locations with fewer exceedances and that risk at a specific location is at least as great as that associated with the COC with the highest PAF or HQ.

7.5 Soil Invertebrate EAE Risk Conclusions

This Upland BERA evaluates the following risk questions related to soil invertebrates within the Terrestrial Study Area:

- Are the concentrations of COPCs in soils in the Terrestrial Study Area greater than soil screening benchmarks for the survival, growth, and reproduction of soil invertebrates such that adverse effects to the local community are expected?
- Are the concentrations of COPCs in soils in the Terrestrial Study Area greater than bioavailability-adjusted soil benchmarks for the survival, growth, and reproduction of soil invertebrates such that adverse effects to the local community are expected?

Based on the analyses discussed in the risk characterization, barium, manganese, and zinc are COCs that present the greatest and most widespread risk to soil invertebrates in the Terrestrial Study Area, based on widespread exceedances of BABs (zinc) and the Eco-SSL (barium and manganese). Exposure to COCs in soils may result in reduced survival, growth, and reproduction of soil invertebrates of a sufficient magnitude and over a sufficient extent that soil invertebrate community attributes such as the abundance and diversity of soil invertebrates may be adversely affected. Uncertainty in the underlying exposure and effects analyses indicate that risks from these COCs may be somewhat under- or overestimated. Risk estimates for zinc are based on a robust toxicity data set and incorporate site-specific bioavailability information into the risk estimates, which increases the reliability and relevance of the analysis, resulting in a moderate degree of confidence in the risk prediction for this COC. Because there are no site-specific measures of effects, risk predictions remain somewhat uncertain.

Because the SSLs for barium and manganese do not explicitly evaluate the site-specific bioavailability of these COCs, there is somewhat greater uncertainty in the extent and magnitude of associated risks. The Eco-SSLs underlying the risk estimates for these COCs are based on more limited toxicity data and because they are derived for screening purposes are more likely to overestimate than underestimate risks. Accordingly, there is a low to moderate degree of confidence in these risk estimates.

The remaining COPCs for soil invertebrates, aluminum, arsenic, chromium, cobalt, copper, iron, molybdenum, silver, thallium, and vanadium pose negligible risk to soil invertebrates and are not carried forward as COCs. While uncertainty may under- or overestimate risk for all COPCs except copper, there is a high degree of confidence that these COPCs pose negligible risk. For copper, there is a moderate likelihood that exceedance of the BAB in one 2015 Bossburg Study sampling location where a PAF of 10 occurs may contribute to adverse effects at this location. Because zinc also exceeds the BAB at this location, any potential adverse effects from copper will likely be addressed by addressing risk from zinc. Thus, for copper, there is a moderate degree of confidence in the conclusion of negligible risks to the soil invertebrate community EAE.

8. Risk Characterization for Birds

This risk characterization section assesses whether exposure of birds to COPCs in soils may pose an unacceptable risk in the Terrestrial Study Area. The problem formulation for upland birds, including an ecological overview, is presented in Section 2. The methods for exposure and effects assessment and analysis are presented in Section 4, and the approach for integrating this information to characterize risks is presented in Section 5.

The risk characterization is structured as follows:

- Risk Estimation: presents the results of the risk analysis
- Comparison to Background Soil Concentrations: compares site soils to BTVs for each COPC
- Uncertainty Analysis: analyzes and discusses the major areas of uncertainty
- Risk Description: describes the likely nature, severity, and spatial scale of effects
- Risk Conclusions: summarizes the risks and overall degree of confidence in the risk estimates.

Five avian feeding guilds represent the bird EAEs with one representative species selected for each:

- Herbivores: California quail
- Invertivores: American robin
- Aerial insectivores: tree swallow
- Omnivores: black-capped chickadee
- Carnivores: American kestrel

For each EAE, the risk question is as follows:

- Do the daily doses of COPCs received by birds (represented by guilds focused on specific avian species) from consumption of the tissues of prey, plants, and soil in the Terrestrial Study Area exceed the TRVs for survival, growth, or reproduction of birds such that adverse effects to the local population are expected?

The single LOE for each representative species used for characterizing risk to the bird EAEs is as follows:

- Dietary doses of COPCs compared to TRVs and dose-response information

8.1 Risk Estimation

This section presents the risk analyses for the local bird population EAEs and summarizes the results. Data sets used for the bird risk characterization are the 2012 Ecology Upland Soil Study (Ecology, 2013), the 2014 UCR Upland Soil Study (TAI, 2015), and the 2015 Bossburg Study (TAI, 2016). COPCs for birds are aluminum, barium, cadmium, chromium, copper, iron, lead, mercury, molybdenum, selenium, vanadium, and zinc (Table 2-7). COIs without a TRV (antimony, beryllium, and thallium) are not quantitatively evaluated but are discussed in the Uncertainty section (Section 8.3).

Daily doses for each representative species are developed for every sample and COPC and are compared to the appropriate COPC-specific TRVs resulting in HQs. As described in Section 4.2.2 and Appendix E, TRVs are based on the ED20 (the dose resulting in a 20 percent reduction in response relative to the control) for three separate endpoints (growth, reproduction, and survival), when possible. HQs are interpreted in the risk characterization as follows:

- **HQ < 1.** The estimated exposure does not exceed the 20 percent effect threshold (based on the ED20, unless otherwise noted) and is unlikely to cause an adverse effect to the EAE. Risk is negligible.
- **HQ ≥ 1**. Unacceptable risk cannot be ruled out. The nature and magnitude of potential adverse effects associated with the estimated exposure at each applicable location exceeds 20 percent, based on the ED20 (unless otherwise stated), and is further evaluated and described.

Accordingly, for those COPCs with HQs ≥ 1 that have dose-response models available, sample-specific effective doses (EDx) are calculated from the models. The EDx values provide an estimate of the magnitude of effect (as a percent, x) that individuals of the local population of receptors may exhibit for the COPC dose estimated for a given soil sample. COPCs and endpoints with available dose-response models are those listed in Table 4-7 that have an ED20 indicated as the TRV type.

The number of samples from each soil study with concentrations resulting in HQs ≥ 1 for each COPC and representative species are reported in Table 8-1. For those species and COPCs with HQs ≥ 1, EDx dose-response estimates (where possible) are summarized in Table 8-2. Calculations and underlying data for HQ and EDx values are provided in Appendix F. For each COPC and receptor of concern Figures 8-1a through 8-12e show cumulative distributions of the number of samples from each soil study data set with concentrations resulting in HQs ≥ 1.

Maps for each representative species are provided for all COPCs and endpoints with HQs ≥ 1 (Maps 8-1 through 8-20). HQs for survival, growth, and reproduction endpoints are depicted using three symbols arranged in two rows, with survival HQs in the top row and growth and reproduction in the bottom row. If a dose-response model is available for a given COPC and endpoint, the respective map presents the calculated EDx value for each sample location using a square. If no dose-response model is available, the map indicates which sample locations have HQs ≥ 1 and HQs ≥ 5 using a circle. HQs < 1 are depicted as small green symbols. HQs ≥ 1 and EDx ≥ 20 are shown in larger symbols with the colors yellow and blue indicating the HQ or EDx range categorically at each location. Locations with COPC concentrations ≤ BTV are shown with an open symbol; locations with COPC concentrations > BTV are shown with solid symbols. The size of the species home range is illustrated with a red circle which could be centered at any location in the Terrestrial Study Area.

For all the representative species, barium, chromium, copper, iron, molybdenum, and vanadium have daily doses below their respective TRV (HQ < 1) for all sample locations in all three soil studies (Table 8-1). Note that molybdenum concentrations were only reported for the 2014 UCR Upland Soil Study. Aluminum, cadmium, lead, mercury, selenium, and zinc have HQs ≥ 1 for two or more species from two or all three studies and are briefly discussed as follows:

- **Aluminum.** American robin, black-capped chickadee, and American kestrel have a relatively small number of samples with concentrations resulting in HQs ≥ 1 for the growth endpoint (Table 8-1). For American robin, 14 percent of the 2012 Ecology Upland Soil Study and 1 percent of the 2014 UCR Upland Soil Study samples have concentrations resulting in estimated doses at or above the bird growth ED20 with HQs up to 32 (Tables 8-1 and 8-2; Figure 8-1b). Both the black-capped chickadee and American kestrel have HQs ≥ 1 for the growth endpoint associated with concentrations in two 2012 Ecology Upland Soil Study sample locations (Tables 8-1 and 8-2).

- **Cadmium.** American robin and black-capped chickadee have HQs ≥ 1. American robin has numerous 2012 Ecology Upland Soil Study and 2014 UCR Upland Soil Study samples with estimated doses at or above the ED20s for growth (57 percent of both the 2012 Ecology Upland Soil Study and the 2014 UCR Upland Soil Study samples), and reproduction (51 percent of the 2012 Ecology Upland Soil Study and 46 percent of the 2014 UCR Upland Soil Study locations), respectively (Tables 8-1 and 8-2; Figure 8-3b). A single 2012 Ecology Upland Soil Study sample location (1 percent of samples) results in an HQ ≥ 1 for survival. Where HQs are ≥ 1, the median magnitude of effect estimated (EDx) is ≥ 24 percent, ranging up to a maximum 52 percent reduction in growth (Table 8-2). For black-capped chickadee, 3 percent and 1 percent of the 2012 Ecology Upland Soil Study samples result in HQs ≥ 1 for the growth and reproduction endpoints, respectively, with magnitude of effects ranging up to 26 percent (Tables 8-1 and 8-2; Figure 8-3e). All other representative species and endpoints have HQs < 1 (Table 8-1).
- **Lead.** All avian receptors have at least one sample location with HQs ≥ 1 for lead (Table 8-1). The representative species with the greatest number of HQs ≥ 1 (from highest to lowest) are American robin, black-capped chickadee, and American kestrel (Table 8-1). California quail and tree swallow have four or fewer samples with concentrations resulting in HQs ≥ 1. California quail has 3 percent of the 2012 Ecology Upland Soil Study samples with concentrations resulting in HQs ≥ 1, and California quail and tree swallow both have a single sample (17 percent of samples) from the 2015 Bossburg Study resulting in HQs ≥ 1. For all species, HQs ≥ 1 occur most frequently for the reproduction, then survival, then growth endpoints. Of the sample locations with survival HQs ≥ 1, estimated doses correspond to 20 percent to 80 percent reductions in survival (Table 8-2). EDxs are not available for the reproduction or growth endpoints. Maximum reproduction and growth HQs are 12 and 2, respectively (Table 8-2).
- **Mercury.** The representative species with the greatest number of HQs ≥ 1 (from highest to lowest) are American robin, black-capped chickadee, and tree swallow (Table 8-1). California quail and American kestrel have HQs < 1 for all endpoints (Table 8-1). Of the sample locations with HQs ≥ 1, median and maximum estimated doses range from 22 percent to 36 percent reduced reproduction (ED22 to ED36) for American robin and black-capped chickadee, and 21 percent to 34 percent reduced reproduction (ED21 to ED34) for tree swallow (Table 8-2). EDxs are not available for the survival endpoint. Maximum survival HQs are 1.6 for American robin and black-capped chickadee and 1.3 for tree swallow (Table 8-2).
- **Selenium.** Black-capped chickadee, tree swallow, American robin, and American kestrel (in order of the number of exceedances) all have a relatively small number of HQs ≥ 1 for the survival, growth, and reproduction endpoints (Tables 8-1 and 8-2; Figures 8-10a through e). No samples result in HQs ≥ 1 for California quail. Of the sample locations with HQs ≥ 1, maximum estimated doses result in up to 77 percent, 74 percent, 38 percent, and 24 percent reduced reproduction (ED77 to ED24) for black-capped chickadee, tree swallow, American robin, and American kestrel, respectively (Table 8-2). EDxs are not available for the survival and growth endpoints. Maximum survival and growth HQs range from 1.0 to 6.1 (Table 8-2). No 2015 Bossburg Study samples result in HQs ≥ 1 (Table 8-1).
- **Zinc**. Black-capped chickadee, American robin, and American kestrel (in order of the number of exceedances) have HQs ≥ 1 for the reproduction and growth endpoints, whereas California quail and tree swallow have HQs < 1 for all endpoints and all samples (Table 8-1). Black-capped chickadee and American robin have a similar number of HQs ≥ 1, with exceedances of the growth and reproduction endpoint TRVs occurring in several samples (Table 8-1). Of the sample locations with reproduction HQs ≥ 1, maximum estimated doses result in up to 96 percent, 70 percent, and 21 percent reduced reproduction (ED96 to ED21) for black-capped chickadee, American robin, and American kestrel, respectively (Table 8-2). EDxs are not available for the growth endpoint. Maximum growth HQs range from 1.0 to 3.5 (Table 8-2). No 2015 Bossburg Study samples result in HQs ≥ 1 (Table 8-1).

8.2 Comparison to Background Soil Concentrations

This section presents a comparison of soil concentrations measured in each of the three data sets to the BTVs shown in Table 2-8. Table 8-3 summarizes the number of samples for each COPC from each soil study data set for each species and endpoint that have concentrations resulting in HQs ≥ 1 and are ≤ the BTV, as well as those that have concentrations resulting in HQs ≥ 1 and are > the BTV.

- COPCs that exceed the BTV and result in an HQ ≥ 1 for at least one endpoint for at least one receptor in at least one sample include: cadmium, lead, mercury, selenium, and zinc (Table 8-3). All aluminum samples are ≤ BTV.
- Mercury is the only COPC for which some samples with concentrations ≤ BTV result in HQs ≥ 1 and some samples with concentrations > BTV result in HQs ≥ 1. Receptors with mercury HQs ≥ 1 from samples > BTV include American robin, tree swallow, and black-capped chickadee for both the survival and reproduction endpoints.

8.3 Uncertainty Analysis

Uncertainties associated with the risk characterization for birds are presented here and in Table 8-4. Key uncertainties associated with the exposure assessment and effects assessment are evaluated in the following subsections.

Uncertainties in several of the parameters underlying the HQ calculations, including exposure model parameters (Equation 4-1) and TRVs, are evaluated by calculating HQs using different scenarios. For each scenario, the change in the number of sample results with HQs ≥ 1 using alternative assumptions from the number calculated using the baseline assumptions described in Section 4 is evaluated. For each scenario, using Excel Solver, a soil risk-based concentration (RBC) is calculated as the soil COPC concentration resulting in an HQ of 1.0. For each scenario RBCs are calculated assuming the maximum sample-specific RBA for each COPC. Because all exposure model parameters are linear and deterministic, and excepting RBA are not sample-specific, the number and magnitude of RBC exceedances is equivalent to the number and magnitude of HQs ≥ 1. For several scenarios, RBCs are calculated only for American robin because American robin had the greatest number of HQs ≥ 1 for all COPCs, except selenium and zinc (for which American robin had a similar number of HQs ≥ 1 as the species with most numerous HQs ≥ 1; Table 8-1).

8.3.1 Exposure Assessment

Calculation of the daily doses of COPCs that each representative bird species is exposed to throughout the site includes many individual parameters (Equation 4-1). Except for soil COPC concentrations, soil to plant bioaccumulation models, and soil RBAs, parameter values are based on literature-reported, rather than site-specific data. The selected values for each parameter have associated variability and uncertainty, which can influence exposure estimates, risk characterization, and risk conclusions. Table 8-4 describes uncertainties in the soil chemistry EPCs, bioaccumulation models, and exposure calculations, and evaluates the likelihood that they contribute to over- or underestimates of risk. As indicated in Table 8-4, it is unknown whether most identified uncertainties contribute to over- or underestimates of Site risks. In general, reasonable conservative assumptions were made to ensure that risk is not underestimated. The bioaccumulation models used to predict COPC uptake from soils to receptor diets have several associated uncertainties which are described in full in Appendix C and are summarized in Table 8-4. In general, bioaccumulation of metals from soil to plant and animal tissues is highly variable and depends on several site-specific and species-specific factors, such that the likelihood for the bioaccumulation models to over- or underpredict risks is unknown. Exposure calculations are generally based on conservative assumptions including selection of wildlife exposure factors and area use factors which contribute to conservative estimates of Site risks. For some parameters, quantitative uncertainty analyses are presented to determine whether additional information provides evidence that baseline exposure estimates result in over- or underestimates of exposure and to refine risk estimates and risk conclusions. Quantitative uncertainty analyses are presented next for RBA (the fraction of a COPC in incidentally ingested soil that is absorbed), soil chemistry EPCs, soil to earthworm bioaccumulation models, the fraction of total mercury that can be apportioned to methylmercury, and spatial aggregation of exposure across sample locations for large home range species.

8.3.1.1 Relative Bioavailability

RBA estimates vary by COPC and sample as a function of IVBA. Uncertainties associated with calculation of RBA are summarized in Table 8-4 and described more fully in Appendix E. Sensitivity of the risk calculations to variability in the RBA is evaluated to determine if differences in predicted RBAs among site samples result in a change in the number of HQs ≥ 1. As shown in Table 8-5, the number of HQs ≥ 1 calculated assuming all samples have the minimum RBA for a given COPC over all site samples is the same as when assuming all samples have the maximum site-specific RBA, except for aluminum and lead. This indicates that risk predictions are generally not sensitive to RBA within the range predicted for site-specific samples. For aluminum and lead, the difference in number of HQs ≥ 1 due to variability in the RBA does not influence risk conclusions because all site aluminum concentrations are ≤ BTV and for lead, numerous samples result in HQ ≥ 1 regardless of which RBA is selected. This indicates that uncertainty analyses calculated assuming the maximum RBA will not influence conclusions for other exposure parameters. Because further uncertainty analyses discussed for lead assumed the maximum RBA, the number of HQs ≥ 1 for lead under baseline assumptions in uncertainty analyses is slightly greater than the numbers calculated using baseline assumptions presented in Table 8-1.

8.3.1.2 Soil Chemistry Exposure Point Concentrations

Given the large area of the Terrestrial Study Area and necessary sampling limitations, the soil chemistry data set only partially characterizes the full magnitude and extent of contamination. The strengths and uncertainties associated with each of the three soil studies used to characterize exposure to birds are the same as those discussed in Section 6.3.2 for plants.

For the 2014 UCR Upland Soil Study and 2015 Bossburg studies, EPCs are calculated as the reported COPC concentrations determined for each DU, which may under- or overestimate the true mean COPC concentration in the DU due to site variability and sampling error. Following the same approach described in Section 6.3.2 for plants, an uncertainty analysis for birds is presented herein where EPCs for each DU in the 2015 Bossburg and 2014 UCR Upland Soil studies data set is calculated as the 95 UCL. The daily COPC doses for each bird receptor are calculated following the methods described in Section 4.1.1, substituting the 95 UCL for the COPC concentration in soil. HQs are calculated as the Daily Dose$_{UCL95}$ divided by the TRV for the most sensitive endpoint (survival, growth, or reproduction) for each COPC. Table 8-6 summarizes the number of HQ ≥ 1 when calculated as the Daily Dose$_{UCL95}$/TRV relative to the Daily Dose$_{mean}$/TRV. When calculated using the 95 UCL, the number of DUs with HQs ≥ 1 in the 2014 UCR Upland Soil Study data set increases for aluminum, cadmium, lead, mercury, and zinc with increases > 10 percent for aluminum and mercury and smaller increases for the other COPCs. These calculations indicate that use of the reported concentration (ICS estimated mean) for each DU as the EPC rather than the 95 UCL may somewhat underestimate the spatial extent and magnitude of risk to bird receptors of concern.

8.3.1.3 Earthworm Bioaccumulation Models

Uncertainty in the earthworm bioaccumulation models was evaluated by comparing the number of HQs ≥ 1 when calculated using the baseline selected models relative to the number of HQs ≥ 1 when COPC doses are calculated using models reported in a recent synthesis of literature-reported earthworm bioaccumulation data. The soil to earthworm models assumed in the baseline calculations are those reported in Sample et al. (1998a) (Table 4-4). The Sample et al. (1998a) models are based on a meta-analysis of soil to earthworm relationships compiled from 32 literature-reported studies. Richardson et al. (2020) report results from a recent meta-analysis of soil to earthworm bioaccumulation relationships reported in 56 literature-reported studies for several metals including the bird COPCs cadmium, chromium, copper, mercury, and zinc. Richardson et al. (2020) evaluated log-linear regressions of metals concentrations in worms relative to concentrations in co-located soils and report that regressions were significant only for mercury and cadmium. This contrasts with Sample et al. (1998a), which found significant regressions for cadmium, lead, and zinc (Table 4-4) but not for other site COPCs including mercury. Richardson et al. (2020) report that bioaccumulation factors (BAFs) significantly decreased with increasing metals concentrations for all COPCs, except lead.

Table 8-7 compares the number of American robin HQs calculated with the Richardson et al. (2020) earthworm bioaccumulation models to the number of HQs calculated using the earthworm bioaccumulation models reported in Table 4-4. The Richardson et al. (2020) reported models used for this analysis were: log-linear models for cadmium and mercury; log-linear BAFs for chromium, copper, and zinc; and the median BAF for lead.

As shown in Table 8-7, when exposure is calculated using the Richardson et al. (2020) model rather than the Sample et al. (1998a) model, no COPCs change from having all HQs < 1 to having some HQs ≥ 1 or vice versa. Use of the Richardson et al. (2020) model has the greatest influence on mercury and zinc risk predictions. For mercury, approximately twice as many HQs ≥ 1 occur when calculated using the Richardson et al. (2020) regression model rather than the Sample et al. (1998a) reported BAF. For zinc, many fewer HQs ≥ 1 (fewer than half as many) are predicted when using the Richardson et al. (2020) median BAF than when using the Sample et al. (1998a) regression model. For cadmium the number of HQs ≥ 1 decreases slightly (< 10 percent) when calculated using the Richardson et al. (2020) model, whereas for lead the number of HQ ≥ 1 increases slightly (< 10 percent) when calculated using the Richardson et al. (2020) model rather than the Sample et al. (1998a) model.

8.3.1.4 Methylmercury Apportionment

An uncertainty in the mercury HQ calculations that likely leads to overestimates of risk is that the exposure data (that is, soil and prey items) reflect total mercury concentrations, but the TRV and other toxicological effects thresholds protective of mercury risks to birds are based principally on methylmercury. To address this uncertainty, mercury HQs are calculated below considering the apportionment of total mercury to methylmercury in bird diets.

He et al. (2018) reports that the ratio of methylmercury to total mercury in different species of earthworms from different soils ranged from 2 percent to 33 percent (He et al., 2018; Zhang et al., 2009; Rieder et al., 2011 as cited in He et al., 2018). In aquatic systems, Malcata-Martins et al. (2021) report that median methylmercury: total mercury ratios for different organisms range from 25 percent to over 95 percent in data from greater than 23 different studies. Among the arthropods analyzed in a study of terrestrial invertebrates (spiders, detritivores, and herbivores) from a site in Tennessee, spiders had the highest mean methylmercury fraction at 58.1 percent. This ratio is similar to the median methylmercury: total mercury ratio reported for spiders in Malcata-Martins et al. (2021). In general, the fraction of methylmercury increases with trophic level and with soil or water methylmercury fractions (He et al., 2018; Standish, 2016; Malcata-Martins et al., 2021).

HQs for methylmercury are evaluated for the three receptors with the highest mercury EDx values, American robin, black-capped chickadee, and tree swallow. The number of HQs ≥ 1 were calculated conservatively assuming 33 percent for worms and 58.1 percent methylmercury for terrestrial arthropods and flying insects in receptor diets. When total mercury is apportioned to methylmercury, 8.5 percent of the 2012 Ecology Upland Soil Study samples and a single sample (17 percent of samples) from the 2015 Bossburg Study have concentrations resulting in HQs ≥ 1 for American robin, 18 percent of the 2012 Ecology Soil Study samples, 5.7 percent of 2014 UCR Upland Soil Study samples, and a single (17 percent of samples) from the 2015 Bossburg Study have HQs ≥ 1 for black-capped chickadee, and for tree swallow, 10 percent of the 2012 Ecology Upland Soil Study and a single sample (17 percent of samples) from the 2015 Bossburg Study have concentrations resulting in HQs ≥ 1 (Table 8-8).

8.3.1.5 Area Use Factor

Area use is assumed to be 100 percent for all receptors. This assumes that a receptor is exposed entirely within a given DU (2014 UCR Upland Soil Study and 2015 Bossburg Study) or sample location (2012 Ecology Upland Soil Study) at all times. This creates uncertainty in COPC exposure to receptors that forage over areas larger or smaller than an exposure unit (or when exposure units are small in size, as is the case for the Ecology and Bossburg Study data sets), populations that encompass multiple locations, and/or receptors that migrate such that they would be present in a given location only a portion of the year (for example, seasonally). As illustrated by the circles depicting the home range (which could be centered at any location in the Terrestrial Study Area) for each respective receptor of concern on Maps 8-1 through 8-20, most samples are located too far apart from one another for more than a single sample to fall within the home range of an individual of a given species. Thus, aggregating concentration data from adjacent sample locations together would not meaningfully change the pattern of HQs ≥ 1 or < 1 across the site.

8.3.2 Effects Assessment

8.3.2.1 Toxicity Reference Values

Uncertainties surrounding the reliability of specific TRVs to estimate ED20s are listed in Table 8-4 for COPCs with HQs ≥ 1. These COPCs are aluminum (growth), cadmium (survival, growth, and reproduction), lead (survival, growth, and reproduction), mercury (survival and reproduction), selenium (survival, growth, and reproduction), and zinc (growth and reproduction). Additional information on the reliability and relevance of these TRVs is provided in Table 4-9. As indicated in Table-8-4, in general, the ability of the TRVs to predict site-specific adverse effects is unknown. Several site-specific factors influence the bioavailability and toxicity of metals and TRVs are selected using a robust process (described in Appendix E) to result in reasonable conservative estimates of the severity of effects resulting from Site exposure. Uncertainties in selection of TRVs are described in detail in Appendix E. Quantitative uncertainty analyses are presented in the paragraphs that follow for the lead and mercury reproduction TRVs to determine whether additional information provides evidence that baseline TRVs result in over- or underestimates of the potential for adverse effects and to refine risk estimates and risk conclusions.

Lead Reproduction

The number of lead reproduction HQs ≥ 1 may be overestimated due to uncertainty in the TRV. The lead reproduction TRV of 4.7 mg/kg bw/day is based on a variable egg production endpoint in Japanese quail (TRV derived from geomean of LOAEL ≥ 20 values from three pooled data sets, with estimated effects between 21 and 59 percent). An analysis of data used for the lead Eco-SSL found that Japanese quail have both the lowest and the highest NOAELs for egg production (Sample et al., 2019). When compared to chickens, Japanese quail are both approximately 10 times more sensitive to lead and 10 times less sensitive to lead (Sample et al., 2019). Because of this variability, selection of the quail LOAEL ≥ 20 from the low end of this range for TRV derivation is likely to overestimate the potential for adverse effects from lead for birds at the site.

A further evaluation of reproduction HQs using the ED20 of 9.8 mg/kg bw/day for chickens from Sample et al. (2019), is presented in Table 8-8. When HQs are calculated using the Sample et al. (2019) reproduction ED20, all lead reproduction HQs are < 1 for California quail and tree swallow. Assuming the alternative TRV, American kestrel HQs are ≥ 1 corresponding to concentrations in two 2012 Ecology Upland Soil Study and one 2015 Bossburg Study sample (2 percent and 17 percent of samples, respectively). Assuming the alternative TRV, American robin HQs are ≥ 1 for 41 percent of the 2012 Ecology Upland Soil Study, 18 percent of the 2014 UCR Upland Soil Study, and 33 percent of the 2015 Bossburg Study samples, respectively and black-capped chickadee HQs are ≥ 1 for 8.5 percent of the 2012 Ecology Upland Soil Study and 33 percent of the 2015 Bossburg Study samples, respectively.

Mercury Reproduction

The number of mercury reproduction HQs ≥ 1 may be overestimated due to uncertainty in the TRV. The selected TRV of 0.013 mg/kg bw/day is the modeled ED20 for reduced survival of offspring of parental finches exposed to dietary methylmercury (Varian-Ramos et al., 2014). In this study effects on reproduction were not dose responsive. Offspring survival was 69 percent of control survival at the lowest dose tested (an ED31 of 0.051 mg/kg bw/day), declined to 46 percent at the second lowest dose tested (an ED54 of 0.10 mg/kg bw/day) and increased over the next two increasing doses to 62 percent survival at the highest dose (an ED38 of 0.41 mg/kg bw/day) (Appendix E, Table E2.B-10). Statistics were not presented in the paper, so no LOAEL was identified. Because of the lack of a clear dose-response relationship, confidence bounds on the modeled ED20 could not be calculated (Appendix E, Table E2.D-1).

Comparison of the selected TRV to those selected for other sites and recommended in the literature indicates that it likely overestimates the risk to birds from mercury. EPA (1995) identifies a LOAEL TRV of 0.078 mg/kg bw/day for protection of Great Lakes birds based on several reproduction and chick behavior endpoints observed in a series of long-term mallard toxicity studies (Heinz et al., 1974, 1975, 1976a, 1976b, and 1979). Both the Portland Harbor BERA (LWG, 2013) and the Hanford BERA (CH2M, 2014) selected a LOAEL of 0.064 mg/kg bw/day, based on the same Heinz et al. studies using slightly different exposure assumptions than EPA (1995). A review of dietary mercury toxicity to birds (Shore et al., 2011) identifies a dietary threshold for effects on bird reproduction based on a Heinz et al. (1976a) mallard study from the same series of studies. Shore et al. (2011) also reports that in a field study, loon productivity was estimated to be reduced by 50 percent at a fish prey concentration of 0.21 mg/kg wet-weight (ww) (Burgess and Meyer, 2008 as cited in Shore et al., 2011). A subsequent meta-analysis of mercury toxicity to loons identifies no observed adverse effects levels for effects on reproduction between 0.03 and 0.15 mg/kg prey, and lowest observed adverse effects levels between 0.16 and > 0.3 mg/kg prey, and an EC20 of 0.07 mg/kg prey (Depew et al. 2012). The productivity EC20 was derived from data presented in Burgess and Meyer (2008). Burgess and Meyer (2008) derived a dose-response regression between prey concentrations and productivity using data from more than 100 lakes located in New Brunswick and Nova Scotia, Canada, and northern Wisconsin. The EC20 from this regression is 0.07 mg/kg ww. Assuming the dietary EC20 of 0.07 mg/kg ww and the weighted average female loon body weight of 3.9 kg from Burgess and Meyer (2008) and a wild adult loon feeding rate of 960 g/d ww (Barr 1996) results in a dietary dose of 0.02 mg/kg bw/day.

A further evaluation of mercury reproduction HQs using a TRV of 0.02 mg/kg bw/day from Burgess and Meyer (2008) is presented in Table 8-8. When HQs are calculated assuming this alternative TRV, for American robin 18 percent of the 2012 Ecology Upland Soil Study, 6.4 percent of 2014 UCR Upland Soil Study, and one (17 percent) of the 2015 Bossburg Study samples result in HQs ≥ 1. For black-capped chickadee 18 percent of the 2012 Ecology Upland Soil Study, 5.7 percent of 2014 UCR Upland Soil Study, and one (17 percent) of the 2015 Bossburg Study samples result in HQs ≥ 1. For tree swallow, 10 percent of the 2012 Ecology Upland Soil Study, 1.4 percent of 2014 UCR Upland Soil Study, and one

(17 percent) of the 2015 Bossburg Study sample results, respectively result in HQs ≥ 1 (Table 8-8) when calculated using the alternative TRV.

8.3.2.2 Combined Mercury and Lead Uncertainties

Several elements of the risk analysis for mercury and lead evaluated are evaluated together to determine their contributions to under- and overestimates of risk. To evaluate the likely combined effects of these uncertainties, an uncertainty analysis is presented in Table 8-8 that combines three separate lead and four separate mercury uncertainty analyses including:

- Use of the 95 UCLs as the lead and mercury EPCs for 2014 UCR Upland Soil Study and 2015 Bossburg Study soil DUs rather than the ICS estimated means used in the baseline calculations.
- Use of alternative soil to earthworm bioaccumulation models for lead and mercury reported in Richardson et al. (2020) rather than the Sample et al. (1998a) models used in the baseline calculations.
- Use of alternative reproduction TRVs for lead and mercury rather than the TRVs used in the baseline calculations.
- The partitioning of mercury to methylmercury in bird dietary items rather than use of total mercury concentrations assumed in the baseline calculations.

For lead, when American robin HQs are calculated with the combined exposure and effects refinements, the numbers of HQs ≥ 1 decrease by close to 15 percent (13 percent to 17 percent) for each of the three soil studies from the number calculated using baseline assumptions (Table 8-8). Use of the 95 UCL and use of the alternative bioaccumulation model both increase the number of HQs ≥ 1 (by 4 percent to 14 percent), whereas the largest change is due to selection of an alternative reproduction TRV, which decreases the number of HQ ≥ 1 by 29 percent to 42 percent for the different soil studies. Because the same assumptions evaluated herein apply to the risk estimates for all species, except the alternative soil to earthworm bioaccumulation model, this analysis indicates that risks to the other representative bird species are also similarly overestimated. The number of American robin HQs ≥ 1 calculated using the combined alternative assumptions exceeds the number of HQs ≥ 1 calculated using baseline or alternative assumptions for any other species (Tables 8-8) indicating that risks calculated for American robin are protective of those for the other species.

For mercury, when American robin, black-capped chickadee, and tree swallow HQs are calculated with the combined exposure and effects refinements, 8.5 percent, 4.7 percent, and 2.8 percent of 2012 Ecology Upland Soil Study samples have concentrations resulting in HQs ≥ 1 for the three species, respectively and a single (17 percent) 2015 Bossburg Study sample, results in HQs ≥ 1 for all three species (Table 8-8). The spatial distribution of mercury HQs ≥ 1 for robin integrating the exposure and effects refinements are shown on Map 8-21, with colors and symbols as described in Section 8.1 for Maps 8-1 through 8-20.

8.3.3 Risk Characterization

Uncertainties in the risk characterization are presented in Table 8-4 including use of HQs to characterize risk, translation of HQs to population-level attributes for birds, and risk from COIs for which HQs could not be calculated because TRVs were not identified. A further analysis of COIs without TRVs is presented in Section 8.3.3.1 to determine the likelihood that they pose a risk to birds in the Terrestrial Study Area.

8.3.3.1 Chemicals of Interest Without Toxicity Reference Values

HQs could not be calculated for birds for three COIs (antimony, beryllium, and thallium) due to lack of TRVs (Section 4.2.2; Appendix E), which creates uncertainty in determining the potential for adverse effects associated with these metals. Soil sample concentrations and the COI-specific BTVs are plotted for each of the three soil study data sets to evaluate concentrations of these COIs relative to background (Figure 8-13). For antimony, all site samples from the 2014 UCR Upland Soil Study and 2015 Bossburg studies and 32 (30 percent) of the samples from the 2012 Ecology Upland Soil Study have concentrations less than the BTV. For beryllium and thallium, all site samples have concentrations less than the BTV except 16 (15 percent) of the 2012 Ecology Upland Soil Study samples which have thallium concentrations less than the BTV. Thus, for beryllium at all locations and thallium at most locations, risks are consistent with background levels. As shown on Figure 8-13 all samples with thallium concentrations less than the BTV and most samples with antimony concentrations less than the BTV result in an HQ ≥ 1 for at least one other COPC. This indicates that addressing the risks for COPCs for which HQs could be calculated will likely address any risk posed by antimony and thallium. Based on this analysis, the lack of HQ calculations for antimony, beryllium, and thallium is not expected to contribute to under- or overestimates of risk to birds.

8.3.3.2 Metal Interactions

There is uncertainty in the approach for assessing risks to birds related to possible interactions among metals. As discussed in Section 1.3, this issue is one of the basic principles to be considered in assessing risks posed by metals identified by EPA (EPA, 2007a). The complexities associated with exposure to mixtures of metals contribute significant uncertainty to the evaluation of risks to birds from exposure to metals.

The use of single-chemical HQs does not account for potential changes in toxicity caused by the presence of chemical mixtures. Interactions among metals occur by competition for binding locations on specific enzymes or on cellular receptors during the processes of absorption, excretion, or sequestration at the target site (EPA, 2007a). Depending on the type of toxicological interaction (for example, additivity, antagonism, potentiation, or synergism) and the respective exposures for the metals, the single-chemical HQ may overestimate or underestimate the potential for adverse effects.

Table 8-9 identifies locations where multiple COPCs exceed soil benchmarks protective of birds (EDx if available, otherwise the HQ). While the specific toxic effects of these exceedances cannot be predicted without site-specific toxicity testing, locations with exceedances of multiple COPCs generally pose a greater risk to birds than those with fewer exceedances, with higher EDx and HQs indicating higher risks.

8.3.3.3 Metal Essentiality

The fact that some metals are essential for maintaining proper health of birds also contributes uncertainty in the bird risk estimates. As discussed in Section 1.2, this issue is one of the basic principles to be considered in assessing risks posed by metals identified by EPA (EPA, 2007a). Of the COPCs for birds, cobalt, copper, manganese, molybdenum, selenium, nickel, and zinc are essential to animals (EPA, 2007a). Different bird species may have different nutritional requirements or optimal conditions for essential metals, and the extent to which the nutritional requirements of the test bird species are representative of the bird species that occur within the Terrestrial Study Area is uncertain. This uncertainty is unlikely to result in over- or underestimates of risk.

8.4 Risk Description

As described in Section 5.3, the risk description integrates the findings from the risk estimation (Section 8.1), comparison to background (Section 8.2), and uncertainty (Section 8.3) sections to describe the risk to birds resulting from COPCs in the Terrestrial Study Area. The risk description also includes discussion of the spatial scale and the nature and severity of potential adverse effects.

8.4.1 Individual Chemicals of Potential Concern

The following COPCs have been eliminated from further evaluation for the avian EAEs in the Terrestrial Study Area because unacceptable risks have been ruled out (for example, HQs < 1, EDx value < ED20, and/or soil concentration ≤ BTV): aluminum, barium, chromium, copper, iron, molybdenum, and vanadium.

COPCs and respective representative species retained for further evaluation include:

- Cadmium: American robin, black-capped chickadee
- Lead: California quail, American robin, tree swallow, American kestrel, and black-capped chickadee
- Mercury: American robin, tree swallow, and black-capped chickadee
- Selenium: American robin, tree swallow, American kestrel, and black-capped chickadee
- Zinc: American robin, American kestrel, and black-capped chickadee

The following risk description for birds focuses on the COPCs and receptors listed in the bullets.

8.4.1.1 Cadmium

American robin and black-capped chickadee have HQs ≥ 1.

- HQs:
  - **American robin:** Numerous 2012 Ecology Upland Soil Study and 2014 UCR Upland Soil Study samples have concentrations that result in estimated doses at or above the ED20s for growth (57 percent of both the 2012 Ecology Upland Soil Study and the 2014 UCR Upland Soil Study samples), and reproduction (51 percent of the 2012 Ecology Upland Soil Study and 46 percent of the 2014 UCR Upland Soil Study samples), respectively (Tables 8-1 and 8-2; Figure 8-3b). A single 2012 Ecology Upland Soil Study sample (1 percent of samples) has a concentration that results in an HQ ≥ 1 for survival.
  - **Black-capped chickadee:** Few samples (3 percent and 1 percent of the 2012 Ecology Upland Soil Study samples) have concentrations that result in HQs ≥ 1 for the growth and reproduction endpoints, respectively (Tables 8-1 and 8-2; Figure 8-3e).
- Nature and severity
  - American robin HQs ≥ 1 correspond to dose-response model estimates ranging from 20 percent to 52 percent reductions in body weight (ED20 to ED52 based on reductions in growth of juvenile chicken exposed to dietary cadmium), 20 to 50 percent reductions in reproduction (ED20 to ED50 based on reductions in egg production of chicken exposed to dietary cadmium), and a 28 percent reduction in survival (ED 28 based on reduced survival of Japanese quail exposed to dietary cadmium) (Table 8-2).
  - Black-capped chickadee HQs ≥ 1 correspond to dose-response model estimates ranging from 20 percent to 26 percent reductions in body weight (ED20 to ED26), and a 24 percent reduction in egg production (ED24) based on the same toxicity data as noted for American robin (Table 8-2).

- Spatial distribution:
  - For American robin, growth and reproduction TRV exceedances occur throughout the Terrestrial Study Area, principally along the river valley and adjacent valley terraces, with fewer in the higher elevations in the west and east of the Terrestrial Study Area (Map 8-4). The survival TRV exceedance occurs in a single 2012 Ecology Upland Soil Study sample from the uplands on the east side of the UCR (Map 8-4).
  - For black-capped chickadee, the three samples with concentrations resulting in growth or reproduction TRV exceedances all occur in 2012 Ecology Upland Soil Study samples located close together in the uplands on the east side of the UCR (Map 8-5).
- **Background:** All site soil concentrations exceed the BTV of 0.74 mg/kg (Table 8-3; Figures 8-3a through 8-3e).
- **Specific uncertainties:** Uncertainties do not strongly influence risk conclusions. Exposure may be slightly underestimated. Use of the 95 UCL rather than the ICS estimated means for COPC concentrations in soil samples from the 2014 UCR Upland Soil Study and 2015 Bossburg Study data set DUs slightly increases (by 5 percent) the number of growth HQs ≥ 1 for the 2014 UCR Upland Soil Study data set (Table 8-6). Effects thresholds are based on reasonable conservative estimates. As noted in Table 8-4, the selected growth and reproduction TRVs are based on toxicity studies in which domesticated bird species were dosed with soluble forms of cadmium that are likely more bioavailable than the forms that occur in dietary items in the Terrestrial Study Area.
- **Conclusions:**
  - American robin: Cadmium exposure poses risk of reduced growth and reproduction throughout the Terrestrial Study Area. Cadmium poses unacceptable risk to American robin and is retained as a COC for the invertivore bird EAE for evaluation in the RI.
  - Black-capped chickadee: Cadmium exposure poses risk of low to moderate reductions in growth and reproduction over a limited spatial extent in the uplands on the east side of the UCR. Because of the limited extent and low magnitude of predicted effects, and use of generally conservative estimates of exposure and effects, cadmium poses negligible risk to black-capped chickadee and the omnivorous birds EAE.

8.4.1.2 Lead

California quail, American robin, tree swallow, American kestrel, and black-capped chickadee all have some samples resulting in HQs ≥ 1 for lead. The representative species with the greatest number of HQs ≥ 1 for lead (from highest to lowest) are American robin, black-capped chickadee, American kestrel, California quail, and tree swallow (Table 8-1).

- **HQs**
  - **American robin:** Numerous samples from the three soil study data sets have concentrations that result in estimated doses at or above the LOAEL ≥ 20 for reproduction (70 percent of the 2012 Ecology Upland Soil Study, 58 percent of the 2014 UCR Upland Soil Study, and 67 percent of the 2015 Bossburg Study samples, respectively) and above the ED20 for survival (28 percent of the 2012 Ecology Upland Soil Study, 7 percent of the 2014 UCR Upland Soil Study, and 17 percent of the 2015 Bossburg Study samples, respectively), whereas few samples exceed the growth LOAEL ≥ 20 (3 percent of the 2012 Ecology Upland Soil Study and 17 percent of the 2015 Bossburg Study samples, respectively) (Tables 8-1 and 8-2; Figure 8-57b).

- **Black-capped chickadee:** Numerous samples from the three soil study data sets have concentrations that result in estimated doses at or above the LOAEL ≥ 20 for reproduction (41 percent of the 2012 Ecology Upland Soil Study samples, 18 percent of the 2014 UCR Upland Soil Study samples, and 33 percent of the 2015 Bossburg Study samples, respectively), whereas few samples have concentrations resulting in exceedance of the ED20 for survival (7 percent of the 2012 Ecology Upland Soil Study and one [17 percent] of the 2015 Bossburg Study locations) (Tables 8-1 and 8-2; Figure 8-7e).
- **American kestrel:** Several samples from the three soil study data sets have concentrations that result in estimated doses at or above the LOAEL ≥ 20 for reproduction (19 percent of the 2012 Ecology Upland Soil Study samples, 4 percent of the 2014 UCR Upland Soil Study samples, and 17 percent of the 2015 Bossburg Study samples, respectively), whereas few exceed the ED20 for survival (1 percent of the 2012 Ecology Upland Soil Study and 17 percent of the 2015 Bossburg Study sample locations), and none exceed the growth LOAEL ≥ 20 (Tables 8-1 and 8-2; Figure 8-7d).
- **California quail and tree swallow:** California quail has 3 percent of the 2012 Ecology Upland Soil Study samples resulting in an HQ ≥1 for the reproduction endpoint, and California quail and tree swallow both have one (17 percent) of the 2015 Bossburg Study sample resulting in an HQ ≥ 1 for the reproduction endpoint (Tables 8-1 and 8-2, Figures 8-7a and 8-7c).

- **Nature and severity:** Of the sample locations with survival HQs ≥ 1, estimated doses correspond to from 20 to 80 percent reductions in survival for American robin, up to 48 percent for black-capped chickadee, up to 29 percent for American kestrel, and up to 20 percent for California quail (based on reductions in pigeon survival when exposed to lead via oral gavage) (Table 8-2). EDxs are not available for the reproduction endpoint (based on the geometric mean of LOAELs for reductions in egg production of Japanese quail exposed to dietary lead) or the growth endpoint (based on LOAEL ≥ 20 for reduced growth of juvenile chicken exposed to dietary lead). Maximum reproduction HQs are 12, 5.2, 3.3, 2.4, and 1.1. for American robin, black-capped chickadee, American kestrel, California quail, and tree swallow, respectively (Table 8-2). The maximum American robin growth HQ is 2.0.
- **Spatial distribution:** The samples with concentrations resulting in HQs ≥ 1 for American robin and black-capped chickadee are generally located near the northern part of the Terrestrial Study Area, along the river valley and adjacent valley terraces, with a few in the higher elevations in the west and east of the Terrestrial Study Area and the 2015 Bossburg Study sampling area (Map 8-10). Samples with concentrations resulting in HQ ≥ 1 for American kestrel are generally located near the northern part of the Terrestrial Study Area, along the river valley, and a single location in the 2015 Bossburg Study sampling area (Map 8-9). The four samples with concentrations resulting in HQs ≥ 1 for California quail are in the uplands in the northern part of the Terrestrial Study Area, and in a single location in the Bossburg Study sampling area (Map 8-6). This 2015 Bossburg Study sample also results in the single HQ ≥ 1 for tree swallow (Map 8-8).
- **Background:** All site soil concentrations exceed the BTV of 27.2 mg/kg (Table 8-3; Figure 8-7).

- **Specific uncertainty**
  - **American robin:** Three specific uncertainties were further evaluated in the uncertainty analysis as presented in Table 8-8 including use of the 95 UCLs as the EPCs for the 2014 UCR Upland Soil Study and 2015 Bossburg Study DUs, use of an alternative soil to earthworm bioaccumulation model, and use of an alternative reproduction TRV. These alternative assumptions provide more realistic estimates of the potential for adverse effects than the baseline assumptions. When HQs are calculated using the alternative assumptions, the number of reproduction HQs ≥ 1 decrease by 13 percent to 17 percent for the three soil studies, though numerous samples from each of the three soil study data sets continue to result in estimated doses at or above the LOAEL ≥ 20 for reproduction (57 percent of the 2012 Ecology Upland Soil Study, 43 percent of the 2014 UCR Upland Soil Study, and 50 percent of the 2015 Bossburg Study samples, respectively). The first two uncertainties also affect the survival endpoint and if calculated would increase in the number of HQs ≥ 1. However, because the survival TRV (11 mg/kg bw/day) is higher than the alternative reproduction TRV (9.5 mg/kg bw/day), the extent and magnitude of HQs ≥ 1 for the reproduction endpoint exceed those for the survival endpoint regardless of whether alternative assumptions are applied.
  - **Black-capped chickadee, American kestrel, and tree swallow:** Two of the three specific uncertainties previously described for American robin apply equally to uncertainty in use of the 95 UCLs as the EPCs for the 2014 UCR Upland Soil Study and 2015 Bossburg Study DUs, and use of an alternative reproduction TRV. When HQs are calculated using the alternative assumptions, reproduction HQs ≥ 1 occur in up to 8.5 percent of the 2012 Ecology Upland Soil Study and up to 17 percent (one sample) of the 2015 Bossburg Study samples, respectively, depending on the specific receptor (Table 8-8). As for robin, the extent and magnitude of HQs ≥ 1 for the reproduction endpoint exceed those for the survival endpoint regardless of whether alternative assumptions are applied.
- **Conclusions**
  - American robin: Lead exposure poses risk of reduced reproduction throughout a substantial portion of the Terrestrial Study Area and risk of reduced survival over a more limited spatial extent and exceeds background in all locations. Lead poses unacceptable risk to American robin and the invertivore bird EAE.
  - Black-capped chickadee, American kestrel, California quail, and tree swallow: Lead exposure poses negligible risk to these receptors due to the limited extent of exceedances when HQs are calculated using more realistic alternative assumptions.

8.4.1.3 Mercury

American robin, tree swallow, and black-capped chickadee all have some samples resulting in HQs ≥ 1 for mercury. The representative species with the greatest number of HQ ≥ 1 (from highest to lowest) are American robin, black-capped chickadee, and tree swallow (Table 8-1).

- **HQs**
  - **American robin:** Numerous samples from the three soil study data sets have concentrations that result in estimated doses at or above the ED20 for reproduction (47 percent of the 2012 Ecology Upland Soil Study samples, 55 percent of the 2014 UCR Upland Soil Study samples, and 50 percent of the 2015 Bossburg Study samples, respectively), whereas a single sample exceeds the LOAEL ≥ 20 for survival (1 percent of the 2012 Ecology Upland Soil Study locations) (Tables 8-1 and 8-2; Figure 8-8b).

- **Black-capped chickadee:** Numerous samples from the three soil study data sets have concentrations that result in estimated doses at or above the ED20 for reproduction (46 percent of the 2012 Ecology Upland Soil Study samples, 50 percent of the 2014 UCR Upland Soil Study samples, and 50 percent of the 2015 Bossburg Study samples, respectively), whereas a single sample exceeds the LOAEL ≥ 20 for survival (1 percent of the 2012 Ecology Upland Soil Study locations) (Tables 8-1 and 8-2; Figure 8-8e).
- **Tree swallow:** Numerous samples from the three soil study data sets have concentrations that result in estimated doses at or above the ED20 for reproduction (34 percent of the 2012 Ecology Upland Soil Study samples, 29 percent of the 2014 UCR Upland Soil Study samples, and 50 percent of the 2015 Bossburg Study samples, respectively), whereas a single sample exceeds the LOAEL ≥ 20 for survival (1 percent of the 2012 Ecology Upland Soil Study locations) (Tables 8-1 and 8-2; Figure 8-8c).

- **Nature and severity:** Of the sample locations with reproduction HQs ≥ 1, estimated doses correspond to 20 percent to 36 percent reductions in reproduction for American robin, 20 percent to 36 percent reductions in reproduction for black-capped chickadee, and 20 percent to 34 percent reductions in reproduction for tree swallow based on toxicity data for survival of juvenile finches whose parents were exposed to dietary methylmercury (Table 8-2). EDxs are not available for the survival endpoint (based on toxicity data for survival of finches). The maximum survival HQs are 1.6 for both American robin and black-capped chickadee, and 1.3 for tree swallow (Table 8-2).
- **Spatial distribution:** HQs ≥ 1 for American robin, black-capped chickadee, and tree swallow occur throughout the Terrestrial Study Area, principally along the river valley and adjacent valley terraces, with fewer in the higher elevations in the west and east of the Terrestrial Study Area (Maps 8-11 through 8-13).
- **Background:** Some samples with concentrations ≤ BTV result in HQ ≥ 1 and some samples with concentrations > BTV result in HQs ≥ 1; Table 8-3).
- **Specific uncertainty**
  - **American robin:** Four specific uncertainties were further evaluated in the uncertainty analysis as presented in Table 8-8 including use of the 95 UCLs as the EPCs for the 2014 UCR Upland Soil Study and 2015 Bossburg Study DUs, use of an alternative soil to earthworm bioaccumulation model, the partitioning of mercury to methylmercury in bird dietary items, and use of an alternative reproduction TRV. These alternative assumptions provide more realistic estimates of the potential for adverse effects than the baseline assumptions. When HQs are calculated using the alternative assumptions, 8.5 percent of the 2012 Ecology Upland Soil Study samples and 17 percent of 2015 Bossburg Study samples have concentrations that result in reproduction HQs ≥ 1 (Table 8-8, Map 8-21). The first three uncertainties also affect the survival endpoint and if calculated would somewhat increase in the number of HQs ≥ 1. However, because the survival TRV (0.051 mg/kg bw/day) is higher than the alternative reproduction TRV (0.02 mg/kg bw/day), the extent and magnitude of HQs ≥ 1 for the reproduction endpoint exceed those for the survival endpoint indicating reproduction HQs are protective of survival endpoint.

- **Black-capped chickadee and tree swallow:** Three of the specific uncertainties discussed previously for American robin were further evaluated in the uncertainty analysis for black-capped chickadee and tree swallow as presented in Table 8-8 including use of the 95 UCLs as the EPCs for the 2014 UCR Upland Soil Study and 2015 Bossburg Study DUs, the partitioning of mercury to methylmercury in bird dietary items, and use of an alternative reproduction TRV. These alternative assumptions provide more realistic estimates of the potential for adverse effects than the baseline assumptions. When HQs are calculated using the alternative assumptions, 4.7 percent of 2012 Ecology Upland Soil and 17 percent of 2015 Bossburg Study samples have concentrations that result in reproduction HQs ≥ 1 for black-capped chickadee and 2.8 percent of 2012 Ecology Upland Soil and 17 percent of 2015 Bossburg Study samples have concentrations that result in reproduction HQs ≥ 1 for tree swallow (Table 8-8). Uncertainties in use of the 95 UCL and partitioning to methylmercury also affect the survival endpoint and if calculated would somewhat increase in the number of HQs ≥ 1. However, because the survival TRV (0.051 mg/kg bw/day) is higher than the alternative reproduction TRV (0.02 mg/kg bw/day), the extent and magnitude of HQs ≥ 1 for the reproduction endpoint exceed those for the survival endpoint indicating that reproduction HQs are protective of risks for the survival endpoint..

- **Conclusion:**
  - American robin - Uncertainty analyses that provide more realistic estimates of risk indicate that American robin reproduction may be adversely affected over a substantial extent in the northern portion of the Terrestrial Study Area. Mercury poses unacceptable risk to American robin and the invertivore bird EAE.
  - Black-capped chickadee and tree swallow - Because uncertainty analyses that provide more realistic estimates of risk indicate that the extent of risk is sufficiently small that the black-capped chickadee and tree swallow populations would be unaffected, thus mercury poses negligible risk to black-capped chickadees and the omnivore EAE, and tree swallow and the aerial insectivore EAE.

8.4.1.4 Selenium

Black-capped chickadee, tree swallow, American robin, and American kestrel (in order of the number of exceedances) all have a relatively small number of samples with concentrations resulting in HQs ≥ 1 for the survival, growth, and reproduction endpoints (Tables 8-1 and 8-2; Figures 8-10a through 8-10e). Because risks are similar for all EAEs, risk conclusions are summarized together for all of the bird receptors of concern.

- **HQs**
  - For black-capped chickadee there are few samples from the three soil study data sets that have concentrations resulting in estimated doses at or above the Eco-SSL for growth (6 percent of the 2012 Ecology and 2 percent of the 2014 UCR Upland Soil Study samples, respectively), whereas 4 percent and 3 percent of 2012 Ecology Upland Soil Study samples have concentrations resulting in exceedance the survival and reproduction TRVs, respectively. A single 2014 UCR Upland Soil Study sample has a concentration resulting in an HQ ≥ 1 for the survival endpoint (Tables 8-1 and 8-2; Figure 8-10e). No 2015 Bossburg Study soil samples have concentrations resulting in HQs ≥ 1 for any endpoint or receptor.
  - American robin growth HQs ≥ 1 correspond to concentrations in 4 percent of the 2012 Ecology Upland Soil Study samples, and 1 percent of 2014 UCR Upland Soil Study samples. A single (1 percent) 2012 Ecology Upland Soil Study sample has a concentration resulting in reproduction and survival HQ ≥ 1 (Table 8-1; Figure 8-10b).

  - Tree swallow growth HQ ≥ 1 correspond to concentrations in 6 percent of the 2012 Ecology Upland Soil Study samples, and 1 percent of the 2014 UCR Upland Soil Study samples (Table 8-1; Figure 8-10c). Reproduction and survival HQs ≥ 1 both correspond to concentrations in 3 percent of the 2012 Ecology Upland Soil Study samples, and 1 percent (that is, 1 sample) of the 2014 UCR Upland Soil Study samples.
  - American kestrel has survival, growth, and reproduction HQs ≥ 1 corresponding to a single 2012 Ecology Upland Soil Study sample, and a single 2014 UCR Upland Soil Study sample results in an HQ ≥ 1 for the growth endpoint (Table 8-1; Figure 8-10d).

- **Nature and severity:** Of the sample locations with concentrations resulting in reproduction HQs ≥ 1, estimated doses correspond to 24 percent to 77 percent reductions in reproduction, with median reductions of 24 to 38 percent based on toxicity data for hatchability of chickens (Table 8-2). EDxs are not available for the growth endpoint (based on the avian Eco-SSL) and the survival endpoint (based on toxicity data for survival of chickens). Maximum growth HQs are 6.1, 5.6, 2.9, and 2.1 for the black-capped chickadee, tree swallow, American robin, and American kestrel, respectively (Table 82). Maximum survival HQs are 3.0, 2.8, 1.4, and 1.0 for the black-capped chickadee, tree swallow, American robin, and American kestrel, respectively (Table 8-2).
- **Spatial distribution:** The six 2012 Ecology Upland Soil Study samples with concentrations exceeding TRVs all occur in the northeastern part of the Terrestrial Study Area in the higher elevations, whereas the three 2014 UCR Upland Soil Study samples that result in exceedances are from spatially disparate locations on the east side of the Terrestrial Study Area (Maps 8-14 through 8-17).
- **Background:** All site soil concentrations exceed the BTV of 0.098 mg/kg (Table 8-3; Figures 8-10a through 8-10e).
- **Specific uncertainty:** The soil concentrations in two of the samples resulting in HQs ≥ 1 (both HQs = 1.2) are below the MDLs, thus, actual concentrations in these samples are likely lower than the reported concentration (the MDL) (Table 8-9). The reproduction and survival TRVs are based on reasonable conservative estimates. The growth TRV is based on the avian Eco-SSL, which is likely to overestimate the potential for adverse effects. The selected TRVs are based on toxicity studies in which domesticated bird species were dosed with soluble forms of selenium that are likely more bioavailable than the forms that occur in dietary items in the Terrestrial Study Area.
- **Conclusion:** Because of the limited spatial extent of samples with concentrations resulting in exceedances of the TRVs, generally low magnitude of potential effects on the reproduction endpoint, and generally conservative estimates of exposure and effects, the extent and magnitude of risk is sufficiently small that the black-capped chickadee, American robin, tree swallow, and American kestrel populations are unlikely to be adversely affected, thus selenium poses negligible risk to these species and the omnivore, invertivore, aerial insectivore, and carnivore EAEs.

8.4.1.5 Zinc

Black-capped chickadee, American robin, and American kestrel (in order of the number of exceedances) have HQs ≥ 1 for the reproduction and growth endpoints, whereas California quail and tree swallow have HQs < 1 for all endpoints (Table 8-1).

- **HQs**
  - Black-capped chickadee growth endpoint HQs ≥ 1 correspond to concentrations occurring in 38 percent of the 2012 Ecology Upland Soil Study and 24 percent of the 2014 UCR Upland Soil Study samples. Reproduction HQs ≥ 1 correspond to concentrations occurring in 30 percent of the 2012 Ecology Upland Soil Study, and 14 percent of the 2014 UCR Upland Soil Study sample locations (Table 8-1, Table 8-2; Figure 8-12e). No Bossburg Study soil samples have concentrations resulting in HQs ≥ 1 for any receptor or endpoint.
  - American robin growth endpoint HQs ≥ 1 correspond to concentrations occurring in 33 percent of the 2012 Ecology Upland Soil Study and 18 percent of the 2014 UCR Upland Soil Study soil sample locations. Reproduction HQs ≥ 1 correspond to concentrations occurring in 19 percent of the 2012 Ecology Upland Soil Study and 4 percent of the 2014 UCR Upland Soil Study soil sample locations (Table 8-1, Table 8-2; Figure 8-12b).
  - American kestrel has relatively few samples with concentrations resulting in HQ ≥ 1 for the growth endpoint (4 percent of the 2012 Ecology Upland Soil Study, and 1 percent of samples from the 2014 UCR Upland Soil Study). For the reproduction endpoint a single 2012 Ecology Upland Soil Study sample location (1 percent of samples) has a concentration resulting in an HQ ≥ 1 (Table 8-1, Table 8-2; Figure 8-12d).
- **Nature and severity:** Of the sample locations with reproduction HQs ≥ 1, estimated doses result in up to 96 percent, 70 percent, and 21 percent reduced reproduction (ED96 to ED21) for black-capped chickadee, American robin, and American kestrel, respectively based on reduced egg production in chickens exposed to dietary zinc (Table 8-2). EDxs are not available for the growth endpoint, which is based on the avian Eco-SSL (calculated as the geometric mean of 34 NOAELs for growth and 9 NOAELs for reproduction). Maximum growth HQs range from 1.0 to 6.1 (Table 8-2).
- **Spatial distribution:** For black-capped chickadee and American robin, samples resulting in estimated doses at or above the growth and reproduction TRVs are distributed throughout the Terrestrial Study Area, principally along the river valley and adjacent valley terraces, with fewer in the higher elevations in the west and east of the Terrestrial Study Area (Map 8-18). For American kestrel, the 2012 Ecology Upland Soil Study samples resulting in HQs ≥ 1 occur on both sides of the river and the 2014 UCR Upland Soil Study sample resulting in an HQ ≥ 1 occurs in the northeastern portion of the Terrestrial Study Area (Map 8-19).
- **Background:** All site soil concentrations exceed the BTV of 111 mg/kg (Table 8-3; Figures 8-10a through 8-10e).
- **Specific uncertainty:** As noted in Table 8-4, the survival and growth TRVs are based on insoluble forms of zinc, which are likely more like zinc in bird diets at the site, whereas the reproduction TRV is based on a highly soluble form of zinc, which likely has greater bioavailability than the zinc in bird diets at the site. Furthermore, in the uncertainty analysis, two specific uncertainties were evaluated for zinc, including use of the 95 UCLs as the EPCs for the 2014 UCR Upland Soil Study and 2015 Bossburg Study DUs, and use of an alternative soil to earthworm bioaccumulation model. Use of the 95 UCLs as the EPCs, though calculated only for American robin, applies to all receptors, whereas use of the alternative soil to earthworm bioaccumulation model applies only to American robin.

- When calculated using the 95 UCL rather than the ICS estimated mean as the concentration of zinc in soil, the number of 2014 UCR Upland Soil Study DUs with estimated doses resulting in American robin HQs ≥ 1 for the growth endpoint increases from 18 percent of DUs to 26 percent of DUs, indicating that use of the ICS estimated mean may underestimate the number of DUs resulting in HQs ≥ 1 by about 8 percent (Table 8-6). No change in the number of HQs ≥ 1 is estimated for the 2015 Bossburg Study when calculated using the 95 UCL rather than the ICS estimated mean. Because the growth endpoint is the most sensitive endpoint for zinc (lowest TRV), and American robin has a similar number of HQs ≥ 1 or more than other receptors, these results illustrate that use of the ICS estimated mean as the concentration of zinc in soil slightly underestimates the risk to birds in the Terrestrial Study Area.
- Use of an alternative soil to earthworm bioaccumulation model to estimate exposure of American robin to zinc results in fewer than half as many samples with concentrations resulting in HQs ≥ 1 for the reproduction endpoint as when calculated using the bioaccumulation model assumed in the baseline calculations. When calculated using the alternative model, the percentage of samples with concentrations resulting in HQs ≥ 1 is 16 percent of the 2012 Ecology Upland Soil Study, 1.4 percent of the 2014 UCR Upland Soil Study, and 0 percent of the 2015 Bossburg Study samples (Table 8-7). Because the growth endpoint is the most sensitive endpoint for zinc (lowest TRV), these results illustrate that risk to American robin is substantially reduced for all endpoints when the alternative earthworm bioaccumulation model is used to estimate exposure.

- **Risk Conclusions**
  - **Black-capped chickadee** and **American robin:** Zinc exposure poses risk of reduced growth and reproduction throughout a substantial portion of the Terrestrial Study Area and risk of reduced survival over a limited spatial extent and exceeds background in all locations. Zinc poses unacceptable risk to black-capped chickadee and the omnivore EAE and American robin and the invertivore EAE.
  - **American kestrel:** Because of the limited spatial extent of samples with concentrations resulting in exceedances of the TRVs, generally low magnitude of potential effects on the growth and reproduction endpoints, and generally conservative estimates of exposure and effects, the extent and magnitude of risk is sufficiently small that the American kestrel populations are unlikely to be adversely affected, thus zinc poses negligible risk to American kestrel and the carnivore EAE.

8.4.2 Multiple Metals

The preceding discussion focuses on individual COCs, with limited discussion of multiple metals with HQ ≥ 1. This section explores locations and portions of the site having the greatest potential for the combined effects of multiple metals. Table 8-9 identifies those sample locations with HQ ≥ 1, listing all of the COCs exceeding the most sensitive endpoint for the most sensitive species (American robin for all COCs, except black-capped chickadee for zinc) at each location.

Maps 8-22a through 8-22d combine results for the bird COCs, cadmium, lead, mercury, and zinc with results for each COC shown in a cluster of four symbols at each location. The HQ comparisons are indicated with colors and different sized symbols as described in Section 8.1 for Maps 8-1 through 8-20. On Maps 8-22a through 8-22d, the four symbols are arranged in two rows, with cadmium and mercury HQs represented by a triangle and hexagon, respectively, in the top row and lead and zinc HQs represented by a circle and square, respectively, in the bottom row.

As illustrated in Table 8-9 and on Maps 8-22a through 8-22d, at most locations where HQs ≥ 1 occur for the most exposed mammal receptor, concentrations of multiple metals results in exceedance of their respective TRVs. Of the 139 locations with HQs ≥ 1 for cadmium, only 7 locations have HQs ≥ 1 for cadmium alone. At the remaining 132 locations, cadmium TRV exceedances are accompanied by lead, mercury, selenium, or zinc, singly or in combination (Table 8-9). Similarly, lead alone results in HQs ≥ 1 at 18 of the 160 locations with HQs ≥ 1 for lead; the remaining locations are accompanied by cadmium, mercury, selenium, and/or zinc. Mercury HQs ≥ 1 are accompanied by HQs ≥ 1 for one or more other COCs at all 37 locations where mercury concentration > BTV and where mercury HQs ≥ 1. In all 74 locations with HQs ≥ 1 for zinc; cadmium, lead, mercury, and/or selenium also exceed their respective TRVs. Lead exceedances are most widespread, occurring near the northern part of the Terrestrial Study Area, along the river valley and adjacent valley terraces, and in a few locations in the higher elevations in the western and eastern parts of the Terrestrial Study Area and the 2015 Bossburg Study sampling area. Cadmium, mercury, and zinc exceedances are distributed similarly across the landscape though over fewer locations.

As described in Section 8.3.3.1, the simultaneous exposure of birds to elevated concentrations of multiple metals results in complex interactions, the effects are difficult to predict, and assumption of independent action may result in underestimates of the combined risks. For the purposes of this risk assessment, it is assumed that locations with multiple COCs exceeding benchmarks pose a greater risk to birds than those locations with fewer exceedances and that risk at a specific location is at least as great as that associated with the COC with the highest HQ.

8.5 Bird Risk Conclusions

This Upland BERA evaluates the following risk question related to birds within the Terrestrial Study Area:

- Do the daily doses of COPCs received by birds (represented by guilds focused on specific avian species) from consumption of the tissues of prey, plants, and soil in the Terrestrial Study Area exceed the TRVs for survival, growth, or reproduction of birds such that adverse effects to the local population are expected?

Based on the analyses discussed in the risk characterization, cadmium, lead, and zinc are COCs that present the greatest, most widespread risk to birds in the Terrestrial Study Area. This conclusion is based on widespread exceedances of TRVs for the three COCs, with the most likely effects being reduced growth and reproduction, though several samples also have concentrations exceeding survival TRVs. Additionally, mercury poses unacceptable risk to American robin over a smaller portion of the Terrestrial Study Area. COPCs and respective EAEs (and respective representative species) retained as COCs include:

- Cadmium: Invertivore EAE (American robin)
- Lead: Invertivore EAE (American robin)
- Mercury: Invertivore EAE (American robin)
- Zinc: Invertivore EAE (American robin), and omnivore (black-capped chickadee).
- All COPCs pose negligible risks to the herbivore (California quail), aerial insectivore (tree swallow), and carnivore (American kestrel) EAEs.

The uncertainty analysis identifies several factors that contribute to over and underestimates of the potential for adverse effects. Considering these uncertainties, exposure and effects data were selected using robust methods to result in reasonable conservative estimates of the severity of effects resulting from exposure in the Terrestrial Study Area. Risk estimates for cadmium, lead, mercury, and zinc are based on robust toxicity data sets and cadmium, lead, and zinc incorporate dose-response information into the risk estimates, which increases the reliability and relevance of the analysis resulting in a moderate degree of confidence in the risk prediction for all four of these COCs. Because there are no site-specific measures of effects, risk predictions remain somewhat uncertain.

The remaining COPCs for birds, aluminum, barium, chromium, copper, iron, molybdenum, selenium, and vanadium present negligible risk and are not carried forward as COCs. Uncertainty is not likely to result in underestimation of risk. Consequently, there is a high degree of confidence that these COPCs pose negligible risk.

9. Risk Characterization for Mammals

This risk characterization section assesses whether exposure of mammals to COPCs in soils may pose an unacceptable risk in the Terrestrial Study Area. The problem formulation for upland mammals, including an ecological overview, is presented in Section 2. The methods for exposure and effects assessment and analysis are presented in Section 4, and the approach for integrating this information to characterize risks is presented in Section 5.

The risk characterization is structured as follows:

- Risk Estimation: presents the results of the risk analysis
- Comparison to Background Soil Concentrations: compares site soils to BTVs for each COPC
- Uncertainty Analysis: analyzes and discusses the major areas of uncertainty
- Risk Description: describes the likely nature, severity, and spatial scale of effects
- Risk Conclusions: summarizes the risks and overall degree of confidence in the risk estimates

Five mammal feeding guilds, plus threatened and endangered mammals represent the mammal EAEs with one representative species selected for each as follows:

- Herbivores: meadow vole
- Invertivores: masked shrew
- Aerial insectivores: little brown bat
- Omnivores: deer mouse
- Carnivores: short-tailed weasel
- Threatened and endangered mammals: gray wolf

For each EAE, the risk question is as follows:

- Do the daily doses of COPCs received by mammals (represented by guilds focused on specific mammalian species) from consumption of the tissues of prey, plants, and soil in the Terrestrial Study Area exceed the TRVs for survival, growth, or reproduction of mammals such that adverse effects to the local population are expected?

The single LOE for each representative species used for characterizing risk to the mammal EAEs is as follows:

- Dietary doses of COPCs compared to TRVs and dose-response information

9.1 Risk Estimation

This section presents the risk analyses for the mammal population EAEs and summarizes the results. Data sets used for the mammal risk characterization are the 2012 Ecology Upland Soil Study (Ecology, 2013), the 2014 UCR Upland Soil Study (TAI, 2015), and the 2015 Bossburg Study (TAI, 2016). COPCs for mammals are aluminum, antimony, cadmium, chromium, copper, iron, lead, mercury, molybdenum, selenium, thallium, and zinc (Table 2-7). The one COPC without a TRV (antimony) is discussed in the Uncertainty section (Section 9.3).

Daily doses for each representative species are developed for every sample and COPC and are compared to the appropriate COPC-specific TRVs resulting in HQs. As described in Section 4.2.2 and Appendix E, TRVs are based on the ED20 (the dose resulting in a 20 percent reduction in response relative to the control) for three separate endpoints (growth, reproduction, and survival), when possible. HQs are interpreted in the risk characterization as follows:

- **HQ < 1**. The estimated exposure does not exceed the 20 percent effect threshold (based on the ED20, unless otherwise noted) and is unlikely to cause an adverse effect to the EAE. Risk is negligible.
- **HQ ≥ 1**. Unacceptable risk cannot be ruled out. The nature and magnitude of potential adverse effects associated with the estimated exposure at each applicable location exceeds 20 percent, based on the ED20 (unless otherwise stated), and is further evaluated and described.

Accordingly, for those COPCs with HQs ≥ 1 that have dose-response models available, sample-specific effective doses (EDx) are calculated from the models. The EDx values provide an estimate of the percentage (x) of the receptor population exhibiting the adverse effect at the COPC dose estimated for a given soil sample. COPCs and endpoints with available dose-response models are those listed in Table 4-7 that have an ED20 indicated as the TRV type.

The number of samples from each soil study with concentrations resulting in HQs ≥ 1 for each COPC and representative species are reported in Table 9-1. For those species and COPCs with HQs ≥ 1, EDx dose-response estimates (where possible) are summarized in Table 9-2. Calculations and underlying data for HQ and EDx values are provided in Appendix F. For each COPC and receptor of concern, Figures 9-1 through 9-11f show cumulative distributions of the number of samples from each soil study data set with concentrations resulting in HQs ≥ 1.

Maps for each representative species are provided for all COPCs and endpoints with HQs ≥ 1 (Maps 9-1 through 9-17). HQs for survival, growth, and reproduction endpoints are depicted using three symbols arranged in two rows, with survival HQs in the top row and growth and reproduction in the bottom row. If a dose-response model is available for a given COPC and endpoint, the respective map presents the calculated EDx value for each sample location using a square. If no dose-response model is available, the map indicates which sample locations have HQs < 1, HQs ≥ 1 and HQs ≥ 5 using a circle. HQs < 1 are depicted as small green circles. HQs ≥ 1 and EDx ≥ 20 are shown in large circles and squares, respectively, with the colors yellow and blue indicating the HQ or EDx range categorically at each location. Locations with COPC concentrations ≤ BTV are shown with an open symbol; locations with COPC concentrations > BTV are shown with solid symbols.

For all the representative species, chromium, iron, mercury, molybdenum, and thallium have daily doses below their respective TRVs (HQ < 1) for all sample locations in all three soil studies (Table 9-1). Note that molybdenum concentrations were only reported for the 2014 UCR Upland Soil Study. Aluminum, cadmium, copper, lead, selenium, and zinc have HQs ≥ 1 for two or more species from two or all three studies and are briefly discussed as follows:

- **Aluminum.** Masked shrew, meadow vole, and short-tailed weasel (in decreasing order of the number of exceedances) have a similar number of HQs ≥ 1. Numerous samples from all three soil studies have concentrations resulting in HQs ≥ 1 for the reproduction endpoint (more than 83 percent, 78 percent, and 33 percent of the 2012 Ecology Upland Soil Study, 2014 UCR Upland Soil Study, and 2015 Bossburg Study samples, respectively for the three species; Table 9-1; Figures 9-1a through 9-1f). Deer mouse has fewer than half as many samples with concentrations resulting in reproduction HQs ≥ 1, and gray wolf has a couple of samples from the 2012 Ecology Upland Soil Study resulting in reproduction HQs ≥ 1 (Table 9-1). Masked shrew, with a single sample resulting in both a growth and a survival HQ ≥ 1, is the only receptor with an HQ ≥ 1 for another endpoint. Where HQs are ≥ 1, median EDxs for the different species range from 23 percent to 41 percent reduced reproduction for the different species and soil studies (Table 9-2). Maximum EDxs range up to an 84 percent reduction in reproduction for masked shrew (Table 9-2). EDxs are not available for the survival and growth endpoints. Masked shrew survival and growth HQs are both 1.3 (Table 9-2).

- **Cadmium.** Masked shrew and deer mouse are the only species with HQs ≥ 1. Masked shrew has numerous 2012 Ecology Upland Soil Study and 2014 UCR Upland Soil Study samples with estimated doses at or above the ED20s for survival (63 percent of the 2012 Ecology Upland Soil Study and 64 percent of the 2014 UCR Upland Soil Study samples), and reproduction (21 percent of the 2012 Ecology Upland Soil and 10 percent of the 2014 UCR Upland Soil studies locations) (Tables 9-1 and 9-2; Figures 9-2a through 9-2f). Three 2012 Ecology Upland Soil Study sample locations (3 percent of samples) result in HQs ≥ 1 for growth. Where masked shrew HQs are ≥ 1, median EDxs are ≥ 21 percent for the different soil studies and endpoints, ranging up to a maximum 40 percent reduction in survival associated with a sample from the 2012 Ecology Upland Soil Study (Table 9-2). For deer mouse, a single 2012 Ecology Upland Soil Study sample (1 percent of samples) results in a survival HQ ≥ 1 with an EDx of 21 percent.
- **Copper.** Copper has samples with concentrations resulting in HQs ≥ 1 only for the masked shrew (Tables 9-1 and 9-2; Figures 9-4a through 9-4f). Numerous samples have concentrations resulting in HQs ≥ 1 for the survival endpoints (54 percent, 32 percent, and 33 percent of the 2012 Ecology Upland Soil Study, 2014 UCR Upland Soil Study, and 2015 Bossburg Study samples, respectively) and about half as many samples result in HQs ≥ 1 for the growth endpoint (Tables 9-1 and 9-2; Figure 9-4b). Of the sample locations with HQs ≥ 1, estimated doses result in up to 97 percent reduced growth (ED97) (Table 9-2). EDxs are not available for the survival endpoint. The maximum survival HQ is 2.8 (Table 9-2).
- **Lead.** Masked shrew, deer mouse, and meadow vole (in decreasing order of the number of exceedances) have HQs ≥ 1 for the reproduction, survival, and growth endpoints (in order of the number of exceedances). Short-tailed weasel has two HQs ≥ 1 for the reproduction endpoint (Tables 9-1 and 9-2; Figures 9-6f). For masked shrew, 77 percent of both the 2012 Ecology Upland Soil Study and 2014 UCR Upland Soil Study samples, and 67 percent of 2015 Bossburg Study samples have concentrations resulting in HQs ≥ 1 for the reproduction endpoint. For the survival endpoint, masked shrew has somewhat fewer HQs ≥ 1 (63 percent, 47 percent, and 50 percent for the 2012 Ecology Upland Soil, 2014 UCR Upland Soil Study, and 2015 Bossburg studies, respectively) and for the growth endpoint, few HQs ≥ 1 (16 percent, 1 percent, and 17 percent for the 2012 Ecology Upland Soil, 2014 UCR Upland Soil Study, and 2015 Bossburg studies, respectively). Deer mouse and meadow vole both have relatively few samples with concentrations resulting in HQs ≥ 1 for the reproduction and survival endpoints (up to 24 percent of 2012 Ecology Upland Soil Study samples, up to 6 percent of 2014 UCR Upland Soil Study samples, and up to 17 percent of 2015 Bossburg Study samples for the two species), and for the survival endpoint a single sample from the 2015 Bossburg Study results in an HQ ≥ 1 for both species (Table 9-2). Of the sample locations with survival HQs ≥ 1, estimated doses correspond to median EDxs of 44 percent to 63 percent reductions in survival for masked shrew across the three soil studies with somewhat similar median EDxs for the meadow vole and deer mouse (34 percent to 88 percent) (Table 9-2). EDxs are not available for the reproduction or growth endpoints. Maximum reproduction and growth HQs are 15 and 3.5, respectively, both for masked shrew for a sample concentration from the 2015 Bossburg Study (Table 9-2).

- **Selenium.** Masked shrew, little brown bat, and deer mouse (in decreasing order of the number of exceedances) all have a relatively small number of HQs ≥ 1 for the survival and growth endpoints (Tables 9-1 and 9-2; Figures 9-9a through 9-9f). For the species and endpoint with the greatest number of HQs ≥ 1, masked shrew growth HQs ≥ 1 are associated with concentrations in 12 percent of the 2012 Ecology Upland Soil Study samples, 16 percent of the 2014 UCR Upland Soil Study samples, and 17 percent of the 2015 Bossburg Soil Study samples (Tables 9-1 and 9-2; Figure 9-2). Of the sample locations with HQs ≥ 1, estimated doses result in up to 91 percent, 68 percent, and 55 percent reduced growth (ED91 to ED55) for masked shrew, little brown bat, and deer mouse respectively (Table 9-2). EDxs are not available for the survival endpoint. Maximum survival HQs range from 1.6 to 5.6 (Table 9-2). No 2015 Bossburg Study sample concentrations result in HQs ≥ 1 (Table 9-1).
- **Zinc**. Masked shrew and deer mouse are the only species with HQs ≥ 1. Masked shrew has numerous samples with concentrations resulting in HQs ≥ 1 for the survival, reproduction, and growth endpoints, whereas deer mouse has relatively few HQs ≥ 1 for the growth and reproduction endpoints (Tables 9-1 and 9-2; Figures 9-11a through 9-11f). For masked shrew, 67 percent of the 2012 Ecology Upland Soil Study and 72 percent of the 2014 UCR Upland Soil Study samples have concentrations resulting in HQs ≥1 for both the growth and reproduction endpoints. For the survival endpoint, masked shrew has relatively few HQs ≥ 1, with concentrations resulting in HQs ≥ 1 occurring in 17 percent and 2 percent of the 2012 Ecology Upland Soil Study and 2014 UCR Upland Soil Study samples, respectively (Table 9-1). EDxs are not available for any of the endpoints. Maximum HQs range up to 6.0 for the growth and reproduction endpoints for masked shrew and up to 1.7 for the growth and reproduction endpoints for deer mouse (Table 9-2). No 2015 Bossburg Study samples result in HQ ≥ 1 (Table 9-1).

9.2 Comparison to Background Soil Concentrations

This section presents a comparison of soil concentrations measured in each of the three data sets to the BTVs shown in Table 2-8. Table 9-3 summarizes the number of samples for each COPC from each soil study data set for each species and endpoint that have concentrations resulting in HQs ≥ 1 and are ≤ the BTV or > the BTV.

- COPCs that exceed the BTV and result in an HQ ≥ 1 for at least one endpoint for at least one receptor in at least one sample include: cadmium, copper, lead, selenium, and zinc (Table 8-3). All aluminum samples are ≤ BTV.
- Copper is the only COPC for which some samples with concentrations ≤ BTV result in HQ ≥ 1 and some samples with concentrations > BTV result in HQs ≥ 1. Masked shrew is the only receptor with copper HQs ≥ 1 from samples > BTV (for both the survival and growth endpoints).

9.3 Uncertainty Analysis

Uncertainties associated with the risk characterization for mammals are presented here and in Table 9-4. Key uncertainties associated with the exposure assessment and effects assessment are also evaluated.

Uncertainties in several of the parameters underlying the HQ calculations, including exposure model parameters (Equation 4-1), are evaluated by calculating HQs using different scenarios. For each scenario, the change in the number of sample results resulting in HQs ≥ 1 using alternative assumptions from the number calculated using the baseline assumptions described in Section 4 is evaluated.

9.3.1 Exposure Assessment

Calculation of the daily doses of COPCs that each representative mammal species is exposed to throughout the site includes many individual parameters (Equation 4-1). Except for soil COPC concentrations, soil to plant bioaccumulation models, and soil RBAs, parameter values are based on literature-reported, rather than site-specific data. The selected values for each parameter have associated variability and uncertainty, which can influence exposure estimates, risk characterization, and risk conclusions. Table 9-4 describes uncertainties in the soil chemistry EPCs, bioaccumulation models, and exposure calculations, and evaluates the likelihood that they contribute to over- or underestimates of risk. As indicated in Table 9-4, it is unknown whether most identified uncertainties contribute to over- or underestimates of site risks. In general, reasonable conservative assumptions were made to ensure that risk is not underpredicted. The bioaccumulation models used to predict uptake from soils to receptor diets have several associated uncertainties, which are described in full in Appendix C and are summarized in Table 9-4. In general, bioaccumulation of metals from soil to plant and animal tissues is highly variable and depends on several site-specific and species-specific factors, such that the likelihood for the bioaccumulation models to over- or underpredict risks is unknown. Exposure calculations are generally based on conservative assumptions including selection of WEFs and AUFs that contribute to conservative estimates of site risks. For some parameters, quantitative uncertainty analyses are presented to determine whether additional information provides evidence that baseline exposure estimates result in over- or underestimates of exposure and to refine risk estimates and risk conclusions. Quantitative uncertainty analyses are presented in the following paragraph for RBA (the fraction of a COPC in incidentally ingested soil that is absorbed), and soil chemistry EPCs, and spatial aggregation of exposure across sample locations for large home range species.

Relative Bioavailability

RBA estimates vary by COPC and sample as a function of IVBA. Uncertainties associated with the calculation of RBA are summarized in Table 9-4 and described more fully in Appendix E. Sensitivity of the risk calculations to variability in the RBA is evaluated to determine if differences in predicted RBAs among site samples result in a change in the number of HQs ≥ 1. As shown in Table 9-5, in most cases the number of HQs ≥ 1 calculated assuming all samples have the minimum RBA for a given COPC over all site samples is the same as when assuming all samples have the maximum site-specific RBA. Exceptions are aluminum, lead, zinc, copper, and iron, with the latter two differing only for shrew. Where there are differences in number of HQs ≥ 1 due to variability in the RBA, only for iron (and then only for shrew for the 2012 Ecology Upland Soil Study data) is there a potential change to risk conclusions.

This indicates that risk predictions are generally not sensitive to RBA within the range predicted for site-specific samples. Site aluminum concentrations are $\leq$ BTV, and for lead, copper, and zinc, numerous samples result in HQ ≥ 1 regardless of which RBA is selected.

Soil Chemistry Exposure Point Concentrations

Given the large area of the Terrestrial Study Area, the soil chemistry data set only partially characterizes the full magnitude and extent of contamination. The strengths and uncertainties associated with each of the three soil studies used to characterize exposure to soil invertebrates are the same as those discussed in Section 6.3.2 for plants.

For the 2014 UCR Upland Soil and 2015 Bossburg studies, EPCs are calculated as the reported COPC concentrations determined for each DU, which may under- or overestimate the true mean COPC concentration in the DU due to site variability and sampling error. Following the same approach described in Section 6.3.2 for plants, an uncertainty analysis for mammals is presented herein where EPCs for each DU in the 2015 Bossburg and 2014 UCR Upland Soil studies data sets is calculated as the 95 UCL. The daily COPC doses for each mammal receptor are calculated following the methods described in Section 4.1.1, substituting the 95 UCL for the COPC concentration in soil. HQs are calculated as the Daily $Dose_{UCL95}$ divided by the TRV for the most sensitive endpoint (survival, growth, or reproduction) for each COPC. Table 9-6 summarizes the number of HQ ≥ 1 when calculated as the Daily $Dose_{UCL95}$/TRV relative to the Daily $Dose_{mean}$/TRV. When calculated using the 95 UCL, the number of DUs with HQs ≥ 1.0 in the 2014 UCR Upland Soil Study data set increases for aluminum, cadmium, copper, iron, lead, selenium, and zinc with increases greater than 10 percent for copper and zinc, and smaller increases for the other COPCs. These calculations indicate that use of the reported concentration (ICS estimated mean) for each DU as the EPC rather than the 95 UCL may somewhat underestimate the spatial extent and magnitude of risk to mammal receptors of concern.

Area Use Factor

Area use is assumed to be 100 percent for all receptors. This assumes that a receptor is exposed entirely within a given DU (2014 UCR Upland Soil Study and 2015 Bossburg Study) or sample location (2012 Ecology Upland Soil Study) at all times. This creates uncertainty in COPC exposure to receptors that forage over areas larger or smaller than an exposure unit, populations that encompass multiple locations, and/or receptors that migrate such that they would be present in a given location only a portion of the year (e.g., seasonally). As illustrated by the circles depicting the home range for each respective receptor of concern on Maps 9-1 through 9-17, most samples are located too far apart from one another for more than a single sample to fall within the home range of an individual of a given species. Thus, aggregating concentration data from adjacent sample locations together would not meaningfully change the pattern of HQs ≥ 1 or < 1 across the site. An exception to this observation is the little brown bat. The home range of the little brown bat illustrated by the circle on Map 9-14 is 780 acres (the lower end of the range in Table 2-11). This indicates that the little brown bat's exposure may be integrated across multiple sample locations, and that HQ ≥ 1 in any single location may over- or underestimate the average exposure, depending on the concentrations in the surrounding locations. As illustrated on Map 9-14, several samples in the northeastern part of the Upland Terrestrial Study Area with selenium concentrations resulting in growth EDx ≥ 20 percent and ≥ 50 percent fall within the home range of the little brown bat suggesting that average exposure in this area poses elevated risk to little brown bat.

9.3.2 Effects Assessment

9.3.2.1 Toxicity Reference Values

Uncertainties surrounding the reliability of specific TRVs to estimate ED20s are listed in Table 9-4 for COPCs with HQs ≥ 1. These COPCs are aluminum (reproduction), cadmium (survival, growth, and reproduction), copper (survival, growth), lead (survival, growth, and reproduction), selenium (survival, growth), and zinc (survival, growth, and reproduction). Additional information on the reliability and relevance of these TRVs is provided in Table 4-10. As indicated in Table 9-4, in general the ability of the TRVs to predict the severity of effects resulting from site exposure is unknown. Several site-specific factors influence the bioavailability and toxicity of metals and TRVs are selected using a robust process (described in Appendix E) to result in reasonable conservative estimates of site-specific adverse effects. Uncertainties in selection of TRVs are described in detail in Appendix E.

9.3.3 Risk Characterization

Uncertainties in the risk characterization are presented in Table 9-4 including use of HQs to characterize risk, translation of HQs to population-level attributes for mammals, and risk from COIs for which HQs could not be calculated because TRVs were not identified. A further analysis of COIs without TRVs is presented in the following paragraph to determine the likelihood that they pose a risk to mammals in the Terrestrial Study Area.

9.3.3.1 Chemicals of Interest without Toxicity Reference Values

HQs could not be calculated for mammals for one COI (antimony) due to lack of TRVs (Section 4.2.2; Appendix E), which creates uncertainty in determining the potential for adverse effects associated with this metal. Soil sample concentrations and the COI-specific BTVs are plotted for each of the three soil study data sets to evaluate concentrations of antimony relative to background (Figure 9-13). All site samples from the 2014 UCR Upland Soil and 2015 Bossburg studies and 32 (30 percent) of the samples from the 2012 Ecology Upland Soil Study have concentrations > the BTV. As shown on Figure 9-13, all samples with antimony concentrations > the BTV result in an HQ ≥ 1 for at least one other COPC, except two 2012 Ecology Upland Soil Study samples for which no COPCs have concentrations resulting in HQs ≥ 1. This indicates that addressing the risks for COPCs for which HQs could be calculated will likely address any risk posed by antimony. Based on this analysis, the lack of HQ calculations for antimony is not expected to contribute to under- or overestimates of risk to mammals.

9.3.3.2 Metal Interactions

There is uncertainty in the approach for assessing risks to mammals related to possible interactions among metals. As discussed in Section 1.3, this issue is one of the basic principles to be considered in assessing risks posed by metals identified by EPA (EPA, 2007a). The complexities associated with exposure to mixtures of metals contribute significant uncertainty to the evaluation of risks to mammals from exposure to metals.

The use of single-chemical HQs does not account for potential changes in toxicity caused by the presence of chemical mixtures. Interactions among metals occur by competition for binding locations on specific enzymes or on cellular receptors during the processes of absorption, excretion, or sequestration at the target site (EPA, 2007a). Depending on the type of toxicological interaction (e.g., additivity, antagonism, potentiation, or synergism) and the respective exposures for the metals, the single-chemical HQ may overestimate or underestimate the potential for adverse effects.

Table 9-7 identifies locations where multiple COPCs exceed soil benchmarks protective of mammals. While the specific toxic effects of these exceedances cannot be predicted without site-specific toxicity testing, locations with exceedances of multiple COPCs generally pose a greater risk to mammals than those with fewer exceedances, with higher EDxs and HQs indicating higher risks.

9.3.3.3 Metal Essentiality

The fact that some metals are essential for maintaining proper health of mammals also contributes uncertainty in the mammal risk estimates. As discussed in Section 1.2, this issue is one of the basic principles to be considered in assessing risks posed by metals identified by EPA (EPA, 2007a). Organisms adapt mechanisms of acquiring limited essential metals and limiting or sequestering overabundant essential metals as well as non-essential metals that mimic essential metals physiologically. Of the COPCs for mammals, chromium, copper, iron, molybdenum, selenium, and zinc are essential to animals (EPA, 2007a). Different mammal species may have different nutritional requirements or optimal conditions for essential metals, and the extent to which the nutritional requirements of the test mammal species are representative of the mammal species that occur within the Terrestrial Study Area is uncertain. This uncertainty is unlikely to result in over- or underestimates of risk.

9.4 Risk Description

As described in Section 5.3, the risk description integrates the findings from the Risk Estimation (Section 9.1), Comparison to Background (Section 9.2), and uncertainty (Section 9.3) sections to describe the risk to mammals resulting from COPCs in the Terrestrial Study Area. The risk description also includes discussion of the spatial scale and the nature and severity of potential adverse effects.

9.4.1 Individual Chemicals of Potential Concern

The following COPCs have been eliminated from further evaluation for the mammal EAEs in the Terrestrial Study Area because unacceptable risks have been ruled out (e.g., HQs < 1, EDx value < ED20, and/or soil concentration ≤ BTV): aluminum, chromium, iron, mercury, molybdenum, and thallium.

COPCs and respective representative species retained for further evaluation include the following:

- Cadmium: Masked shrew, deer mouse
- Copper: Masked shrew
- Lead: Meadow vole, masked shrew, short-tailed weasel, deer mouse
- Selenium: Masked shrew, little brown bat, deer mouse
- Zinc: Masked shrew, deer mouse

The following risk description for mammals focuses on the COPCs and receptors listed in the bullets.

9.4.2 Cadmium

Masked shrew and deer mouse are the only species with HQs ≥ 1.

- HQs
  - **Masked shrew:** Numerous samples from the 2012 Ecology Upland Soil and 2014 UCR Upland Soil studies have concentrations that result in HQs ≥ 1 for survival (63 percent of the 2012 Ecology Upland Soil Study and 64 percent of the 2014 UCR Upland Soil Study samples, respectively) and fewer for reproduction (21 percent of the 2012 Ecology Upland Soil Study and 10 percent of the 2014 UCR Upland Soil Study samples, respectively) (Tables 9-1 and 9-2; Figure 9-2b). Of 2012 Ecology Upland Soil Study samples, 3 percent result in an HQ ≥ 1 for growth. No Bossburg Study samples result in HQs ≥ 1.
  - **Deer mouse:** Of the 2012 Ecology Upland Soil Study, 1 percent of samples have a concentration that results in an HQ ≥ 1 for survival (Tables 9-1 and 9-2; Figure 9-2f).
- **Nature and severity**
  - HQs ≥ 1 correspond to dose-response model estimates ranging from 20 percent to 40 percent reductions in survival for masked shrew (based on reductions in survival of voles exposed to cadmium in diet and water). For deer mouse the single survival HQ ≥ 1 corresponds to an estimated 21 percent reduction in survival based on the same effects data. For masked shrew, HQs ≥ 1 correspond to dose-response model estimates ranging from 20 to 26 percent reductions in growth (based on growth of juvenile rat exposed to dietary cadmium), and 20 to 36 percent reductions in reproduction (based on reduced reproduction in rat exposed to dietary cadmium) (Table 9-2, Table 4-7, and Table 4-10).
- **Spatial distribution:** For masked shrew, samples from the three soil surveys with concentrations resulting in survival HQs ≥ 1 occur along the river valley and adjacent valley terraces, with fewer in the higher elevations in the west and east of the Terrestrial Study Area (Map 9-7). Samples with concentrations resulting in reproduction and growth HQs ≥ 1 occur over a limited spatial extent in the northeastern portion of the Terrestrial Study Area. The single deer mouse HQ ≥ 1 occurs in the northeastern part of the Terrestrial Study Area (Map 9-6).
- **Background:** All site soil concentrations exceed the BTV of 0.74 mg/kg (Figures 9-2a through 9-2f), thus all samples with HQs ≥ 1 also exceed the BTV (Table 9-3).

- **Specific uncertainties:** Uncertainties do not strongly influence risk conclusions. Exposure may be slightly underestimated. Use of the 95 UCL rather than the ICS estimated means for COPC concentrations in soil samples from the 2014 UCR Upland Soil and 2015 Bossburg studies data set DUs slightly increases (by 5 percent) the number of survival HQs ≥ 1 for the 2014 UCR Upland Soil Study data set (Table 9-6). Effects thresholds are based on reasonable conservative estimates. As noted in Table 9-4, the selected survival and reproduction TRVs are based on toxicity studies in which vole and rat, respectively were dosed with soluble forms of cadmium that are likely more bioavailable than the forms that occur in dietary items in the Upland Terrestrial Study Area.
- Conclusions
  - Masked shrew: Cadmium exposure poses risk of reduced survival and reproduction throughout the Terrestrial Study Area. Cadmium poses unacceptable risk to masked shrew and is retained as a COC for the invertivore mammal EAE for evaluation in the RI.
  - Deer mouse: Cadmium exposure poses risk of low reductions in growth over a limited spatial extent in the uplands on the east side of the UCR. Because of the limited extent and low magnitude of predicted effects and use of generally conservative estimates of exposure and effects, cadmium poses negligible risk to deer mouse and the omnivore mammal EAE.

9.4.3 Copper

Masked shrew is the only receptor with copper HQs ≥ 1 (Table 9-1).

- **HQs:** Numerous samples from the three soil study data sets have concentrations that result in HQs ≥ 1 for survival (54 percent of the 2012 Ecology Upland Soil Study, 32 percent of the 2014 UCR Upland Soil Study, and 33 percent of the 2015 Bossburg Study samples, respectively), and somewhat fewer for growth (28 percent of the 2012 Ecology Upland Soil Study, 11 percent of the 2014 UCR Upland Soil Study, and 17 percent of the 2015 Bossburg Study samples, respectively) (Table 8-1; Figure 9-4b).
- **Nature and severity:** HQs ≥ 1 correspond to 20 to 97 percent reductions in growth based on reductions in juvenile pig growth when exposed to dietary lead (Table 9-2, Table 4-7, Table 4-10). EDxs are not available for the survival endpoint (based on the geometric mean of two LOAELs ≥ 20 for 25 percent and 100 percent reductions in juvenile pig survival when exposed to dietary lead) (Table 9-2, Table 4-7, and Table 4-10). The maximum survival HQ is 2.8 (Table 9-2).
- **Spatial distribution:** Samples from the three soil surveys with concentrations > BTV resulting in survival TRV exceedances occur along the river valley and adjacent valley terraces, with fewer in the higher elevations in the west and east of the Terrestrial Study Area (Map 9-8).
- **Background:** Some samples with concentrations ≤ BTV of 0.740 mg/kg result in HQs ≥ 1 (42 percent of the 2012 Ecology Upland Soil Study, 27 percent of the 2014 UCR Upland Soil Study, and 17 percent of the 2015 Bossburg Study samples, respectively) and some samples with concentrations > BTV result in HQs ≥ 1 (12 percent of the 2012 Ecology Upland Soil Study, 5 percent of the 2014 UCR Upland Soil Study, and 17 percent of the 2015 Bossburg Study samples, respectively) (Figures 9-4a through 9-4f, Table 9-3).

- **Specific uncertainties:** Uncertainties may somewhat underestimate risks. Use of the 95 UCL rather than the ICS estimated means for copper concentrations in soil samples from the 2014 UCR Upland Soil Study and 2015 Bossburg Study data set DUs increases the number of growth HQs ≥ 1 by 19 percent and 17 percent, respectively (Table 9-6). Effects thresholds are based on reasonable conservative estimates with uncertainties contributing to both under- and overestimates of risk. The survival TRV is based on the geometric mean of two studies with relatively severe effects, including 25 percent and 100 percent reductions in survival of juvenile pigs; thus, lesser impacts on survival may occur at lower concentrations so risks may be somewhat underestimated. The TRV studies used soluble forms of the metal (copper sulfate) that are likely more bioavailable than the forms that occur in dietary items in the Upland Terrestrial Study Area so risks may be somewhat overestimated.
- **Conclusions:** Copper exceeds background and poses risk of reduced survival throughout a substantial portion of the Terrestrial Study Area and poses risk of reduced growth over a more limited spatial extent. Copper poses unacceptable risk to masked shrew and the invertivore mammal EAE.

9.4.4 Lead

Meadow vole, masked shrew, short-tailed weasel, and deer mouse all have some samples with concentrations resulting in HQs ≥ 1 for lead. The representative species with the greatest number of HQs ≥ 1 for lead (from highest to lowest) are masked shrew, deer mouse, meadow vole, and short-tailed weasel (Table 9-1).

- **HQs**
  - **Masked shrew:** Numerous samples from the three soil study data sets have concentrations that result in HQs ≥ 1 for reproduction (77 percent of both the 2012 Ecology Upland Soil Study and 2014 UCR Upland Soil Study, and 67 percent of the 2015 Bossburg Study samples, respectively), and survival (63 percent of the 2012 Ecology Upland Soil Study, 47 percent of the 2014 UCR Upland Soil Study, and 50 percent of the 2015 Bossburg Study samples, respectively), whereas relatively few samples result in HQs ≥ 1 growth (16 percent of the 2012 Ecology Upland Soil Study, 1 percent of the 2014 UCR Upland Soil Study, and 17 percent of the 2015 Bossburg Study soil samples, respectively) (Tables 9-1 and 9-2; Figure 9-6b).
  - **Deer mouse:** Several samples from the three soil study data sets have concentrations that result in HQs ≥ 1 for reproduction (24 percent of the 2012 Ecology Upland Soil Study samples, 6 percent of the 2014 UCR Upland Soil Study samples, and 17 percent of the 2015 Bossburg Study samples, respectively), whereas few result in HQs ≥ 1 for survival (8 percent of the 2012 Ecology Upland Soil Study and 17 percent of the 2015 Bossburg Study sample locations), and 17 percent of the 2015 Bossburg Study samples exceed the growth LOAEL ≥ 20 (Tables 9-1 and 9-2; Figure 9-6f).
  - **Meadow vole:** Several samples from the three soil study data sets have concentrations that result in HQs ≥ 1 for reproduction (17 percent of the 2012 Ecology Upland Soil Study samples, 1 percent of the 2014 UCR Upland Soil Study samples, and 16 percent of the 2015 Bossburg samples, respectively), whereas few result in HQs ≥ 1 for survival (7 percent of the 2012 Ecology Upland Soil Study and 17 percent of the 2015 Bossburg Study sample locations), and one sample exceeds the growth LOAEL ≥ 20 (Tables 9-1 and 9-2; Figure 9-6a).
  - **Short-tailed weasel:** Only a single sample each from the 2012 Ecology Upland Soil Study and the 2015 Bossburg Study have concentrations resulting in HQs ≥ 1 for the reproduction endpoint.

- **Nature and severity:** HQs ≥ 1 correspond to dose-response model estimates ranging from 20 to 99 percent reductions in survival for masked shrew and up to 73 percent for meadow vole and deer mouse (based on reductions in rabbit survival when exposed to lead via oral gavage) (Table 9-2, Table 4-7, and Table 4-10). EDxs are not available for the reproduction endpoint (based on the Eco-SSL, which is the highest bounded NOAEL) or the growth endpoint (based on a LOAEL ≥ 20 for reduced growth of juvenile rabbits exposed to lead via oral gavage). Maximum reproduction HQs are 15, 4.5, 4.3, and 1.3 for masked shrew, meadow vole, deer mouse, and short-tailed weasel, respectively (Table 9-2, Table 4-7, and Table 4-10). The maximum masked shrew growth HQ is 3.5 (Table 9-2). The maximum short-tailed weasel reproduction HQs is 1.3 (Table 9-2).
- **Spatial distribution:** The samples with concentrations resulting in HQs ≥ 1 for masked shrew for the reproduction endpoint occur throughout the Terrestrial Study Area, whereas samples resulting in exceedances of the survival endpoint occur along the river valley and adjacent valley terraces, with fewer in the higher elevations in the west and east of the Terrestrial Study Area and those resulting in exceedance of the growth endpoint occur close to the UCR near the USA-Canada border and in the 2015 Bossburg Study sampling area (Map 9-10). Samples with concentrations resulting in HQ ≥ 1 for deer mouse and meadow vole are located near the northern part of the Terrestrial Study Area along the river valley, for deer mouse extending further along the valley bottom, and in a single location in the 2015 Bossburg Study sampling area (Map 9-11). Of the two samples with concentrations resulting in HQs ≥ 1 for short-tailed weasel, one is in the uplands in the northeast of the Terrestrial Study Area and the other is in the 2015 Bossburg Study sampling area (Map 9-12).
- **Background:** All site soil concentrations exceed the BTV of 27.2 mg/kg (Figure 96a through 9-6f), thus all locations with HQ ≥ 1 are > the BTV (Table 9-3).
- **Specific uncertainty:** Uncertainties do not strongly influence risk conclusions. Exposure may be slightly underestimated. Soil RBA estimated using IVBA may slightly underestimate exposure for deer mouse, masked shrew, and meadow vole. Variability in site-specific RBAs could increase the number of HQs ≥ 1 by up to 6 percent (Table 9-5). Use of the 95 UCL rather than the ICS estimated means for COPC concentrations in soil samples from the 2014 UCR Upland Soil Study and 2015 Bossburg Study data set DUs slightly increases (by 7 percent) the number of growth HQs ≥ 1 for the 2014 UCR Upland Soil Study data set (Table 9-6). Effects thresholds may somewhat underestimate effects. As noted in Table 9-4, the selected reproduction TRV (which is the most sensitive endpoint) is the Eco-SSL because the reproduction TRV selected in Appendix E is lower than the Eco-SSL, indicating that ≥ 20 percent effects on reproduction may occur at lower doses, contributing to potential underestimates of risk. The survival and growth TRVs are based on toxicity studies in which rabbits were dosed via oral gavage with soluble forms of lead that are likely more bioavailable than the forms that occur in dietary items in the Upland Terrestrial Study Area, contributing to potential overestimates of risk.
- **Conclusions**
  - Masked shrew, deer mouse, and meadow vole: Lead exposure poses risk of reduced reproduction throughout most of the Terrestrial Study Area for masked shrew, throughout a substantial portion of the study area for deer mouse, and in a more limited area for meadow vole. Risk of reduced survival occurs over a more limited extent. Lead poses unacceptable risk to masked shrew and the invertivore mammal EAE, deer mouse and the omnivore EAE, and meadow vole and the herbivore EAE.
  - Short-tailed weasel: Lead exposure poses negligible risk due to the limited extent of exceedances.

9.4.5 Selenium

Masked shrew, little brown bat, and deer mouse (in order of the number of exceedances) all have some samples with concentrations resulting in HQs ≥ 1 for the survival and growth endpoints (Tables 9-1 and 9-2; Figure 9-9b, 9-9c, and 9-9f).

- HQs
  - **Masked shrew:** There are several samples from the three soil study data sets that have concentrations resulting in HQs ≥ 1 for growth (12 percent of the 2012 Ecology Upland Soil Study, 16 percent of the 2014 UCR Upland Soil Study, and 17 percent of the 2015 Bossburg Study samples, respectively), whereas 6 percent and 1 percent of the 2012 Ecology Upland Soil Study and 2014 UCR Upland Soil Study samples, respectively have concentrations resulting in survival HQs ≥ 1 (Table 9-1; Figure 9-9b).
  - **Little brown bat:** Growth HQs ≥ 1 correspond to concentrations in 5 percent of the 2012 Ecology Upland Soil Study samples, and 1 percent of 2014 UCR Upland Soil Study samples. A single 2012 Ecology Upland Soil Study sample and a single 2014 UCR Upland Soil Study sample have concentrations resulting in survival HQs ≥ 1 (Table 9-1; Figure 9-9c).
  - **Deer mouse:** Growth HQs ≥ 1 correspond to concentrations in 3 percent of the 2012 Ecology Upland Soil Study samples and 1 percent of 2014 UCR Upland Soil Study samples (Table 9-1; Figure 9-9f). A single survival HQ ≥ 1 corresponds to the concentration in a single 2012 Ecology Upland Soil Study sample.
- **Nature and severity:**
  - HQs ≥ 1 correspond to 20 percent to 91 percent reductions in growth, with median reductions of 22 percent to 52 percent growth reduction for the different species and soil studies based on toxicity data for growth of rats (Table 9-2, Table 4-7, and Table 4-10). EDxs are not available for the survival endpoint (based on toxicity data for survival of rats exposed to dietary selenium) (Table 9-2, Table 4-7, and Table 4-10). Maximum survival HQs are 5.6, 2.2, and 1.6 for the masked shrew, little brown bat, and deer mouse, respectively (Table 9-2).
- **Spatial distribution:** Samples from the three soil surveys with concentrations resulting in growth HQs ≥ 1 for masked shrew occur along the river valley and adjacent valley terraces and in one sample location in the 2015 Bossburg Study area, with more detected samples resulting in HQs ≥ 1 occurring on the east side of the UCR and in the northeastern portion of the Terrestrial Study Area (Map 9-15). Samples with concentrations resulting in survival HQs ≥ 1 for masked shrew and growth and survival HQs ≥ 1 for the little brown bat and deer mouse occur over a limited extent in the northeastern portion of the Terrestrial Study Area (Maps 9-13, 9-14, and 9-15).
- **Background:** All site soil concentrations exceed the BTV of 0.098 mg/kg (Figures 9-9a through 9-9f), thus all locations with HQ ≥ 1 are > the BTV (Table 9-3).

- **Specific uncertainty:** The soil concentrations in 96 out of 170 of the samples resulting in HQs ≥ 1 are below the MDLs, thus, actual concentrations in these samples are likely lower than the reported concentration (the MDL) (Table 9-7). Use of the 95 UCL rather than the ICS estimated means for COPC concentrations in soil samples from the 2014 UCR Upland Soil Study and 2015 Bossburg Study data set DUs slightly increases (by 6 percent) the number of growth HQs ≥ 1 for the 2014 UCR Upland Soil Study data set (Table 9-6). Little brown bat has a large foraging area and as illustrated on Map 9-14, three samples within the home range of the little brown bat in the northeastern part of the Upland Terrestrial Study Area have selenium concentrations resulting in growth EDx ≥ 20 percent and ≥ 50 percent suggesting that average exposure in this area poses elevated risk to little brown bat. Effects thresholds are based on reasonable conservative estimates. The growth TRV, which is the most sensitive endpoint, is based on a modeled ED20 for pig growth. The survival TRV is based on a LOAEL ≥ 20 with an estimated effect level of 62 percent, thus lesser impacts on survival may occur at lower concentrations so risks may be somewhat underestimated. The TRV studies used soluble forms of the metal (sodium selenite and D-selenomethionine), which are likely more bioavailable than the forms that occur in dietary items in the Upland Terrestrial Study Area contributing to potential overestimates of risk.
- **Conclusion**
  - **Masked shrew and little brown bat:** Selenium exposure poses risk of reduced growth to masked shrew throughout a substantial portion of the study area and to little brown bat over a limited spatial extent and poses risk of reduced survival to masked shrew over a limited spatial extent. Selenium poses unacceptable risk to masked shrew and the invertivore mammal EAE and to little brown bat and the aerial insectivore mammal EAE.
  - **Deer mouse:** Because of the limited spatial extent of samples with concentrations resulting in exceedances of the TRVs and generally conservative estimates of exposure deer mouse populations are unlikely to be adversely affected, thus selenium poses negligible risk to deer mouse and the omnivore EAE.

9.4.6 Zinc

Masked shrew and deer mouse (in order of the number of exceedances) have HQs ≥ 1 (Table 9-1).

- **HQs**
  - Masked shrew growth and reproduction HQs ≥ 1 correspond to concentrations occurring in 67 percent of the 2012 Ecology Upland Soil Study and 72 percent of the 2014 UCR Upland Soil Study samples. Survival HQs ≥ 1 correspond to concentrations occurring in 17 percent of the 2012 Ecology Upland Soil Study and 2 percent of the 2014 UCR Upland Soil Study sample locations (Tables 9-1 and 9-2; Figure 9-11b). No 2015 Bossburg Study soil samples have concentrations resulting in HQs ≥ 1 for any receptor or endpoint.
  - Deer mouse growth and reproduction HQs ≥ 1 correspond to concentrations occurring in 8 percent of the 2012 Ecology Upland Soil Study and 1 percent of the 2014 UCR Upland Soil Study sample locations.
- **Nature and severity:** EDxs are not available for the survival, growth, or reproduction endpoints. The reproduction and growth TRVs are both based on the same mammalian Eco-SSL (based on the geometric mean of 25 NOAELs for reproduction and 44 NOAELs for growth). Maximum HQs are 6.0 and 1.7 for masked shrew and deer mouse, respectively for the reproduction and growth endpoints. The maximum masked shrew survival HQ is 2.4 based on the geometric mean of three LOAELs ≥ 20 corresponding to 25 percent to 37.5 percent reduced survival of juvenile pigs exposed to dietary zinc (Table 4-7, Table 4-10, Table 9-2, and Table 9-4).

- **Spatial distribution:** For masked shrew, samples resulting in estimated doses at or greater than the growth and reproduction TRVs are distributed throughout the Terrestrial Study Area, principally along the river valley and adjacent valley terraces, with fewer in the higher elevations in the west and east of the Terrestrial Study Area (Map 9-17). For masked shrew survival and deer mouse growth and reproduction, samples resulting in HQs ≥ 1 occur near the UCR on both sides of the river near the US-Canada border and in the northeastern portion of the Terrestrial Study Area (Map 9-16).
- **Background:** All site soil concentrations exceed the BTV of 111 mg/kg (Figures 9-11a through 9-11f), thus all locations with HQ ≥ 1 are > the BTV (Table 9-3).
- **Specific uncertainty:** Use of the 95 UCL rather than the ICS estimated means for COPC concentrations in soil samples from the 2014 UCR Upland Soil Study and 2015 Bossburg Study data set DUs slightly increases (by 10 percent) the number of growth HQs ≥ 1 for the 2014 UCR Upland Soil Study data set (Table 9-6). Uncertainty in the selected TRVs may somewhat underestimate the potential for adverse effects. The reproduction and growth TRVs are based on the mammalian Eco-SSL (based on reproduction and growth NOAELs) because the growth and reproduction TRVs selected in Appendix E are lower than the Eco-SSL, indicating that ≥ 20 percent effects on reproduction may occur at lower doses, contributing to potential underestimates of risk (Table 9-4). The survival and growth TRVs are based on insoluble forms of zinc, which are likely more like zinc in mammal diets at the site, whereas the reproduction TRV is based on a highly soluble form of zinc, which likely has greater bioavailability than the zinc in mammal diets at the site.
- **Risk Conclusions**
  - **Masked shrew:** Zinc exposure poses risk of reduced growth and reproduction throughout a substantial portion of the Terrestrial Study Area and risk of reduced survival over a limited spatial extent. Zinc poses unacceptable risk to masked shrew and the invertivore EAE.
  - **Deer mouse:** Because of the limited spatial extent of samples with concentrations resulting in exceedances of the TRVs generally low magnitude of potential effects on the growth and reproduction endpoints, the extent and magnitude of risk is sufficiently small that the deer mouse populations are unlikely to be adversely affected, thus zinc poses negligible risk to deer mouse and the omnivore EAE.

9.4.7 Multiple Metals

The preceding discussion focuses on individual COPCs, with limited discussion of multiple metals with HQ ≥ 1. This section explores locations and portions of the site having the greatest potential for the combined effects of multiple metals. Table 9-7 identifies those sample locations with HQ ≥ 1, listing all of the COCs exceeding the most sensitive endpoint for the most sensitive species (masked shrew for all COCs) at each location.

Maps 9-18a through 9-18d combine results for the mammal COCs, cadmium, copper, lead, selenium, and zinc with results for each COC shown in a cluster of five symbols at each location. The HQ comparisons are indicated with colors and different sized symbols as described in Section 9.1 for Maps 9-1 through 9--17. On Maps 9-18a through 9-18d, the symbols are arranged in two rows, with cadmium and copper HQs represented by right-side-up and upside-down triangles in the top row, and lead, selenium, and zinc HQs represented by a circle, hexagon, and square, respectively, in the bottom row.

As illustrated in Table 9-7 and on Maps 9-18a through 9-18d, of the 222 locations where at least one HQ exceedance occurs, 175 locations have concentrations of multiple metals that exceed their respective TRVs. HQs ≥ 1 for cadmium only occur in locations where other COCs also exceed their TRVs. Copper, lead, selenium, and zinc exceed their respective TRVs in locations where no other COCs have

HQ ≥ 1 in 7, 32, 2, and 7 locations, respectively, whereas each of these COCs has HQs ≥ 1 in locations where multiple COCs have HQs ≥ 1 in 98, 163, 34, 165 locations, respectively. In contrast, in one sample, the lead concentration results in an HQ of 15, whereas concentrations of all other COPCs result in HQs < 1 for that sample.

As described in Section 9.3, the simultaneous exposure of mammals to elevated concentrations of multiple metals results in complex interactions, the effects are difficult to predict, and assumption of independent action may result in underestimates of the combined risks. For the purposes of this risk assessment, it is assumed that locations with multiple COCs exceeding benchmarks pose a greater risk to mammals than those locations with fewer exceedances and that risk at a specific location is at least as great as that associated with the COC with the highest HQ.

9.5 Mammal Risk Conclusions

This Upland BERA evaluates the following risk question related to mammals within the Terrestrial Study Area:

- Do the daily doses of COPCs received by mammals (represented by guilds focused on specific mammalian species) from consumption of the tissues of prey, plants, and soil in the Terrestrial Study Area exceed the TRVs for survival, growth, or reproduction of mammals such that adverse effects to the local population are expected?

Based on the analyses discussed in the risk characterization, lead, cadmium, and zinc are COCs that present the greatest and most widespread risk to mammals in the Terrestrial Study Area, based on widespread exceedances of TRVs. The endpoint resulting in the greatest number of HQs ≥ 1 is reproduction followed closely by both reduced growth and survival. Copper and selenium pose unacceptable risk to masked shrew over a smaller portion of the Terrestrial Study Area. COPCs and respective EAEs (and representative species) retained as COCs include:

- Cadmium: invertivore EAE (masked shrew)
- Copper: invertivore EAE (masked shrew)
- Lead: herbivore EAE (meadow vole), invertivore EAE (masked shrew), and omnivore EAE (deer mouse)
- Selenium: invertivore EAE (masked shrew), aerial insectivore EAE (little brown bat)
- Zinc: invertivore EAE (masked shrew)

The uncertainty analysis identifies several factors that contribute to over- and underestimates of the potential for adverse effects. Exposure and effects data were selected using robust methods to result in reasonable conservative estimates of the severity of effects resulting from site exposure. Risk estimates are based on reasonable conservative estimates of exposure and effects estimates are based on robust toxicity data sets for all COCs. The cadmium data set incorporates dose-response information into the risk estimates, which increases the reliability and relevance of the analysis. Copper and selenium TRVs are based on LOAELs, which are unlikely to underestimate the potential for adverse effects. Lead and zinc TRVs are based on the Eco-SSLs, which are meant to be conservative screening values; however, lower TRVs were identified following robust site-specific data analyses indicating that the extent and magnitude of effects on sensitive species could be underestimated for these COCs. Overall, the underlying data and uncertainties result in a moderate degree of confidence in the risk prediction for all COCs (cadmium, copper, lead, selenium, and zinc). Because there are no site-specific measures of effects, risk predictions remain somewhat uncertain.

The remaining COPCs for mammals, aluminum, chromium, iron, mercury, molybdenum, and thallium present negligible risk and are not carried forward as COCs. Uncertainty is not likely to underestimate risk and there is a high degree of confidence that these COPCs pose negligible risk.

10. Summary and Conclusions

The objective of the Upland BERA is to assess risk from hazardous substances in soils to EAEs within the Terrestrial Study Area of the site, under both current conditions and expected future conditions assuming no steps are taken to remediate the environment. EAEs, risk questions, and measures of exposure and effects used in the risk assessment are described in Section 2.7. The three data sets used in the Upland BERA, the 2014 UCR Upland Soil Study (TAI, 2015), the 2012 Ecology Upland Soil Study (Ecology, 2013), and the 2015 Bossburg Study (TAI, 2016), are described in Section 3. The approach to exposure and effects assessment and risk analyses is described in Section 4, and the methods used to characterize risk, including risk estimation, comparisons to background, uncertainty analysis, risk description, and risk conclusions, are provided in Section 5.

10.1 Summary of Findings

The risk characterizations for all of the receptor groups evaluated identify several COPCs that pose unacceptable risk to one or more EAEs within the receptor group. Nine of the 19 COPCs evaluated are determined to be COCs that present unacceptable risk to at least one representative ecological community or species selected as an EAE: arsenic, barium, cadmium, copper, lead, manganese, mercury, selenium, and zinc. The outcome for all COPCs is presented in Table 10-1. Ten of the 19 COPCs evaluated are found to pose negligible risk to EAEs so are not identified as COCs: aluminum, antimony, chromium, cobalt, iron, molybdenum, nickel, silver, thallium, and vanadium. Risk characterizations could not be completed for one or more EAE for antimony, beryllium, and thallium, due to the absence of sufficient toxicity information for some receptor groups so risks posed by these COI/COPCs are uncertain. Uncertainty analyses indicate that areas with elevated concentrations of these metals co-occur with identified COCs suggesting that the extent of risks from these metals is captured by that of the identified COCs. Although risk due to interaction of elevated concentrations of multiple metals cannot be quantitatively assessed with the available data, for the purposes of this risk assessment, it is assumed that locations with multiple COCs with HQs ≥ 1 pose a greater risk to receptors than those locations with fewer exceedances and that risk at a specific location is at least as great as that associated with the COC with the highest HQ.

10.1.1 Chemical of Concern and Ecological Assessment Endpoint Pairs

For plant and invertebrate receptor groups, the HQs presented in Table 10-1 for each COC are based on comparison of soil concentrations with BABs protective of 95% of plant or invertebrate species if available, and with Eco-SSL or SSLs (below which no adverse effects are expected) if BABs were not available. For the bird and mammal receptor groups, HQs calculations are based on estimates of average daily dietary COPC doses compared with dietary dose-based TRVs protective of ≥20% increases in the incidence of survival, growth, or reproduction in sensitive species. Table 10-1 presents bird and mammal HQs for only the most sensitive endpoint (survival, growth, or reproduction) for the most sensitive species. COPCs and respective EAEs (and respective representative species) retained as COCs include:

- Arsenic: plants EAE
- Barium: invertebrates EAE
- Cadmium: invertivore birds EAE (American robin) and invertivore mammal EAE (masked shrew)
- Copper: invertivore mammals EAE (masked shrew)
- Lead: plants EAE, invertivore birds EAE (American robin), herbivore mammals EAE (meadow vole), invertivore mammals EAE (masked shrew), and omnivore mammals EAE (deer mouse)

- Manganese: plants EAE, invertebrates EAE
- Mercury: invertivore birds EAE (American robin)
- Selenium: plants EAE, invertivore mammals EAE (masked shrew), aerial insectivore mammals EAE (little brown bat)
- Zinc: plants EAE, invertebrates EAE, invertivore birds EAE (American robin), carnivore birds EAE (black-capped chickadee).

All COPCs (including the COCs) pose negligible risks to the herbivore birds EAE (California quail), aerial insectivore birds EAE (tree swallow), and carnivore birds EAE (American kestrel).

10.1.2 Uncertainty Evaluation

Uncertainty analyses for each line of evidence in the risk assessment consider the uncertainty and variability of the data to determine whether risk analyses may over- or underestimate the potential for adverse effects. The exposure estimates for each receptor group are based on limited site-specific data and rely on several assumptions. As such, it is unknown whether many of the uncertainties identified for each receptor group contribute to over- or underestimates of Site exposures. In general, reasonable conservative assumptions were made to ensure that exposure is not underestimated. Additionally, in general, the ability of the TRVs used to estimate site-specific adverse effects is unknown. Several site-specific factors influence the bioavailability and toxicity of metals and the TRVs used to estimate the potential severity of site exposures were selected using a robust process to result in reasonable conservative estimates of effects.

Quantitative uncertainty analyses were performed to evaluate some uncertainties for each receptor group (Sections 6 through 8). Sources of uncertainty that were quantitatively evaluated included the following:

- Uncertainty associated with using a single incremental composite sample at most locations or the maximum of triplicate ICS samples in a subset of locations (2014 UCR Upland Soil Study and the 2015 Bossburg Study) rather than using an estimated 95 UCL. This was evaluated for all receptor groups.
- Uncertainty around median BABs as soil ecological screening benchmarks for plant and invertebrates determined from the results of the predicted no-effect concentration calculator.
- Uncertainty associated with the relative bioavailability assessments for bird and mammal dietary exposure.
- Uncertainty associated with earthworm bioaccumulation models for dietary exposure to birds.
- Uncertainty associated with several dietary exposure TRVs and effects assumptions for lead and mercury for birds.

The quantitative uncertainty analyses indicate that none of the sources of uncertainty are substantial enough to change the determination of unacceptable risk from any COC for all EAEs evaluated. In most cases, the number or distribution of HQs ≥ 1 could vary under alternative scenarios (such as use of the 95 UCL instead of mean EPC), but in no case would COPCs with all HQs < 1 under the baseline scenario have HQs ≥ 1 (or vice versa) under the alternative scenarios used to evaluate uncertainty.

10.1.3 Chemical of Concern Summary

Maps showing HQ ranges for each receptor group (plants, invertebrates, birds, and mammals) for each COC are presented on Maps 10-1 through 10-9. Multiple symbols are shown at each location, with symbol shape indicating receptor group, symbol size and color indicating HQ ranges, closed symbols indicate exceedance of the BTV and open symbols indicate concentrations less than the BTV. Maps 10a through 10d show the greatest HQ at each sample location among the nine COCs, endpoints, and receptors for each assessment endpoint.

For each receptor group, Table 10-2 presents the COC, with the greatest HQ for each location and for mammals and birds, the receptor and endpoint resulting in the greatest HQ. For both Map 10-10 and Table 10-2, at each location the maximum HQ among COCs with concentrations exceeding their respective BTV is shown; if no COC concentrations exceed respective BTVs, then the maximum HQ among all COCs is shown. As illustrated in the maps, HQs ≥ 1 are present in samples on the periphery of the Terrestrial Study Area suggesting that risks likely extend beyond the Terrestrial Study Area. Risks associated with each COC are as follows:

- Arsenic: Arsenic poses unacceptable risk to plants based on exceedances of the Eco-SSL, which is based on the plant growth endpoint (Table 10-1). Arsenic concentrations are greater than the BTV and have HQ ≥ 1 in many locations (Table 10-2, Map 10-1), particularly in the north-central portion of the Terrestrial Study Area near the U.S.-Canada border. Arsenic poses negligible risk to invertebrates, birds, or mammals.
- Barium: Barium poses unacceptable risk to invertebrates based on exceedances of the Eco-SSL, which is based on the threshold for 20 percent reductions in invertebrate reproduction (Table 10-1). Because the barium BTV is close to the Eco-SSL, most locations with HQs ≥ 1 also exceed the BTV. Sample locations with barium HQs ≥ 1 for invertebrates are scattered throughout the central and northern portion of the Terrestrial Study Area (Map 10-2). Although three locations have HQs ≥ 1 for plants, because the HQs are low and the Eco-SSL is conservative, barium poses negligible risk to plants. Barium poses negligible risk to birds and mammals because all samples have concentrations resulting in HQs < 1.0.
- Cadmium: Cadmium poses unacceptable risk to birds and mammals (Table 10-1). There is risk of reduced growth and reproduction to invertivore birds (American robin) with EDxs ≥ 20 throughout the Terrestrial Study Area (Map 10-3). Cadmium poses risk of reduced survival, and to a lesser extent reduced growth, with EDxs ≥ 20 for invertivore mammals (masked shrew) throughout the Terrestrial Study Area. All site soil concentrations exceed the BTV. Cadmium is not a COPC for plants or invertebrates and poses negligible risk to these receptor groups.
- Copper: Copper poses unacceptable risk of reduced survival and growth to invertivore mammals (masked shrew) (Table 10-1). Sample locations with HQs ≥ 1, and EDxs ≥ 50, with copper concentrations exceeding the BTV, occur throughout the Terrestrial Study Area but are most frequent near the river in the northern portion of the Terrestrial Study Area (Map 10-4). Copper poses negligible risk to plants and birds because all samples have concentrations resulting in HQs < 1.0 and poses negligible risk to invertebrates because few samples result in HQs ≥ 1.
- Lead: Lead poses unacceptable risk to plants, birds, and mammals. Lead concentrations result in BAB HQs ≥ 1 for reduced plant growth in approximately one-fourth of sample locations in the Terrestrial Study Area (Table 10-1). Lead poses unacceptable risk to invertivore birds (American robin) based on concentrations resulting in HQs ≥ 1 or EDxs ≥ 20 throughout a substantial portion of the Terrestrial Study Area and are greatest in the center of the Terrestrial Study Area near the U.S.-Canada border (Map 10-5) where concentrations result in HQs ≥ 5 for reproduction and EDxs ≥ 50 for survival. Lead poses unacceptable risk to multiple mammal feeding guilds including invertivores (masked shrew), omnivores (deer mouse), and herbivores (meadow vole). For mammals, lead exposure poses risk of reduced reproduction throughout most of the Terrestrial Study Area for masked shrew, throughout a substantial portion of the study area for deer mouse, and in a more limited area for meadow vole. Risk of reduced survival in mammals occurs over a more limited extent. Lead is not a COPC for soil invertebrates so poses negligible risk to this receptor group.

- Manganese: Manganese poses unacceptable risk to plants and invertebrates in the Terrestrial Study Area (Table 10-1). Manganese concentrations result in HQs ≥ 1 for the Eco-SSL (based on reduced growth of plants) in nearly all sample locations and HQs ≥ 1 for the Eco-SSL (based on reduced reproduction in invertebrates) in about three-fourths of sample locations (Map 10-6). The BTV is five- and three-fold greater than the plant and invertebrate Eco-SSLs Roughly one-third of sample locations have concentrations resulting in both HQs ≥ 1 for plants and invertebrates and exceedance of the BTV (Map 10-6). Manganese is not a COPC for birds and mammals so poses negligible risk to these receptor groups.
- Mercury: Mercury poses unacceptable risk to birds (Table 10-1). Mercury has dietary dose EDxs ≥ 20 for invertivore bird (American robin) reproduction at numerous locations in the Terrestrial Study Area. Locations with EDxs ≥ 20 for reproduction and concentrations > BTV occur primarily in the northern portion of the Terrestrial Study Area (Map 10-7). An evaluation of alternative more realistic assumptions regarding mercury exposure and effects on bird reproduction indicates that American robin risk is of lower magnitude and smaller extent than risk estimates illustrated in Tables 10-1, 10-2, and Map 10-7 indicate but that reproduction may be adversely affected over a still substantial area in the northern portion of the Terrestrial Study Area (Section 8.4). Map 8-21 illustrates the distribution of HQs ≥ 1 using the alternative assumptions. Mercury is not a COPC for plants and invertebrates so poses negligible risk to these receptor groups and poses negligible risk to mammals because all samples have concentrations resulting in HQs < 1.0.
- Selenium: Selenium poses unacceptable risk to plants and mammals (Table 10-1). Sample locations with selenium concentrations resulting in HQs ≥ 1 for plants (based on the Eco-SSL representing plant growth) are scattered throughout the Terrestrial Study Area and are concentrated near the U.S.-Canada Border on the east side of the river (Map 10-8). Selenium HQs ≥ 1 for mammal invertivores (masked shrew) and aerial insectivores (little brown bat) for reduced growth occur primarily on the east side of the river between Northport and the U.S.-Canada Border. In some cases, selenium concentrations reported in the 2012 Ecology Upland Soil Study that result in HQs ≥ 1 are below analytical detection limits. This adds uncertainty to the risk characterization for a portion of the 2012 Ecology Upland Soil Study locations (nondetects are shown with hashed symbols on Map 10-8) but TRV exceedances by detected concentrations are of sufficient magnitude and extent to result in unacceptable risk. Selenium is not a COPC for invertebrates. For birds, HQs ≥ 1 occur over a limited spatial extent and uncertainties (including elevated detection limits in some samples and TRVs based on more bioavailable forms of selenium than likely occur in natural bird diets) likely result in overestimates of the magnitude of risk. Thus, selenium poses negligible risk to invertebrates and birds.
- Zinc: Zinc poses unacceptable risk for all four receptor groups (Table 101). Zinc has HQs ≥ 1 in about one-third of locations for multiple bird feeding guilds for principally reduced growth and reproduction but also reduced survival in some locations; about two-thirds of locations for invertivore mammal for reduced growth and reproduction; three quarters of locations for reduced plant growth, and in approximately one-third of locations for reduced growth of invertebrates (Table 10-1, Map 10-9). Zinc concentrations were greater than background in all but a few locations (Map 10-9).

10.1.4 Multiple Metals

Although risk due to interaction of elevated concentrations of multiple metals cannot be quantitatively assessed with the available data, for the purposes of this risk assessment, it is assumed that locations with multiple COCs with HQs ≥1 pose a greater risk to receptors than those locations with fewer exceedances and that risk at a specific location is at least as great as that associated with the COC with the highest HQ. Tables (Tables 8-9 and 9-7) and maps (Maps 6-12a through 6-12d, 7-8a through 7-8d, 8-22, and 9-18) of

COC co-occurrence indicate considerable spatial overlap of HQs ≥ 1 or EDxs ≥ 20 for multiple COCs for each receptor group.

10.1.5 Background

Locations with and soil COC concentrations ≤ BTVs are summarized along with HQs for the four receptor groups in Maps 10-1 through 10-9. Concentrations of the COCs copper, manganese, and mercury at some locations are lower than the BTV and result in HQs ≥ 1. However, all nine COCs have concentrations > BTV resulting in HQs ≥ 1 or EDxs ≥ 20 in multiple locations. For cadmium, lead, and zinc, all or nearly all locations in the Terrestrial Study Area are greater than the BTVs.

10.2 Conclusions

Nine COCs in the UCR Terrestrial Study Area present unacceptable risk to one or more EAE representative of site plants, invertebrates, birds, and mammals. For each of the receptor groups, at least one COC-EAE pair has HQs ≥ 1 in nearly every sample location (Maps 10-10a through 10-10d). For birds and mammals, HQs ≥ 1 and concentrations > BTV occur in every sample location for at least one COC. Locations near the U.S.-Canada border generally have greater COC concentrations in soils and more COCs with HQs ≥ 1 for multiple receptor groups, suggesting a greater likelihood and severity of adverse effects in this area.

Cadmium, lead, and zinc present the greatest and most widespread risk in the Terrestrial Study Area. These COCs exceed background in nearly all sample locations in all three soil studies evaluated. Conclusions about risk due to cadmium, lead, and zinc have a moderate level of confidence due to the availability of significant toxicity information, use of BABs and estimates of PAFs for plant and invertebrate benchmarks, and dietary dose-response information for multiple bird and mammal effects endpoints (survival, growth, and reproduction).

Copper presents widespread unacceptable risk primarily to invertivore mammals. The risk characterization has a similar level of confidence to that of cadmium, lead, and zinc, but copper concentrations are less elevated relative to background. Locations with HQ ≥ 1 or ED ≥ 20 occur primarily in the northern portion of the Terrestrial Study Area.

Mercury presents unacceptable risk to invertivore birds (American robin) in the northern portion of the Terrestrial Study Area. The risk characterization has a similar level of confidence to that of cadmium, copper, lead, and zinc, due to the availability of a robust toxicity data set, although mercury does not have sufficient dose-response information to incorporate dose-response estimates into the risk estimates.

Selenium presents widespread unacceptable risk primarily to invertivore mammals (masked shrew) and to a lesser extent to aerial insectivore mammals (little brown bat). The risk characterization has a similar level of confidence to that of cadmium, lead, and zinc. Locations with HQs ≥ 1 or EDxs ≥ 20 occur occur mainly along the river valley and adjacent valley terraces and in the northeastern most portion of the Terrestrial Study Area.

Arsenic, barium, manganese, and selenium also present unacceptable risk to plant and/or invertebrate EAEs in the Terrestrial Study Area, but the risk characterization has a lower level of confidence due to the lack of BABs and dose-response information.

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