

## APPENDIX P

### ECOLOGICAL RESOURCES AND RISK ANALYSIS

This appendix presents the ecological resources (see Section P.1) at the Hanford Site and lists the plants and animals evaluated in this *Tank Closure and Waste Management Environmental Impact Statement for the Hanford Site, Richland, Washington*. Potential impacts of both airborne releases during operations and groundwater discharges under the various alternatives are evaluated in this appendix. The purpose of the risk analysis is to compare alternatives quantitatively. The modeling and risk methods used to evaluate ecological impacts of the proposed alternatives on terrestrial resources are presented in Section P.2; on aquatic resources, in Section P.3.

Although impacts on ecological resources of air and groundwater releases are considered long-term impacts for the purposes of this environmental impact statement, some would occur in the near future after completion of waste management operations. Short-term impacts on ecological resources are evaluated in Chapter 4. Air emissions and their subsequent deposition on soils would be possible under all action alternatives, as well as the Tank Closure No Action Alternative. Immediately following operations, cumulative soil concentrations of radionuclides and chemicals would be at their maximum levels after accumulating during operations and then attenuating following completion of operations. Thus, the projected impacts represent conservative estimates of the impacts of exposure to contaminated soils in the more distant future. Potential adverse impacts on Columbia River aquatic and riparian resources would be more likely to occur in the more distant future after waste management operations have been terminated and chemical and radioactive constituents have migrated through the groundwater to the Columbia River.

#### P.1 ECOLOGICAL RESOURCES

The ecological resources at the Hanford Site (Hanford) and Idaho National Laboratory (INL) are described in detail in Chapter 3. The scientific names of plant and animal species cited in Chapter 3 and throughout this *Tank Closure and Waste Management Environmental Impact Statement for the Hanford Site, Richland, Washington (TC & WM EIS)* are listed in Table P-1. Species are grouped by common name and are listed in alphabetical order. Although 48 plant communities and land use areas exist on Hanford, they may be grouped into six basic types, with sagebrush-dominated shrublands being the most extensive (see Chapter 3, Figure 3-15). Pristine shrub-steppe habitat is considered a priority habitat by the Washington State Department of Ecology because of its relative scarcity in the state and because it is home to a number of sensitive species. A total of 727 vascular plant, 1,500 insect, 5 amphibian, 10 reptile, 258 bird, and 46 mammal species have been identified on Hanford. Section 3.2.7.4 and Table 3-8 of Chapter 3 provide information on threatened and endangered species occurring at Hanford.

INL lies in a cool desert ecosystem dominated by some of the best-condition shrub-steppe communities in the United States, as reflected by the establishment of the Sagebrush-Steppe Ecosystem Reserve in the north-central part of the site. Although sagebrush communities occupy about 80 percent of INL, a total of 11 plant communities have been identified (see Chapter 3, Figure 3-38). A total of 398 plant taxa and 1,240 insect, 1 amphibian, 11 reptile, 210 bird, and 47 mammal species have been identified on the INL site. Threatened and endangered species present at INL are discussed in Chapter 3, Section 3.3.7.4, and are listed in Table 3-36.

**Table P-1. Scientific Names of Plant and Animal Species**

Common Name	Scientific Name
<b>Plants</b>	
Alkali saltgrass	<i>Distichlis spicata</i>
Big sagebrush	<i>Artemisia tridentata</i>
Bitterbrush	<i>Purshia tridentata</i>
Black greasewood	<i>Sarcobatus vermiculatus</i>
Black locust	<i>Robinia pseudoacacia</i>
Bluebunch wheatgrass	<i>Agropyron spicatum</i>
Bulbous bluegrass	<i>Poa bulbosa</i>
Bullrush	<i>Scirpus</i> sp.
Cattail	<i>Typha</i> sp.
Cheatgrass	<i>Bromus tectorum</i>
Cottonwood	<i>Populus</i> sp.
Crested wheatgrass	<i>Agropyron desertorum (cristatum)</i>
Gray rabbitbrush	<i>Chrysothamnus nauseosus</i>
Green rabbitbrush	<i>Chrysothamnus viscidiflorus</i>
Indian ricegrass	<i>Achnatherum hymenoides</i>
Juniper	<i>Juniperus</i> sp.
Low sagebrush	<i>Artemisia arbuscula</i>
Lupine	<i>Lupinus</i> spp.
Mugwort	<i>Artemisia vulgaris</i>
Mulberry	<i>Morus</i> sp.
Needle-and-thread grass	<i>Stipa comata</i>
Peachleaf willow	<i>Salix amygdaloides</i>
Plantain	<i>Plantago</i> spp.
Pondweed	<i>Potamogeton</i> spp.
Poplar	<i>Populus</i> sp.
Reed canary grass	<i>Phalaris arundinacea</i>
Rigid sagebrush	<i>Artemisia rigida</i>
Rock buckwheat	<i>Eriogonum sphaerocephalum</i>
Rush	<i>Juncus</i> spp.
Russian olive	<i>Elaeagnus angustifolia</i>
Russian thistle	<i>Salsola kali</i>
Sagebrush	<i>Artemisia</i> spp.
Salt rattlepod	<i>Swainsona salsula</i>
Sand dropseed	<i>Sporobolus cryptandrus</i>
Sandberg's bluegrass	<i>Poa sandbergii (secunda)</i>
Scrufpea	<i>Psoralidium tenuiflorum</i>
Sedge	<i>Carex</i> sp.
Siberian elm	<i>Ulmus pumila</i>
Snow buckwheat	<i>Eriogonum niveum</i>
Spike rush	<i>Eleocharis</i> spp.

**Table P-1. Scientific Names of Plant and Animal Species (continued)**

Common Name	Scientific Name
<b>Plants (continued)</b>	
Spiny hopsage	<i>Grayia spinosa</i>
Sycamore	<i>Platanus occidentalis</i>
Thickspike wheatgrass	<i>Agropyron dasystachyum</i>
Threetip sagebrush	<i>Artemisia tripartita</i>
Thymeleaf buckwheat	<i>Eriogonum thymoides</i>
Watercress	<i>Nasturtium</i> sp.
Water smartweed	<i>Polygonum amphibium</i>
Willow	<i>Salix</i> spp.
Winterfat	<i>Krascheninnikovia lanata</i>
Yarrow	<i>Achillea millefolium</i>
<b>Fish</b>	
American shad	<i>Alosa sapidissima</i>
Brook trout	<i>Salvelinus fontinalis</i>
Channel catfish	<i>Ictalurus punctatus</i>
Char	<i>Salvelinus</i> sp.
Chinook salmon	<i>Oncorhynchus tshawytscha</i>
Coho salmon	<i>Oncorhynchus kisutch</i>
Common carp	<i>Cyprinus carpio</i>
Crappie	<i>Pomoxis</i> spp.
Kokanee salmon	<i>Oncorhynchus nerka</i>
Mountain whitefish	<i>Prosopium williamsoni</i>
Northern pikeminnow (squawfish)	<i>Ptychocheilus oregonensis</i>
Rainbow trout	<i>Oncorhynchus mykiss</i>
Redside shiner	<i>Richardsonius balteatus</i>
Shorthead sculpin	<i>Cottus confusus</i>
Smallmouth bass	<i>Micropterus dolomieu</i>
Sockeye salmon	<i>Oncorhynchus nerka</i>
Speckled dace	<i>Rhinichthys osculus</i>
Steelhead trout	<i>Oncorhynchus mykiss</i>
Walleye	<i>Stizostedion vitreum</i>
White sturgeon	<i>Acipenser transmontanus</i>
Yellow perch	<i>Perca flavescens</i>
<b>Amphibians</b>	
Bullfrog	<i>Rana catesbeiana</i>
Great Basin spadefoot toad	<i>Spea intermontana</i>
Pacific tree frog	<i>Pseudacris regilla</i>
Tiger salamander	<i>Ambystoma tigrinum</i>
Western toad	<i>Bufo boreas</i>
Woodhouse's toad	<i>Bufo woodhousii</i>

**Table P-1. Scientific Names of Plant and Animal Species (continued)**

Common Name	Scientific Name
<b>Reptiles</b>	
Great Basin gopher snake	<i>Pituophis melanoleucus</i>
Short-horned lizard	<i>Phrynosoma douglasii</i>
Side-blotched lizard	<i>Uta stansburiana</i>
Western rattlesnake	<i>Crotalus viridis</i>
Western yellow-bellied racer	<i>Coluber constrictor</i>
<b>Birds</b>	
American kestrel	<i>Falco sparverius</i>
American robin	<i>Turdus migratorius</i>
Bald eagle	<i>Haliaeetus leucocephalus</i>
Barn swallow	<i>Hirundo rustica</i>
Black-billed magpie	<i>Pica hudsonia</i>
Black-crowned night heron	<i>Nycticorax nycticorax</i>
Brewer's sparrow	<i>Spizella breweri</i>
Burrowing owl	<i>Athene cunicularia</i>
California gull	<i>Larus californicus</i>
Canada goose	<i>Branta canadensis</i>
Cliff swallow	<i>Petrochelidon pyrrhonota</i>
Common raven	<i>Corvus corax</i>
Dark-eyed junco	<i>Junco hyemalis</i>
European starling	<i>Sturnus vulgaris</i>
Ferruginous hawk	<i>Buteo regalis</i>
Forster's tern	<i>Sterna forsteri</i>
Golden eagle	<i>Aquila chrysaetos</i>
Great blue heron	<i>Ardea herodias</i>
Horned lark	<i>Eremophila alpestris</i>
Killdeer	<i>Charadrius vociferous</i>
Lark sparrow	<i>Chondestes grammacus</i>
Loggerhead shrike	<i>Lanius ludovicianus</i>
Long-billed curlew	<i>Numenius americanus</i>
Mourning dove	<i>Zenaida macroura</i>
Northern harrier	<i>Circus cyaneus</i>
Peregrine falcon	<i>Falco peregrinus</i>
Pigeon	<i>Columba livia</i>
Prairie falcon	<i>Falco mexicanus</i>
Red-tailed hawk	<i>Buteo jamaicensis</i>
Ring-billed gull	<i>Larus delawarensis</i>
Rock wren	<i>Salpinctes obsoletus</i>
Rough-winged swallow	<i>Stelgidopteryx serripennis</i>
Sage sparrow	<i>Amphispiza belli</i>
Sage thrasher	<i>Oreoscoptes montanus</i>

**Table P-1. Scientific Names of Plant and Animal Species (continued)**

Common Name	Scientific Name
<b>Birds (continued)</b>	
Say's phoebe	<i>Sayornis saya</i>
Short-eared owl	<i>Asio flammeus</i>
Song sparrow	<i>Melospiza melodia</i>
Spotted sandpiper	<i>Actitis macularia</i>
Swainson's hawk	<i>Buteo swainsoni</i>
Vesper sparrow	<i>Pooecetes gramineus</i>
Western meadowlark	<i>Sturnella neglecta</i>
<b>Mammals</b>	
Badger	<i>Taxidea taxus</i>
Black-tailed jackrabbit	<i>Lepus californicus</i>
Bobcat	<i>Lynx rufus</i>
Coyote	<i>Canis latrans</i>
Gray wolf	<i>Canis lupus</i>
Great Basin pocket mouse	<i>Perognathus parvus</i>
Ground squirrel	<i>Citellus sp.</i>
Harvest mouse	<i>Reithrodontomys megalotis</i>
Least weasel	<i>Mustela nivalis</i>
Mink	<i>Mustela vison</i>
Mountain lion	<i>Puma concolor</i>
Mule deer	<i>Odocoileus hemionus</i>
Muskrat	<i>Ondatra zibethicus</i>
Northern pocket gopher	<i>Thomomys talpoides</i>
Porcupine	<i>Erethizon dorsatum</i>
Pronghorn	<i>Antilocapra americana</i>
Raccoon	<i>Procyon lotor</i>
Rocky Mountain elk	<i>Cervus elaphus</i>
Townsend's ground squirrel	<i>Spermophilus townsendii</i>

Key: sp.=species; spp.=species (plural).

## P.2 IMPACTS ON TERRESTRIAL RESOURCES RESULTING FROM CONTAMINANT RELEASES

Terrestrial ecological resources at Hanford potentially would be adversely impacted by surface disturbances and contaminant releases during site and Waste Treatment Plant (WTP) construction and operations under the various Tank Closure, FFTF Decommissioning, and Waste Management alternatives. The different alternatives would result in different surface disturbances in the vicinity of the 200 Areas, the 400 Area, and Borrow Area C (see Chapter 4). The different actions also would result in different amounts and timing of air emissions and their dispersion to terrestrial habitats at Hanford. Potential long-term impacts on terrestrial ecological resources at on- and offsite locations of chemical and radionuclide releases to air during site and WTP operations are evaluated in Sections P.2.2.1 and P.2.2.2. Potential long-term impacts of air releases during operations and groundwater releases in the future on Columbia River aquatic and riparian ecological resources are evaluated in Section P.3.

The potential for adverse effects on terrestrial ecological resources of radionuclide- and chemical-modeled air releases under the various Tank Closure, FFTF Decommissioning, and Waste Management alternatives was evaluated primarily using a quantitative ecological risk assessment approach (63 FR 26846; EPA 1992, 1997). Concentrations of radionuclides and chemicals resulting from deposition of airborne contaminants during construction and operations associated with the alternatives were predicted, as described in Appendix G. These predicted release concentrations were used to evaluate the impacts on terrestrial ecological resources at Hanford both during construction and operations and immediately following operations. The general approach to the assessment of the potential for adverse effects or impacts on ecological resources is discussed in Section P.2.1.

Terrestrial ecological resources would be potentially impacted by contaminant releases to air and soil “on site,” i.e., within the Hanford boundaries, and “off site,” i.e., outside the Hanford boundaries. Potential impacts on terrestrial ecological resources of exposure to contaminants in soil and air were evaluated using the maximum average annual air concentration and cumulative soil concentrations resulting from air deposition. The onsite maximum-exposure location would be in the vicinity of the tank farms and the 200 Areas because the WTP and ground-level facilities are located adjacent to the 200 Areas, the air dispersion model is a Gaussian plume, and air concentrations decrease in magnitude moving away from the source. For consistency with other *TC & WM EIS* assessments of long-term impacts (see Chapter 5), the line of analysis for the onsite maximum-exposure location is the Core Zone Boundary in the predominant downwind direction. The offsite maximum-exposure location would be at the Columbia River because the river forms the Hanford boundary in the predominant downwind direction.

Air emissions and their subsequent deposition on soils would be possible under all action alternatives, as well as under the Tank Closure No Action Alternative (Tank Closure Alternative 1). Radionuclides and chemicals emitted to the air during construction and operations would be potentially transported away from the source to on- and offsite locations (e.g., the Columbia River floodplain), where they could impact terrestrial resources, and the Columbia River, where they could impact aquatic and riparian resources. The evaluation of impacts at these locations was made at a single point in time, that is, what would be the completion of operations. The duration of operations would vary by alternative (see Chapter 2). Immediately following operations, cumulative soil concentrations are expected to be at their maximum level after accumulating during operations and before attenuating following completion of operations. Therefore, ignoring losses from soil and radioactive decay is a conservative approach. The evaluation of potential adverse impacts on aquatic and riparian ecological resources at the Columbia River is described in Section P.3. The evaluation of potential long-term impacts on terrestrial ecological resources of contaminants released to air under the various alternatives is discussed in the following subsections.

### **P.2.1        Methods**

The potential for adverse effects on ecological resources of potential radionuclide and chemical releases under the different alternatives was evaluated using quantitative modeling (ANL 1999; DOE 1995, 1998; DOE Standard 1153-2002; Eslinger et al. 2002). The general approach was to estimate the exposure of ecological receptors to radionuclides and chemicals that would result from operations and actions under each alternative and then compare the estimated doses to benchmark doses, i.e., doses associated with a known level of adverse effect. Dose estimates were made for selected receptor organisms judged to be representative of groups of species known to occur and be exposed at Hanford, including federally and state-listed protected species; to be sensitive to chemicals and radionuclides potentially released; and to be among the highest exposed in their groups (ANL 1999). The benchmark doses used in this approach are associated with no or minimal adverse effect, so they are expected to be protective of all ecological resources, including special status species that may occur at Hanford (see Chapter 3, Section 3.2.7.4). Special status species are species protected by Federal and state laws, e.g., the Endangered Species Act of

1973, as amended (16 U.S.C. 1531 et seq.). Exposure estimates and Hazard Quotients allow the impacts under the different alternatives to be compared, as required by the National Environmental Policy Act. Comparing alternatives is the primary purpose of the ecological risk analysis in this *TC & WM EIS*.

A secondary purpose of the ecological risk analysis in this *TC & WM EIS* is to identify alternatives that would be unlikely to result in unacceptable risk to ecological receptors. Assessing the risk to highly exposed receptors and using conservative exposure assumptions and benchmarks allow those alternatives that are unlikely to result in adverse impacts on ecological resources to be identified with a high degree of confidence. In other words, if a conservatively estimated dose does not exceed the benchmark dose, then it is highly likely there would be no adverse impact of the exposure. On the other hand, this approach cannot be used to unequivocally conclude that any alternative would result in an unacceptable probability of an adverse impact on ecological resources. A conservatively estimated dose exceeding a benchmark dose does not imply that the receptor would be adversely impacted by the exposure because the actual dose may be less than the benchmark dose. In such a case, a more precise evaluation would be required to resolve the uncertainty. This “screening” approach is consistent with U.S. Environmental Protection Agency (EPA) (EPA 1997, 1999) and U.S. Department of Energy (DOE) guidelines (ANL 1999; DOE Standard 1153-2002; Eslinger et al. 2002) and is appropriate for prospective risk assessments for actions that have not yet occurred (Suter 1993).

Exposure was calculated using models that are consistent with EPA and DOE guidelines and with the ECEM [Ecological Contaminant Exposure Model], which was described in the *User Instructions for the Systems Assessment Capability, Rev. 0, Computer Codes, Volume 2: Impact Modules* (Eslinger et al. 2002) and used in the *Screening Assessment and Requirements for a Comprehensive Assessment, Columbia River Comprehensive Impact Assessment (CRCIA)* (DOE 1998) and the *Tank Waste Remediation System, Hanford Site, Richland, Washington, Final Environmental Impact Statement* (DOE and Ecology 1996). The model exposure equations are consistent with those used in the DOE technical standard, *A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota* (DOE Standard 1153-2002). These are equilibrium steady state models, as opposed to dynamic time-varying models (Eslinger et al. 2002). The ECEM software was not used to make exposure calculations; however, the exposure calculations in this *TC & WM EIS* are functionally equivalent. Wherever possible, the representative receptors were selected from the ECEM model receptors, and the same receptor exposure parameters were used in this assessment. The selected receptors are presented in Table P-2.

The combined total dose from internal and external exposures to all radionuclides was calculated using equations based on those in *Methods for Estimating Doses to Organisms from Radioactive Materials Released into the Aquatic Environment* (Baker and Soldat 1992) and using the dose conversion factors (DCFs), activation energies, and other radiological parameters used in the ECEM. Chemical doses were calculated using published rates of ingestion of different media and estimated concentrations in the ingested media. Body burdens of chemicals and radionuclides were estimated using concentrations in ambient or ingested media and bioaccumulation factors (BAFs) for the receptor and the radionuclide or chemical in the media. As with the ECEM model (Eslinger et al. 2002), BAFs for animal receptors are constants at steady state, reflecting the net result of ingestion, inhalation, absorption, excretion, and elimination. For this assessment, inhalation of radionuclides and chemicals was estimated where possible, even though the dose from inhalation by biota would be small compared with ingestion and direct external radiation (DOE Standard 1153-2002). Dermal exposure was calculated only for external doses from radionuclides because dermal uptake of chemicals was judged to be small compared with direct exposure to chemicals in soil by incidental ingestion and indirect exposure of contaminated biota by ingestion. The exposure of animals to chemicals in soil by dermal contact would likely be small due to fur, feather, and epidermis barriers (EPA 2000).

Table P-2. Receptors and Exposure Pathways Evaluated for Long-Term Impacts of Air and Groundwater Releases

Receptor	Ingestion					Inhalation of Suspended Soil	Internal Exposure	Soil Exposure		Air Exposure	Near Water	Immersion		Sediment Surface Contact
	Plants	Soil/Sediment Biota <sup>a</sup>	Vertebrate Prey <sup>b</sup>	Solid Substrate <sup>c</sup>	Surface Water <sup>d</sup>			Above Ground	Below Ground			Water	Sediment	
<b>Terrestrial Environment</b>														
Plants	-	-	-	-	-	-	A	A	A <sup>e</sup>	-	-	-	-	-
Soil-dwelling invertebrates	-	-	-	-	-	-	A	A	A <sup>e</sup>	-	-	-	-	-
Side-blotched lizard	-	A	-	A	-	A	A	A	A	A	-	-	-	-
Mule deer	A	-	-	A	-	A	A	A	-	A	-	-	-	-
Mourning dove	A	-	-	A	-	A	A	A	-	A	-	-	-	-
Great Basin pocket mouse	A	A	-	A	-	A	A	A	A	A	-	-	-	-
Western meadowlark	A	A	-	A	-	A	A	A	A	A	-	-	-	-
Coyote	-	-	A	A	-	A	A	A	A	A	-	-	-	-
Burrowing owl	-	-	A	A	-	A	A	A	A	A	-	-	-	-
<b>Riparian Environment</b>														
Woodhouse's toad	-	A	-	A	-	A	A	A	A	A	-	-	-	-
Muskrat	-	-	-	-	GW	-	GW	GW	GW	-	-	-	-	-
<b>Aquatic Environment</b>														
Benthic invertebrates	-	-	-	-	-	-	A, GW	-	-	-	-	A, GW	A <sup>e</sup> , GW	-
Aquatic biota	-	-	-	-	-	-	A, GW	-	-	-	-	A <sup>e</sup> , GW	-	A, GW
Salmonids	-	-	-	-	-	-	A, GW	-	-	-	-	A <sup>e</sup> , GW	-	A, GW
Raccoon	-	A, GW	-	A, GW	A, GW	-	A, GW	A	A	-	A, GW	-	-	-
Spotted sandpiper	-	A, GW	-	A, GW	A, GW	-	A, GW	A	-	-	A, GW	-	-	-
Least weasel	-	-	A, GW	A, GW	A, GW	-	A, GW	A	A	-	A, GW	A, GW	-	-
Bald eagle	-	-	A, GW	A, GW	A, GW	-	A, GW	-	-	-	A, GW	-	-	-

<sup>a</sup> Soil-dwelling invertebrates for terrestrial and riparian; benthic invertebrates for aquatic.

<sup>b</sup> Small mammals for terrestrial; fish for aquatic.

<sup>c</sup> Surface soil for terrestrial; sediment for aquatic.

<sup>d</sup> For future impacts of groundwater release, the source of water ingested was assumed to be groundwater discharging at seeps along the Columbia River; otherwise, it was assumed to be nearshore surface water.

<sup>e</sup> For chemicals.

**Note:** Includes all direct and indirect exposure pathways.

**Key:** - = pathway not evaluated; A = pathway evaluated for air releases; GW = pathway evaluated for groundwater releases.

The exposure model equations are presented in the following sections for each of the impact assessments. The modeled pathways were assumed to be the largest exposure pathways for the receptors because of the habitat associated with each alternative and the source of contamination that was present. Partial doses were calculated where there was insufficient information to calculate the total dose. For example, an uptake or excretion parameter required to estimate the dose from inhalation might not have been available for a receptor, so inhalation could not be calculated for that receptor for any contaminant. The resulting underestimates of dose and risk were balanced by the overestimates from the conservative exposure assumptions. Calculated doses were adequate for comparing alternatives because they were consistent across the alternatives for a given receptor.

The benchmarks for combined internal and external exposure from all radionuclides are associated with no adverse impact (NCRP 1991; IAEA 1992) and were used in the DOE technical standard for evaluating radiation doses (DOE Standard 1153-2002). The chemical benchmarks for plants; soil-dwelling invertebrates; aquatic biota, including salmonids (e.g., salmon, trout, char); and sediment biota exposed to soil, water, and sediment, as appropriate, come from a variety of sources. The chemical benchmarks for wildlife are doses associated with no observed adverse effect levels measured in laboratory toxicity tests on test species (EPA 2009; Sample, Opresko, and Suter 1996). Data are available for mammals and birds for some of the chemical contaminants that could be released to air or groundwater and are evaluated in this *TC & WM EIS*; data for birds were used for amphibians and lizards without adjustment. Unlike radionuclides, impacts of exposure to chemicals were evaluated individually, and doses from different chemicals were not summed or otherwise mathematically combined.

The assumptions, receptors, exposure pathways and uptake mechanisms (routes), predicted soil concentrations, exposure model equations, and benchmarks used to model exposure for terrestrial ecological resources potentially impacted by contaminant releases are described in the relevant sections below. The quantitative evaluations of long-term adverse impacts on terrestrial resources of air releases, based on Hazard Quotients, Hazard Indices, and soil pH, are summarized and discussed in Section P.2.2. Impacts of sulfur and nitrogen oxide deposition on the soil's pH were evaluated based on buffering capacity and predicted concentrations.

### **P.2.1.1 Key Assumptions**

The following key assumptions were made in the evaluation of potential impacts on terrestrial resources of exposure to radionuclides and chemicals released to air during operations:

- Ecological receptors would not be exposed to onsite soil after operations once any proposed soil cover is in place.
- Major exposure pathways were evaluated.
- Toxicity benchmarks were protective.
- No loss, biological or chemical degradation, or radioactive decay of constituents of potential concern (COPCs) would occur in soil.

### **P.2.1.2 Receptors and Exposure Pathways and Routes**

The receptors that were selected to represent the terrestrial ecological resources are listed in Table P-2. They are a subset of those listed in Table P-1. Representative receptors were selected because they are expected to have higher exposures than those not selected from their group, due to their higher ingestion rates per unit body weight for prey, water, and soil. The selected representative receptors are expected to be as highly exposed and/or sensitive as any other species. The receptors include plants and soil-dwelling invertebrates, as well as the side-blotched lizard, Woodhouse's toad, mule deer, mourning dove,

Great Basin pocket mouse, western meadowlark, coyote, and burrowing owl. Plants and soil-dwelling invertebrates live in close contact with soil and are important food items for other receptors. The mourning dove, Great Basin pocket mouse, western meadowlark, and burrowing owl are not among the 52 ECEM receptors because the ECEM focuses on Columbia River riparian habitats more than the surrounding shrub-steppe habitat where these four receptors occur. The Great Basin pocket mouse was selected as a receptor for terrestrial habitats in the *Tank Waste Remediation System, Hanford Site, Richland, Washington, Final Environmental Impact Statement* (DOE and Ecology 1996) and is expected to be an important prey item for coyotes and burrowing owls. The mourning dove, western meadowlark, and burrowing owl are representative of birds exposed in terrestrial habitats at Hanford. Terrestrial receptors in common with the ECEM are the side-blotched lizard, mule deer, and coyote. Woodhouse's toad was evaluated instead of the side-blotched lizard for the offsite maximum-exposure location (the Columbia River) because side-blotched lizards are unlikely to occur in the Columbia River floodplain.

The exposure pathways evaluated in the ecological risk analysis for this *TC & WM EIS* are shown in Table P-2 for all ecological receptors. The exposure medium, exposure route, and receptor are indicated for each pathway evaluated in the analysis of impacts on terrestrial resources of releases to air.

### **P.2.1.3 Predicted Soil and Air Concentrations**

The cumulative surface-soil and maximum air concentrations under Tank Closure Alternatives 1 through 6C; FFTF Decommissioning Alternatives 1, 2, and 3 (Hanford and Idaho Options); and Waste Management Alternatives 1, 2, and 3 were calculated from the modeled air deposition rates resulting from site and WTP operations (see Appendix G). The onsite soil concentrations were calculated from the maximum-modeled air deposition rates. The modeled soil concentrations assumed persistence of existing soil contamination and accumulation of deposited contamination over the duration of the operations period. The surface-soil concentrations were calculated assuming that the amount of material deposited on the soil surface over the operations period would be mixed throughout the upper 1 centimeter (0.39 inches) of soil. The deposition flux per unit area (grams per square meter per year or curies per square meter per year) was multiplied by the duration of operations (years) and divided by the mass of soil per unit area (grams per square meter) to estimate the concentration (grams of contaminant per gram of soil or curies per gram), and these results were converted to milligrams per kilogram or picocuries per gram. The mass of soil per unit area was estimated as the depth of soil (0.01 meters [0.03 feet]) multiplied by the soil density ( $1.7 \times 10^6$  grams per cubic meter). The instantaneous air concentration (milligrams per cubic meter or picocuries per cubic meter) was estimated as the annual average deposition flux (milligrams per second or picocuries per second) divided by the unitized flux rate (cubic meters per second). The conservative estimates of surface-soil concentrations for radionuclides were used for both above- and belowground external exposures.

Air concentrations at the ground surface resulting from resuspension of soil were calculated for each location for which soil concentrations were predicted. Modeled air concentrations of radionuclides were used to calculate external exposure to terrestrial ecological resources.

Soil and air concentrations were used as the source term in the exposure model described below.

### **P.2.1.4 Exposure Model Calculations**

The exposure model calculated external and internal doses from radioactive COPCs for all receptors and ingestion and inhalation doses from chemical COPCs for all wildlife receptors. To calculate internal doses for radioactive COPCs in receptors exposed by direct contact with soil (plants and soil-dwelling invertebrates) and to calculate the ingested doses for wildlife receptors exposed by ingestion of these biota to chemical COPCs, the concentrations in these biota were required.

For plants, the concentration was calculated as follows:

$$C_p = P_v + P_r$$

where:

$$P_v = (D / \rho) \times Bv \times Fv \times VG \times 0.2$$

and

$$P_r = C_{\text{soil}} \times SP \times 0.2$$

where:

$C_p$	=	concentration in plants, milligrams per kilogram or picocuries per gram
$P_v$	=	concentration in plants from vapor, milligrams per kilogram or picocuries per gram
$P_r$	=	concentration in plants from root uptake, milligrams per kilogram or picocuries per gram
$D$	=	concentration in air, milligrams per cubic meter or picocuries per cubic meter
$\rho$	=	air density, 1.2 kilograms per cubic meter for chemical COPCs and 1,200 grams per cubic meter for radioactive COPCs
$Bv$	=	air-to-plant uptake factor, unitless
$Fv$	=	vapor fraction, 0 or 1
$VG$	=	empirical correction factor for air-to-plant transfer (1 for chemical COPCs and radioactive COPCs with a $\log K_{ow} < 4$ or no $\log K_{ow}$ [EPA 2005]), unitless
0.2	=	dry weight-to-wet weight conversion factor (moisture content of plants assumed to be 0.8), unitless
$C_{\text{soil}}$	=	concentration in soil, milligrams per kilogram or picocuries per gram dry soil
$SP$	=	soil-to-plant uptake factor, unitless

Soil-to-plant uptake factors were used for all radioactive COPCs except carbon-14 and hydrogen-3 (tritium). For carbon-14 and tritium, internal activities were based on equilibrium with stable isotopes in tissue and water, as discussed in Section P.2.1.4.3.

For soil-dwelling invertebrates, the concentration was calculated as follows:

$$C_a = C_{\text{soil}} \times BAF-S$$

where:

$C_a$	=	concentration in soil-dwelling invertebrates, milligrams per kilogram or picocuries per gram
$C_{\text{soil}}$	=	concentration in soil, milligrams per kilogram or picocuries per gram dry soil
$BAF-S$	=	soil-to-soil invertebrate bioaccumulation factor, unitless

Per the *Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA 1999),  $BAF-S$  values for organic chemical COPCs were derived from water-to-tissue bioconcentration factors (BCFs) for daphnids (EPA 1999) because there are no published values based on soil measurements. This approach assumed that soil-dwelling invertebrates are exposed to soil pore water in equilibrium with soil. The  $BAF-S$  values for the organic chemical COPCs were calculated as the *Daphnia* BCF for the chemical COPC divided by the product of the equilibrium partitioning coefficient ( $K_{oc}$ ) and the soil organic carbon content, which was assumed to be 0.01 (DOE 1998). The  $BAF-S$  value for inorganic chemical COPCs was the arithmetic mean of the recommended values for those inorganic substances with empirical data available: arsenic, cadmium, chromium, copper, lead, inorganic mercury, nickel, and zinc (EPA 1999).

#### **P.2.1.4.1 External Dose from Radionuclides**

External radiation doses from air, soil, water, and sediment were calculated by methods presented in *Methodology for Estimating Radiation Dose Rates to Freshwater Biota Exposed to Radionuclides in the Environment* (Blaylock, Frank, and O’Neal 1993) and *Methods and Tools for Estimation of the Exposure of Terrestrial Wildlife to Contaminants* (Sample et al. 1997), based on *Methods for Estimating Doses to Organisms from Radioactive Materials Released into the Aquatic Environment* (Baker and Soldat 1992). External irradiation by immersion in air containing radioactive COPCs and by standing, sitting, or lying on the soil surface (aboveground radiation) was modeled using external DCFs, which are presented in *External Exposure to Radionuclides in Air, Water, and Soil* (Eckerman and Ryman 1993), and the activity of the radioactive COPCs in the medium. Aboveground external radiation from soil was adjusted for the fraction of time the receptor was assumed to spend on the soil surface or for the fraction of the receptor’s body located above ground. Those fractions (based on professional judgment) are as follows: plants, 0.5; soil-dwelling invertebrates, 0.5; side-blotched lizard, 0.5; mule deer, 1; mourning dove, 1; Great Basin pocket mouse, 0.3; western meadowlark, 1; coyote, 0.7; and burrowing owl, 0.5. The DCFs used for the Woodhouse’s toad were extrapolated from values for similarly sized receptors presented in *Methods and Tools for Estimation of the Exposure of Terrestrial Wildlife to Contaminants* (Sample et al. 1997). The Woodhouse’s toad’s fraction of time above ground and fraction of time below ground were 0.5 and 0.5, respectively.

A roughness factor ( $F_{\text{ruf}}$ ) was used to correct for absorption of radiation by uneven soil contours, and an elevation correction factor (ECF) was used to adjust DCFs to account for most ecological receptors whose bodies are closer to the ground than the humans for which the DCFs were derived. The  $F_{\text{ruf}}$  for all receptors was set at 0.7, which was assumed to be a representative average correction for ground roughness (1.0 equates to a paved surface, whereas 0.5 equates to a deeply plowed field). The ECF was 2 for all receptors except the mule deer (ECF = 1), which are large enough to receive radiation at approximately the same height as humans (Sample et al. 1997).

Belowground external radiation from soil was modeled by using the decay energies and tissue absorption fractions. Equations to calculate belowground external exposure are presented in *Methods and Tools for Estimation of the Exposure of Terrestrial Wildlife to Contaminants* (Sample et al. 1997). Belowground exposure was adjusted for the fraction of time the receptor was assumed to be exposed under ground or the fraction of the body located above ground. Those fractions (based on professional judgment) are as follows: plants, 0.5; soil-dwelling invertebrates, 0.5; the side-blotched lizard, 0.5; the Woodhouse’s toad, 0.5; the mule deer, 0; the mourning dove, 0; the Great Basin pocket mouse, 0.7; the western meadowlark, 0; the coyote, 0.3; and the burrowing owl, 0.5. Belowground and aboveground external exposure equations for soil were combined to form the equation for the external exposure to soil ( $RD_{\text{Ext-soil}}$ ) given below.

Therefore, the external dose from radionuclides in soil and air ( $RD_{\text{Ext}}$ ) was calculated as follows:

$$RD_{\text{Ext}} = RD_{\text{Ext-soil}} + RD_{\text{Ext-air}}$$

where:

$RD_{\text{Ext-soil}}$  = external radiation dose from soil, rad per day  
 $RD_{\text{Ext-air}}$  = external radiation dose from air, rad per day

For all receptors, the external dose from soil was calculated as follows (Eckerman and Ryman 1993):

$$RD_{\text{Ext-soil}} = C_{\text{soil}} \times DF_{\text{soil}}$$

where:

- $RD_{\text{Ext-soil}}$  = external radiation dose from soil, rad per day
- $C_{\text{soil}}$  = activity of radionuclide in untilled soil, picocuries per gram
- $DF_{\text{soil}}$  = factor for converting activity in soil to external dose from untilled soil

The total external dose from all radioactive COPCs in soil was the sum of the external doses from each radioactive COPC.

The external dose factor for soil ( $DF_{\text{soil}}$ ) was calculated as follows (Sample et al. 1997):

$$DF_{\text{soil}} = F_{\text{above}} \times F_{\text{ruf}} \times DCF \times CFb \times ECF + 1.05 \times F_{\text{below}} \times E_{\gamma} n_{\gamma} \times \Phi_{\gamma} \times CFa$$

where:

- $F_{\text{above}}$  = fraction of time spent above ground, unitless
- $F_{\text{ruf}}$  = dose rate reduction factor accounting for ground roughness, unitless
- $DCF$  = dose conversion factor for external radiation from soil contaminated to a depth of 1 centimeter (0.39 inches) (Eckerman and Ryman 1993), sieverts per second per becquerel per cubic meter
- $CFb$  =  $5.12 \times 10^{11}$ , factor for converting sieverts per second per becquerel per cubic meter to rad per day per picocurie per gram
- $ECF$  = elevation correction factor to adjust dose coefficient for effective height of receptor above ground (Sample et al. 1997), unitless
- 1.05 = conversion factor to account for immersion in soil rather than water
- $F_{\text{below}}$  = fraction of time spent below ground, unitless
- $E_{\gamma} n_{\gamma}$  = photon energy emitted during transition from a higher to a lower energy state, 1 million electron volts (MeV)  $\times$  proportion of disintegrations producing gamma radiation
- $\Phi_{\gamma}$  = absorbed fraction of energy from gamma energy  $E_{\gamma}$
- $CFa$  = unit conversion factor,  $5.11 \times 10^{-5}$  rad per day per picocurie per gram per MeV per disintegration

Only gamma radiation was relevant to the external dose.

The external dose to all receptors from air was calculated as follows (Eckerman and Ryman 1993):

$$RD_{\text{Ext-air}} = D \times DF_{\text{air}}$$

where:

- $RD_{\text{Ext-air}}$  = external radiation dose from air, rad per day
- $D$  = activity of radionuclide in air, picocuries per cubic meter
- $DF_{\text{air}}$  = factor for converting activity in air to external dose from air

The external dose conversion factor for air ( $DF_{\text{air}}$ ) was calculated as follows:

$$DF_{\text{air}} = 3.2 \times 10^5 \times DCF$$

where:

- $3.2 \times 10^5$  = factor for converting sieverts per second per becquerel per cubic meter to rad per day per picocurie per cubic meter (Eckerman and Ryman 1993)  
 $DCF$  = dose conversion factor for external radiation from immersion in air (Eckerman and Ryman 1993), sieverts per second per becquerel per cubic meter

#### **P.2.1.4.2 Internal Dose from Radionuclides**

Internal exposure to radionuclides was calculated from the activity in the receptor's tissues. The internal activities of radionuclides were calculated using uptake factors and activities in soil and food. Internal radiation doses were calculated by multiplying the activity in tissues by the sum of alpha, beta, and gamma decay energies, where alpha and beta energies were assumed to be completely absorbed. Because gamma rays, like x-rays, may pass through the tissues without depositing their energy, gamma energies were adjusted to account for greater absorption by larger organisms (e.g., the mule deer) at a given energy level and for greater absorption by all receptors at lower energy levels.

The internal doses (rad per day) to plants, soil-dwelling invertebrates, and wildlife receptors were calculated as follows (Sample et al. 1997):

$$RD_{\text{Int}} = C_n \times DF_{\text{Int}}$$

where:

$$DF_{\text{Int}} = Cfa \times (QF \times E_{\alpha}n_{\alpha} \times \Phi_{\alpha} + E_{\beta}n_{\beta} \times \Phi_{\beta} + E_{\gamma}n_{\gamma} \times \Phi_{\gamma})$$

and where:

- $RD_{\text{Int}}$  = internal radiation dose, rad per day  
 $C_n$  = activity of radionuclide in receptor tissue, picocuries per gram  
 $DF_{\text{Int}}$  = factor for converting radioactive COPC activity in tissue to internal dose  
 $Cfa$  = unit conversion factor,  $5.11 \times 10^{-5}$  rad per day per picocurie per gram per MeV per disintegration  
 $QF$  = 5, quality factor for biological effect of alpha radiation (Kocher and Trabalka 2000), unitless  
 $E_{\alpha}n_{\alpha}$  = average energy emitted as alpha radiation, MeV per disintegration  $\times$  proportion of disintegrations producing an alpha particle  
 $\Phi_{\alpha}$  = absorbed fraction of energy from alpha energy  $E_{\alpha}$   
 $E_{\beta}n_{\beta}$  = average energy emitted as beta radiation, MeV per disintegration  $\times$  proportion of disintegrations producing a beta particle  
 $\Phi_{\beta}$  = absorbed fraction of energy from beta energy  $E_{\beta}$   
 $E_{\gamma}n_{\gamma}$  = photon energy emitted during transition from a higher to a lower energy state, MeV  $\times$  proportion of disintegrations producing gamma radiation  
 $\Phi_{\gamma}$  = absorbed fraction of energy from gamma energy  $E_{\gamma}$

In addition to estimating internal exposures, activities of radioactive COPCs and concentrations of chemical COPCs in some receptor tissues were also used to estimate the ingestion dose to predators eating those receptors.

### P.2.1.4.3 Tissue Concentrations and Activities

The activity of a radioactive COPC and the concentration of a chemical COPC in receptor tissue results from ingestion and inhalation of radioactive and chemical COPCs in soil and food. Accumulation from ingested matter was modeled according to EPA guidelines (EPA 1999). The *CRCIA* (DOE 1998) contains a model for receptor- and chemical-specific accumulation from inhalation of particulates in air as a result of absorption and excretion (see *CRCIA*, Appendix I-D). For radionuclides, inhalation was normalized to ingestion of soil (DOE Standard 1153-2002). Because of a lack of available receptor- and chemical-specific data, absorption was assumed to be a receptor-specific parameter equal for all chemical and radioactive COPCs, and excretion was assumed to be a chemical-specific parameter common to all receptors.

The activities of radioactive COPCs and concentrations of chemical COPCs in receptor tissue, except for carbon-14 and tritium, were calculated as follows:

$$C_n = C_{n\text{-ing}} + C_{n\text{-inh}}$$

- $C_n$  = activity of radioactive COPCs and concentration of chemical COPCs in receptor tissue, picocuries per gram or milligrams per kilogram
- $C_{n\text{-ing}}$  = activity of radioactive COPCs and concentration of chemical COPCs in receptor tissue resulting from ingestion, picocuries per gram or milligrams per kilogram
- $C_{n\text{-inh}}$  = activity of radioactive COPCs and concentration of chemical COPCs in receptor tissue resulting from inhalation, picocuries per gram or milligrams per kilogram

where for radioactive COPCs:

$$C_{n\text{-inh}} = D_s \times IR_{\text{air}} \times PT/IT \times Ba_{\text{receptor}} \times BW_{\text{receptor}} \times 0.001$$

where:

- $C_{n\text{-inh}}$  = activity of radioactive COPCs in receptor tissue resulting from inhalation, picocuries per gram
- $D_s$  = concentration in air from resuspended, untilled soil particles, milligrams per cubic meter air or picocuries per cubic meter air
- $IR_{\text{air}}$  = daily inhalation rate of soil, cubic meters air per kilogram body weight per day
- $PT/IT$  = unitless factor to adjust inhalation relative to ingestion for radionuclides (DOE Standard 1153-2002)
- $Ba_{\text{receptor}}$  = biotransfer rate of chemical in receptor, days per kilogram
- $BW_{\text{receptor}}$  = body weight of receptor, kilograms
- 0.001 = factor for converting kilograms to grams for radioactive COPCs, kilograms per gram

and  $D_s$  was calculated as follows:

$$D_s = C_{\text{soil}} \times L_d$$

where:

- $C_{\text{soil}}$  = concentration in untilled soil, milligrams per kilogram or picocuries per gram
- $L_d$  = dust loading constant, 150 micrograms per cubic meter, converted to kilograms per cubic meter or grams per cubic meter (Zach 1985).

and where for chemical COPCs:

$$C_{n\text{-inh}} = D_s \times IR_{\text{air}} \times \alpha / K$$

where:

- $C_{n\text{-inh}}$  = concentration of chemical COPCs in receptor tissue resulting from inhalation, milligrams per kilogram
- $IR_{\text{air}}$  = daily inhalation rate of air, cubic meters air per kilogram body weight per day
- $\alpha$  = fractional absorption coefficient, unitless
- $K$  = excretion constant, day<sup>-1</sup>

$IR_{\text{air}}$  was the receptor's inhalation rate of air (cubic meters air per kilogram body weight per day). It was receptor-specific and was derived from EPA guidelines (EPA 1993) using the fraction of dioxygen in dry atmosphere and the average annual Hanford temperature, as was done in the *CRCIA* (DOE 1998).  $IR_{\text{air}}$  values were obtained from regression equations based on body weight, except for the value for the Woodhouse's toad, which was based on the metabolic rate of an adult bullfrog (EPA 1993).

For both radioactive and chemical COPCs, the concentrations of contaminants from ingestion were calculated as follows:

$$C_{n\text{-ing}} = C_{\text{soil}} \times BAF\text{-}Ts + C_w \times BAF\text{-}Tw + C_a \times BAF\text{-}Ta + C_p \times BAF\text{-}Tp$$

where:

- $C_{n\text{-ing}}$  = concentration of contaminant in receptor tissue from ingestion, picocuries per gram or milligrams per kilogram
- $C_{\text{soil}}$  = concentration of contaminant in untilled soil, picocuries per gram or milligrams per kilogram
- $C_w$  = concentration of contaminant in surface water, picocuries per milliliter or milligrams per liter
- $C_a$  = concentration of contaminant in animals, picocuries per gram or milligrams per kilogram
- $C_p$  = concentration of contaminants in plants, picocuries per gram or milligrams per kilogram

where  $C_a$ , the concentration of chemicals or radionuclides in animal food, was calculated as  $C_n$  for the prey item as a receptor and  $BAF\text{-}Ts$ ,  $BAF\text{-}Tw$ ,  $BAF\text{-}Ta$ , and  $BAF\text{-}Tp$  were the receptor's uptake factors for the different ingested media: soil or sediment (kilogram per kilogram), water (liter per kilogram or milliliter per gram), animals (kilogram per kilogram), and plants (kilogram per kilogram), respectively, as follows:

$$\begin{aligned} BAF\text{-}Ts &= I_s \times Ba_{\text{receptor}} \\ BAF\text{-}Tw &= I_w \times Ba_{\text{receptor}} \\ BAF\text{-}Ta &= I_a \times Ba_{\text{receptor}} \\ BAF\text{-}Tp &= I_p \times Ba_{\text{receptor}} \end{aligned}$$

and

$$Ba_{\text{receptor}} = Ba_{\text{cow}} \times BW_{\text{cow}} / BW_{\text{receptor}}$$

where:

$Ba_{\text{receptor}}$	=	biotransfer rate of chemical in receptor, days per kilogram
$Ba_{\text{cow}}$	=	biotransfer rate of chemical in cow, days per kilogram
$BW_{\text{cow}}$	=	body weight of cow, kilograms = 200 kilograms (440 pounds)
$BW_{\text{receptor}}$	=	body weight of receptor, kilograms
$I_s$	=	daily ingestion rate of soil or sediment, kilograms dry matter per day
$I_w$	=	daily ingestion rate of water, liters per day
$I_a$	=	daily ingestion rate of animal matter, kilograms wet weight animal per day
$I_p$	=	daily ingestion rate of plant matter, kilograms wet weight plant per day

BAFs for wildlife receptors corrected the biotransfer factors for a 200-kilogram (440-pound) cow (Baes et al. 1984) for differences in body weight between cow and receptor. This approach was conservative and assumed that net uptake and assimilation efficiency would be more similar across organisms than the biotransfer factor, which is a function of body weight, uptake efficiency (absorption, elimination), and excretion.

$I_s$ ,  $I_w$ ,  $I_a$ , and  $I_p$  were the receptor's ingestion rates for soil or sediment, water, animal food, and plant food, respectively. The ingestion rates for solid matter were calculated as follows:

$$I_s = IR_f \times SF \times BW$$

$$I_p = IR_f \times PF \times BW$$

$$I_a = IR_f \times AF \times BW$$

where:

$IR_f$	=	daily specific ingestion rate of food, kilograms wet weight per kilograms body weight per day
$SF$	=	dry soil or sediment ingested as a fraction of daily food (wet weight) ingested, unitless
$BW$	=	body weight, kilograms
$PF$	=	fraction of diet that is plant, unitless
$AF$	=	fraction of diet that is animal, unitless

The ingestion rate for water ( $I_w$ ) was calculated as follows:

$$I_w = IR_w \times BW$$

where:

$IR_w$	=	daily specific ingestion rate of water, liters per kilogram body weight per day
$BW$	=	body weight, kilograms

These were the general equations; not all receptors ingested plant, animal, soil, sediment, and water. Only receptors exposed to soil were assumed to inhale untilled soil particles resuspended in air. Per the simplifying assumptions, exposure models for on- and offsite terrestrial receptors at Hanford did not include ingestion of water and sediment. Models for riparian receptors at the Columbia River (see Sections P.3.1.2 and P.3.2.1.2) included ingestion of water and sediment, but not soil. When a receptor did not ingest a medium, the concentration and ingestion rate for that medium were taken to be zero and the calculated BAF and fraction of total dose were zero; thus, that medium did not contribute to the receptor's tissue concentration.

Exposure calculations for most radioactive COPCs were based on the assumption that radionuclides would be present as particulates in soil or vapors in air. However, special consideration was given to carbon-14 and tritium, as these radioactive COPCs are processed by vegetation with natural carbon and hydrogen, respectively. Thus, the vegetation pathways for carbon-14 and tritium would depend on the exchange of carbon and hydrogen between plants and the environment. For this assessment, guidance from U.S. Nuclear Regulatory Commission Regulatory Guide 1.109 (NRC 1977) was used to account for the bioaccumulation of carbon-14 and tritium in plants. This was done through the use of correction factors, along with the assumption that all carbon-14 would be released in oxide form (carbon monoxide or carbon dioxide) and tritium would be released as water vapor. These correction factors were applied to the air concentration (e.g., picocuries per cubic meter) estimated at the point of exposure by the air model.

The concentration of carbon-14 in vegetation was calculated under the assumption that its ratio to the natural carbon in vegetation would equal the ratio of carbon-14 to the natural carbon in the atmosphere surrounding the vegetation, as follows (NRC 1977):

$$C_{p(C-14)} = D_{C-14} \times p \times 0.11 / 0.16$$

where:

- $C_{p(C-14)}$  = concentration of carbon-14 in vegetation, picocuries radioactive COPC per gram wet plant tissue
- $D_{C-14}$  = concentration of carbon-14 in the surrounding air, picocuries per cubic meter air
- $p$  = ratio of the total annual release time to the total annual time during which photosynthesis occurs; a conservative ratio of 1.0 was used
- 0.11 = fraction of the total plant mass that is natural carbon, grams carbon per gram wet plant tissue
- 0.16 = concentration of natural carbon in the atmosphere, grams carbon per cubic meter air

The concentration of tritium in vegetation was calculated based on the equilibrium between moisture in the air and water in plants, as follows (NRC 1977):

$$C_{p(H-3)} = D_{H-3} \times 0.80 \times (0.5 / \text{humidity})$$

where:

- $C_{p(H-3)}$  = concentration of tritium in vegetation, picocuries radioactive COPC per gram wet plant tissue
- $D_{H-3}$  = concentration of tritium in the surrounding air, picocuries per cubic meter air
- 0.80 = site-specific assumed fraction of the total plant mass that is water, grams plant water per gram wet plant tissue
- 0.5 = ratio of tritium concentration in plant water to tritium concentration in atmospheric water, curies per gram plant water per curies per gram water in air
- humidity = humidity of the atmosphere, grams water per cubic meter air

A site-specific value of 68 percent or 0.68 grams per cubic meter (USFS, NPS, and USFWS 2000) was used for humidity.

The concentration of carbon-14 and tritium in vegetation was used as the total plant concentration for these radioactive COPCs throughout the risk assessment instead of estimating concentrations for specific parts of the plants (i.e., above and below ground). The concentrations of carbon-14 and tritium in the tissues of all terrestrial animal receptors were assumed to be equal to the concentrations in plants.

#### P.2.1.4.4 Exposure Doses from Chemicals

Exposure was estimated only for wildlife exposed to chemical COPCs via ingestion and inhalation. The average daily dose (ADD) for chemical COPCs was compared with benchmark doses to characterize risk. For plants and soil-dwelling invertebrates exposed to chemicals by multiple pathways (direct contact, ingestion) resulting from living in soil, exposure was not calculated. The assessment of impacts on plants and soil-dwelling invertebrates was made by comparing estimated soil concentrations to soil benchmark concentrations for these receptors (see Section P.2.1.5).

The doses to terrestrial wildlife receptors from chemical COPCs in soil were calculated as the sum of doses from inhaling air containing suspended soil and ingesting soil, food (plant and animal fractions), and water as follows:

$$ADD_{total} = ADD_{plant} + ADD_{animal} + ADD_{soil} + ADD_{water} + ADD_{air}$$

where:

- $ADD_{total}$  = total ingestion-equivalent dose of chemical from plant food, animal food, soil, and air, milligrams per kilogram body weight per day
- $ADD_{plant}$  = dose of chemical from ingestion of plants, milligrams per kilogram body weight per day
- $ADD_{animal}$  = dose of chemical from ingestion of animals, milligrams per kilogram body weight per day
- $ADD_{soil}$  = dose of chemical from ingestion of soil, milligrams per kilogram body weight per day
- $ADD_{water}$  = dose of chemical from ingestion of water, milligrams per kilogram body weight per day
- $ADD_{air}$  = ingestion-equivalent dose of chemical from inhalation of soil in air, milligrams per kilogram body weight per day

The dose of chemical from ingestion of plants ( $ADD_{plant}$ ) was calculated as follows:

$$ADD_{plant} = C_p \times IR_p = C_p \times IR_f \times PF$$

where:

- $C_p$  = concentration in plants, milligrams per kilogram wet weight
- $IR_p$  = daily ingestion rate of plant matter, kilograms fresh plant per kilograms body weight per day
- $IR_f$  = daily food ingestion rate, kilograms wet weight per kilograms body weight per day
- $PF$  = plant fraction of diet ( $ADD_{animal}$ )

The dose of chemical from ingestion of animals ( $ADD_{animal}$ ) was calculated as follows:

$$ADD_{animal} = C_a \times IR_a = C_a \times IR_f \times AF$$

where:

- $C_a$  = concentration in animal prey, milligrams per kilogram wet weight
- $IR_a$  = daily ingestion rate of animal matter, kilograms wet weight animal per kilogram body weight per day
- $IR_f$  = daily food ingestion rate, kilograms wet weight per kilogram body weight per day
- $AF$  = animal fraction of diet

Soil-dwelling invertebrates were the animal prey of the side-blotched lizard, Woodhouse's toad, Great Basin pocket mouse, and western meadowlark. The Great Basin pocket mouse was the animal prey of the coyote and the burrowing owl. Note that, for predators of the Great Basin pocket mouse,  $C_a$  was calculated as  $C_n$ , with the Great Basin pocket mouse treated as a receptor.

The dose of chemical from ingestion of soil ( $ADD_{soil}$ ) was calculated as follows:

$$ADD_{soil} = C_{soil} \times IR_s = C_{soil} \times IR_f \times SF$$

where:

- $C_{soil}$  = concentration in soil, milligrams per kilogram dry soil
- $IR_s$  = ingestion rate of soil by the receptor, kilograms dry soil per kilogram body weight per day
- $IR_f$  = daily food ingestion rate, kilograms wet weight per kilogram body weight per day
- $SF$  = dry soil ingested as a fraction of daily food (wet weight) ingested, unitless

The dose of chemical from ingestion of water ( $ADD_{water}$ ) was calculated as follows:

$$ADD_{water} = C_w \times IR_w$$

where:

- $C_w$  = concentration in water, milligrams per liter water
- $IR_w$  = daily specific ingestion rate of water, liters per kilogram body weight per day

The dose of chemical from inhalation of soil in air ( $ADD_{air}$ ) was calculated as follows:

$$ADD_{air} = D_s \times IR_{air} \times \alpha / K / (Ba_{receptor} \times BW_{receptor})$$

where:

- $D_s$  = concentration in air from resuspended untilled soil particles, milligrams per cubic meter of air
- $IR_{air}$  = daily inhalation rate of air, cubic meters per kilogram body weight per day
- $\alpha$  = fractional absorption coefficient, unitless
- $K$  = excretion constant, day<sup>-1</sup>
- $Ba_{receptor}$  = biotransfer rate of chemical in receptor, days per kilogram
- $BW_{receptor}$  = receptor body weight, kilograms

The factor,  $\alpha / K / (Ba_{receptor} \times BW_{receptor})$ , relates the efficiency of uptake into blood from the lung to the efficiency of uptake into blood from the gastrointestinal tract and was used to convert inhaled dose to ingested dose for the purposes of estimating the risk from exposure of inhaled substance in terms of ingestion-based toxicity reference values (TRVs). This factor was derived by taking the ratio of the equations for bioaccumulation in tissue of a substance inhaled (DOE 1998:I-D.10) and that of the substance ingested (EPA 1999:Equation 5-3), written in terms of dose. This approach assumes that, once a molecule of the substance is in the bloodstream, its fate is independent of the pathway by which it came to be there. In other words, a unit tissue concentration could result from either inhalation or ingestion of soil:

$$C_{n-ing} = C_{n-inh}$$

and

$$C_{\text{soil}} \times BAF-Ts = C_{\text{soil}} \times L_d \times IR_{\text{air}} \times \alpha / K$$

$$C_{\text{soil}} \times Ba_{\text{receptor}} \times I_s = D_s \times IR_{\text{air}} \times \alpha / K$$

$$C_{\text{soil}} \times IR_s \times Ba_{\text{receptor}} \times BW_{\text{receptor}} = D_s \times IR_{\text{air}} \times \alpha / K$$

$$\text{Dose}_{\text{ingested}} \times (Ba_{\text{receptor}} \times BW_{\text{receptor}}) = \text{Dose}_{\text{inhaled}} \times \alpha / K$$

$$\text{Dose}_{\text{ingested}} = \text{Dose}_{\text{inhaled}} \times \alpha / K / (Ba_{\text{receptor}} \times BW_{\text{receptor}})$$

where:

$$I_s = IR_s \times BW_{\text{receptor}}$$

and where:

$C_{\text{soil}}$	= concentration of contaminant in untilled soil, picocuries per gram or milligrams per kilogram
$BAF-Ts$	= $I_s \times Ba_{\text{receptor}}$
$L_d$	= dust loading constant, 150 micrograms per cubic meter, converted to kilograms per cubic meter or grams per cubic meter (Zach 1985)
$IR_{\text{air}}$	= daily inhalation rate of air, cubic meters air per kilogram body weight per day
$\alpha$	= fractional absorption coefficient, unitless
$K$	= excretion constant, day <sup>-1</sup>
$Ba_{\text{receptor}}$	= biotransfer rate of chemical in receptor, days per kilogram
$I_s$	= daily ingestion rate of soil or sediment, kilograms dry matter per day
$D_s$	= concentration in air from resuspended untilled soil particles, milligrams per cubic meter air
$IR_s$	= ingestion rate of soil by the receptor, kilograms dry soil per kilogram body weight per day
$BW_{\text{receptor}}$	= body weight of receptor, kilograms
$\text{Dose}_{\text{ingested}}$	= dose of chemical from ingestion resulting in unit of chemical in tissue, milligrams per kilogram body weight per day
$\text{Dose}_{\text{inhaled}}$	= dose of chemical from inhalation resulting in unit of chemical in tissue, milligrams per kilogram body weight per day

Area use factors and temporal use factors were assumed to equal 1 for conservatism; thus, these factors did not appear in the exposure equations.

### P.2.1.5 Toxicological Benchmarks

The benchmark for combined internal and external exposure from all radionuclides was 0.1 rad per day for the side-blotched lizard, Woodhouse's toad, mule deer, mourning dove, Great Basin pocket mouse, meadowlark, coyote, and burrowing owl, and 1 rad per day for plants and soil-dwelling invertebrates (IAEA 1992). Chemical benchmarks (TRVs) for plants and soil-dwelling invertebrates exposed to soil were soil concentrations (milligrams per kilogram), and TRVs for terrestrial receptors potentially impacted by chemicals in surface soil were doses (milligrams per kilogram body weight per day). All TRVs used were chemical-specific literature values from a variety of published sources (e.g., Efrogmson, Will, and Suter 1997; Efrogmson et al. 1997; EPA 2009; Sample, Opresko, and Suter 1996).

### P.2.1.6 Risk Indices

As discussed earlier in the introduction to Section P.2.1, the long-term impacts on ecological resources of potential radionuclide and chemical releases were evaluated by comparing estimates of exposure for a given ecological receptor exposed to a given chemical or radioactive COPC under each alternative to the

threshold exposures associated with a known level of the adverse effect of the COPC on that type of receptor. The estimate of chemical exposure for plants and soil-dwelling invertebrates was the predicted soil concentration under each alternative (see Appendix G). The methods for estimating exposure doses for terrestrial receptors from predicted air and soil concentrations were defined in Section P.2.1.4. The exposure concentrations or doses associated with a known level of adverse effect were the TRVs (see Section P.2.1.5). These two values were compared by calculating a risk index, the dimensionless ratio of the exposure estimate (concentration or dose) to the corresponding TRV (concentration or dose). These calculated risk indices, i.e., the Hazard Quotients for individual chemical COPCs and the Hazard Indices for all radioactive COPCs combined, were used to compare the *TC & WM EIS* alternatives (see Chapter 5) and to identify exposures posing little or no risk (Hazard Quotient or Hazard Index less than or equal to unity [1]).

The risk indices were calculated as follows:

For plants and soil-dwelling invertebrates exposed to chemical COPCs in soil:

$$HQ = C_{\text{soil}} / TRV$$

where:

$HQ$  = Hazard Quotient  
 $C_{\text{soil}}$  = concentration in untilled soil, milligrams per kilogram or picocuries per gram  
 $TRV$  = toxicity reference value, milligrams per kilogram

For wildlife receptors exposed to chemical COPCs in soil and air:

$$HQ = ADD_{\text{total}} / TRV$$

where:

$HQ$  = Hazard Quotient  
 $ADD_{\text{total}}$  = total ingestion-equivalent dose of chemical from plant food, animal food, soil, and air, milligrams per kilogram body weight per day  
 $TRV$  = toxicity reference value, milligrams per kilogram body weight per day

For all receptors, the Hazard Index is the sum of external and internal doses from all radioactive COPCs divided by the TRV, as follows:

$$HI = (RD_{\text{Ext}} + RD_{\text{Int}}) / TRV$$

where:

$HI$  = Hazard Index  
 $RD_{\text{Ext}}$  = external radiation dose from exposure to all radioactive COPCs in air, soil, sediment, and/or water, rad per day  
 $RD_{\text{Int}}$  = internal radiation dose from all radioactive COPCs, rad per day  
 $TRV$  = toxicity reference value, rad per day

Except where an exposure parameter or TRV was not available for a given receptor or COPC, the dose ( $ADD_{total}$ ) and Hazard Quotient for all chemical COPCs, as well as the dose ( $RD_{Ext} + RD_{Int}$ ) summed over all radioactive COPCs and the Hazard Index, were calculated for all terrestrial receptors potentially exposed at the two locations under all *TC & WM EIS* alternatives using predicted air and soil concentrations resulting from air releases during operations. Tables with predicted air and soil concentrations, input parameters, and calculations of dose and risk indices are provided in *Calculating Risk Indices for Long-Term Impacts to Ecological Receptors – Releases to Air* (SAIC 2011a). Results are summarized in Section P.2.2 using maximum Hazard Quotients and Hazard Indices.

## **P.2.2 Results and Discussion**

Radiological and chemical hazards estimated for terrestrial ecological receptors due to exposure to contaminant release to the air and subsequent deposition are discussed below. Hazards due to releases into the air and subsequent deposition in the Columbia River and releases into the groundwater for aquatic receptors and terrestrial wildlife feeding in the Columbia River are discussed in Section P.3.

### **P.2.2.1 Onsite Terrestrial Resources**

The results of the assessment for radioactive and chemical contaminant releases to air and subsequent deposition estimated for terrestrial receptors at the onsite maximum-exposure location under the various Tank Closure, FFTF Decommissioning, and Waste Management alternatives, as well as the alternative combinations, are summarized in Tables P-3, P-4, and P-5.

The maximum combined radiological Hazard Index from emissions under all alternatives was calculated to be 0.026 for the Great Basin pocket mouse under Tank Closure Alternative 6A, Option Case. Table P-3 presents the maximum Hazard Indices associated with air emissions of radioactive COPCs that are calculated to reach the onsite receptors under each of the alternatives. There would be no releases of radioactive COPCs under FFTF Decommissioning Alternative 1 and Waste Management Alternative 1. Exposures to radioactive COPCs from air emissions under all alternatives would be below the 1-rad-per-day benchmark for plants and soil-dwelling invertebrates, as well as the 0.1-rad-per-day benchmark for terrestrial wildlife receptors (i.e., side-blotched lizard, mule deer, mourning dove, Great Basin pocket mouse, western meadowlark, coyote, and burrowing owl). Estimated hazards for the representative species indicated that no adverse effects are expected for onsite terrestrial receptors from exposure to radioactive COPCs from air emissions. Because the direct impacts of air exposure are expected to be small, any associated, potential indirect impacts on the ecosystem are expected to be correspondingly minor.

**Table P-3. Long-Term Impacts of Radioactive COPC Air Deposition on Terrestrial Resources at the Onsite Maximum-Exposure Location: Maximum Hazard Indices by Receptor and Alternative**

Alternative	Maximum Hazard Index by Receptor								
	Plants	Soil Invertebrates	Side-Blotched Lizard	Mule Deer	Mourning Dove	Great Basin Pocket Mouse	Western Meadowlark	Coyote	Burrowing Owl
<b>Tank Closure</b>									
1	7.67×10 <sup>-4</sup>	8.51×10 <sup>-3</sup>	7.35×10 <sup>-3</sup>	6.48×10 <sup>-3</sup>	9.81×10 <sup>-3</sup>	7.33×10 <sup>-3</sup>	9.58×10 <sup>-3</sup>	9.24×10 <sup>-3</sup>	8.15×10 <sup>-3</sup>
2A	3.43×10 <sup>-3</sup>	1.17×10 <sup>-2</sup>	1.09×10 <sup>-2</sup>	7.35×10 <sup>-3</sup>	1.54×10 <sup>-2</sup>	1.67×10 <sup>-2</sup>	1.24×10 <sup>-2</sup>	1.12×10 <sup>-2</sup>	1.29×10 <sup>-2</sup>
2B	2.77×10 <sup>-3</sup>	3.18×10 <sup>-3</sup>	3.52×10 <sup>-3</sup>	9.47×10 <sup>-4</sup>	5.53×10 <sup>-3</sup>	9.10×10 <sup>-3</sup>	2.85×10 <sup>-3</sup>	2.02×10 <sup>-3</sup>	4.64×10 <sup>-3</sup>
3A	3.09×10 <sup>-3</sup>	3.60×10 <sup>-3</sup>	7.82×10 <sup>-3</sup>	5.10×10 <sup>-3</sup>	9.93×10 <sup>-3</sup>	1.37×10 <sup>-2</sup>	7.12×10 <sup>-3</sup>	6.24×10 <sup>-3</sup>	9.00×10 <sup>-3</sup>
3B	2.62×10 <sup>-3</sup>	3.00×10 <sup>-3</sup>	3.30×10 <sup>-3</sup>	8.23×10 <sup>-4</sup>	5.21×10 <sup>-3</sup>	8.64×10 <sup>-3</sup>	2.65×10 <sup>-3</sup>	1.85×10 <sup>-3</sup>	4.37×10 <sup>-3</sup>
3C	3.15×10 <sup>-3</sup>	3.85×10 <sup>-3</sup>	8.33×10 <sup>-3</sup>	5.22×10 <sup>-3</sup>	1.04×10 <sup>-2</sup>	1.46×10 <sup>-2</sup>	7.39×10 <sup>-3</sup>	6.72×10 <sup>-3</sup>	9.55×10 <sup>-3</sup>
4	2.91×10 <sup>-3</sup>	3.35×10 <sup>-3</sup>	4.22×10 <sup>-3</sup>	1.49×10 <sup>-3</sup>	6.33×10 <sup>-3</sup>	1.01×10 <sup>-2</sup>	3.51×10 <sup>-3</sup>	2.63×10 <sup>-3</sup>	5.40×10 <sup>-3</sup>
5	2.61×10 <sup>-3</sup>	3.07×10 <sup>-3</sup>	4.22×10 <sup>-3</sup>	1.64×10 <sup>-3</sup>	6.18×10 <sup>-3</sup>	9.78×10 <sup>-3</sup>	3.56×10 <sup>-3</sup>	2.72×10 <sup>-3</sup>	5.34×10 <sup>-3</sup>
6A, Base Case	4.80×10 <sup>-3</sup>	6.64×10 <sup>-3</sup>	8.76×10 <sup>-3</sup>	1.98×10 <sup>-3</sup>	1.33×10 <sup>-2</sup>	2.29×10 <sup>-2</sup>	7.01×10 <sup>-3</sup>	4.89×10 <sup>-3</sup>	1.17×10 <sup>-2</sup>
6A, Option Case	5.50×10 <sup>-3</sup>	7.92×10 <sup>-3</sup>	9.86×10 <sup>-3</sup>	2.29×10 <sup>-3</sup>	1.50×10 <sup>-2</sup>	<b>2.57×10<sup>-2</sup></b>	7.91×10 <sup>-3</sup>	5.54×10 <sup>-3</sup>	1.31×10 <sup>-2</sup>
6B, Base Case	4.93×10 <sup>-3</sup>	6.81×10 <sup>-3</sup>	9.06×10 <sup>-3</sup>	2.14×10 <sup>-3</sup>	1.38×10 <sup>-2</sup>	2.35×10 <sup>-2</sup>	7.28×10 <sup>-3</sup>	5.11×10 <sup>-3</sup>	1.20×10 <sup>-2</sup>
6B, Option Case	5.27×10 <sup>-3</sup>	7.52×10 <sup>-3</sup>	9.34×10 <sup>-3</sup>	2.31×10 <sup>-3</sup>	1.42×10 <sup>-2</sup>	2.41×10 <sup>-2</sup>	7.53×10 <sup>-3</sup>	5.31×10 <sup>-3</sup>	1.24×10 <sup>-2</sup>
6C	2.5×10 <sup>-3</sup>	3.13×10 <sup>-3</sup>	3.52×10 <sup>-3</sup>	9.39×10 <sup>-4</sup>	5.50×10 <sup>-3</sup>	9.08×10 <sup>-3</sup>	2.85×10 <sup>-3</sup>	2.02×10 <sup>-3</sup>	4.64×10 <sup>-3</sup>
<b>FFTF Decommissioning</b>									
1	0	0	0	0	0	0	0	0	0
2, Hanford Option	1.10×10 <sup>-5</sup>	1.38×10 <sup>-5</sup>	1.29×10 <sup>-4</sup>	1.09×10 <sup>-4</sup>	1.86×10 <sup>-4</sup>	1.36×10 <sup>-4</sup>	1.76×10 <sup>-4</sup>	2.00×10 <sup>-4</sup>	1.86×10 <sup>-4</sup>
2, Idaho Option	4.84×10 <sup>-12</sup>	6.64×10 <sup>-11</sup>	1.41×10 <sup>-10</sup>	2.65×10 <sup>-11</sup>	1.96×10 <sup>-10</sup>	3.63×10 <sup>-10</sup>	1.12×10 <sup>-10</sup>	7.86×10 <sup>-11</sup>	1.89×10 <sup>-10</sup>
3, Hanford Option	1.10×10 <sup>-5</sup>	1.38×10 <sup>-5</sup>	1.29×10 <sup>-4</sup>	1.09×10 <sup>-4</sup>	1.86×10 <sup>-4</sup>	1.36×10 <sup>-4</sup>	1.76×10 <sup>-4</sup>	2.00×10 <sup>-4</sup>	1.86×10 <sup>-4</sup>
3, Idaho Option	0	0	0	0	0	0	0	0	0
<b>Waste Management</b>									
1	0	0	0	0	0	0	0	0	0
2, DG1	9.49×10 <sup>-11</sup>	9.70×10 <sup>-10</sup>	2.23×10 <sup>-12</sup>	6.59×10 <sup>-12</sup>	2.72×10 <sup>-11</sup>	1.40×10 <sup>-11</sup>	2.70×10 <sup>-12</sup>	2.21×10 <sup>-12</sup>	2.01×10 <sup>-12</sup>
2, DG2	9.49×10 <sup>-11</sup>	9.70×10 <sup>-10</sup>	2.23×10 <sup>-12</sup>	6.59×10 <sup>-12</sup>	2.72×10 <sup>-11</sup>	1.40×10 <sup>-11</sup>	2.70×10 <sup>-12</sup>	2.21×10 <sup>-12</sup>	2.01×10 <sup>-12</sup>
2, DG3	9.49×10 <sup>-11</sup>	9.70×10 <sup>-10</sup>	2.23×10 <sup>-12</sup>	6.59×10 <sup>-12</sup>	2.72×10 <sup>-11</sup>	1.40×10 <sup>-11</sup>	2.70×10 <sup>-12</sup>	2.21×10 <sup>-12</sup>	2.01×10 <sup>-12</sup>
3, DG1	9.49×10 <sup>-11</sup>	9.70×10 <sup>-10</sup>	2.23×10 <sup>-12</sup>	6.59×10 <sup>-12</sup>	2.72×10 <sup>-11</sup>	1.40×10 <sup>-11</sup>	2.70×10 <sup>-12</sup>	2.21×10 <sup>-12</sup>	2.01×10 <sup>-12</sup>
3, DG2	9.49×10 <sup>-11</sup>	9.70×10 <sup>-10</sup>	2.23×10 <sup>-12</sup>	6.59×10 <sup>-12</sup>	2.72×10 <sup>-11</sup>	1.40×10 <sup>-11</sup>	2.70×10 <sup>-12</sup>	2.21×10 <sup>-12</sup>	2.01×10 <sup>-12</sup>
3, DG3	9.49×10 <sup>-11</sup>	9.70×10 <sup>-10</sup>	2.23×10 <sup>-12</sup>	6.59×10 <sup>-12</sup>	2.72×10 <sup>-11</sup>	1.40×10 <sup>-11</sup>	2.70×10 <sup>-12</sup>	2.21×10 <sup>-12</sup>	2.01×10 <sup>-12</sup>
<b>Alternative Combination</b>									
1	7.67×10 <sup>-4</sup>	8.51×10 <sup>-3</sup>	7.35×10 <sup>-3</sup>	6.48×10 <sup>-3</sup>	9.81×10 <sup>-3</sup>	7.33×10 <sup>-3</sup>	9.58×10 <sup>-3</sup>	9.24×10 <sup>-3</sup>	8.15×10 <sup>-3</sup>
2	2.78×10 <sup>-3</sup>	3.19×10 <sup>-3</sup>	3.65×10 <sup>-3</sup>	1.06×10 <sup>-3</sup>	5.71×10 <sup>-3</sup>	9.23×10 <sup>-3</sup>	3.03×10 <sup>-3</sup>	2.22×10 <sup>-3</sup>	4.82×10 <sup>-3</sup>
3	4.94×10 <sup>-3</sup>	6.83×10 <sup>-3</sup>	9.19×10 <sup>-3</sup>	2.25×10 <sup>-3</sup>	1.39×10 <sup>-2</sup>	2.37×10 <sup>-2</sup>	7.46×10 <sup>-3</sup>	5.31×10 <sup>-3</sup>	1.22×10 <sup>-2</sup>

**Note:** The maximum Hazard Index is indicated by **bold** text. Hazard Index is unitless.

**Key:** COPC=constituent of potential concern; DG=Disposal Group; FFTF=Fast Flux Test Facility.

**Table P-4. Long-Term Impacts of Chemical COPC Air Deposition on Terrestrial Resources at the Onsite Maximum-Exposure Location: Maximum Risk Index by Alternative**

Alternative	Maximum Hazard Quotient	Chemical COPC	Receptor
<b>Tank Closure</b>			
1	1.16	Xylene	Great Basin pocket mouse
2A	$1.52 \times 10^2$	Mercury	Side-blotched lizard
2B	$1.66 \times 10^2$	Mercury	Side-blotched lizard
3A	$3.92 \times 10^2$	Mercury	Side-blotched lizard
3B	$1.23 \times 10^2$	Xylene	Great Basin pocket mouse
3C	$3.92 \times 10^2$	Mercury	Side-blotched lizard
4	$1.57 \times 10^2$	Mercury	Side-blotched lizard
5	$1.49 \times 10^2$	Xylene	Great Basin pocket mouse
6A, Base Case	$2.70 \times 10^2$	Xylene	Great Basin pocket mouse
6A, Option Case	$2.74 \times 10^2$	Xylene	Great Basin pocket mouse
6B, Base Case	$1.73 \times 10^2$	Mercury	Side-blotched lizard
6B, Option Case	$1.71 \times 10^2$	Mercury	Side-blotched lizard
6C	$1.71 \times 10^2$	Mercury	Side-blotched lizard
<b>FFTF Decommissioning</b>			
1	$2.11 \times 10^3$	Xylene	Great Basin pocket mouse
2, Hanford Option	7.63	Xylene	Great Basin pocket mouse
2, Idaho Option	3.71	Xylene	Great Basin pocket mouse
3, Hanford Option	7.68	Xylene	Great Basin pocket mouse
3, Idaho Option	3.76	Xylene	Great Basin pocket mouse
<b>Waste Management</b>			
1	1.65	Xylene	Great Basin pocket mouse
2, DG1	$8.70 \times 10^1$	Xylene	Great Basin pocket mouse
2, DG2	$3.44 \times 10^2$	Xylene	Great Basin pocket mouse
2, DG3	$4.67 \times 10^2$	Xylene	Great Basin pocket mouse
3, DG1	$8.36 \times 10^1$	Xylene	Great Basin pocket mouse
3, DG2	$3.41 \times 10^2$	Xylene	Great Basin pocket mouse
3, DG3	$4.63 \times 10^2$	Xylene	Great Basin pocket mouse
<b>Alternative Combination</b>			
1	<b><math>2.12 \times 10^3</math></b>	Xylene	Great Basin pocket mouse
2	$1.92 \times 10^2$	Xylene	Great Basin pocket mouse
3	$5.03 \times 10^2$	Xylene	Great Basin pocket mouse

**Note:** The maximum Hazard Quotient of all receptors is indicated by **bold** text. Risk indices are unitless.

**Key:** COPC=constituent of potential concern; DG=Disposal Group; FFTF=Fast Flux Test Facility.

**Table P-5. Long-Term Impacts of Chemical COPC Air Deposition on Terrestrial Resources at the Onsite Maximum-Exposure Location: Maximum Risk Index by Receptor**

Receptor	Alternative	Maximum Hazard Quotient	Chemical COPC
Plants	Alternative Combination 1	4.68×10 <sup>1</sup>	Toluene
Soil-dwelling invertebrates	Tank Closure Alternatives 3A, 3C	2.33	Mercury
Side-blotched lizard	Tank Closure Alternatives 3A, 3C	3.92×10 <sup>2</sup>	Mercury
Great Basin pocket mouse	Alternative Combination 1	2.12×10 <sup>3</sup>	Xylene
Coyote	Alternative Combination 1	2.69×10 <sup>2</sup>	Xylene
Mule deer	Waste Management Alternative 2, DG3	9.97×10 <sup>1</sup>	Formaldehyde
Western meadowlark	Tank Closure Alternatives 3A, 3C	2.35×10 <sup>2</sup>	Mercury
Mourning dove	Tank Closure Alternatives 3A, 3C	1.94×10 <sup>1</sup>	Mercury
Burrowing owl	Tank Closure Alternatives 3A, 3C	1.64×10 <sup>1</sup>	Mercury

**Note:** Risk indices are unitless.

**Key:** COPC=constituent of potential concern; DG=Disposal Group.

Exposure to chemicals from air emissions under all alternatives exceeds the Hazard Quotient criterion of 1 for one or more receptors at the onsite maximum-exposure location. The highest Hazard Quotient for each alternative or alternative combination was either for side-blotched lizard exposed to mercury or Great Basin pocket mouse exposed to xylene (see Table P-4). Mercury had the highest Hazard Quotient for soil-dwelling invertebrates, lizards, and birds (Tank Closure Alternatives 3A and 3C), and Hazard Quotients for mercury exceeded 1 for plants, soil-dwelling invertebrates, and mammals under one or more Tank Closure alternative and alternative combination, except for Tank Closure Alternative 1 and Alternative Combination 1. Xylene had the highest Hazard Quotient for the Great Basin pocket mouse and coyote (Alternative Combination 1), and Hazard Quotients for xylene exceeded 1 for mammals under all Tank Closure, FFTF Decommissioning, and Waste Management alternatives and alternative combinations. Toluene had the highest Hazard Quotient for plants (Alternative Combination 1), and formaldehyde had the highest Hazard Quotient for the mule deer (Waste Management Alternative 2, Disposal Group 3). Hazard Quotients for toluene exceeded 1 for mammals under all alternatives except Tank Closure Alternative 1, Waste Management Alternative 1, and the Idaho Option for FFTF Decommissioning Alternatives 2 and 3. Hazard Quotients for formaldehyde exceeded 1 for the Great Basin pocket mouse and mule deer under all Tank Closure, FFTF Decommissioning, and Waste Management alternatives except Tank Closure Alternative 1, Waste Management Alternative 1, FFTF Decommissioning Alternative 2, Idaho Option, and Alternative 3. The maximum Hazard Quotient from emissions under all alternatives was calculated to be 2,120 for the Great Basin pocket mouse exposed to xylene under Alternative Combination 1, which comprises the No Action Alternatives for Tank Closure, FFTF Decommissioning, and Waste Management (see Table P-5). Three other chemical COPCs, benzene, toluene, and mercury, had Hazard Quotients between 1 and 20 for terrestrial receptors at the onsite maximum-exposure location.

The benzene, toluene, xylene, and formaldehyde Hazard Quotients above 1 would be unlikely to indicate significant risks to mammals for three reasons. First, benzene, toluene, xylene, and formaldehyde concentrations were overestimated because these substances are expected to dissipate (volatilization, biodegradation), not accumulate in soil, as was assumed for the risk calculations. High-end estimates of the half-lives of benzene, toluene, xylene, and formaldehyde in soil are 39 days, 22 days, 28 days, and 20 days, respectively (Howard et al. 1991). Second, the soil-dwelling invertebrate *BAF-S* might have been overestimated. The *BAF-S* was based on a *Daphnia* BCF using a log  $K_{ow}$  regression applied to soil-dwelling invertebrates exposed to soil pore water in equilibrium with soil at 1 percent organic carbon. *Daphnia* are aquatic organisms, and uptake via water by aquatic biota is expected to be greater than uptake via soil water by terrestrial biota. The Great Basin pocket mouse feeds on soil-dwelling invertebrates, so an overestimate of the *BAF-S* would result in greater chemical intake via ingestion of

soil-dwelling invertebrates. Third, the use of lowest-observed adverse effect levels (LOAELs), which are greater than no-observed-adverse-effect levels, would result in further reduction of the Hazard Quotients. LOAELs are toxicological benchmarks associated with low levels of adverse effects on individuals, but which may not cause significant adverse impacts on populations. LOAELs are acceptable benchmarks for species that are not threatened or endangered. Thus, Hazard Quotients for the representative species likely overestimated the potential for adverse effects on onsite terrestrial resources.

The mercury Hazard Quotients above 1 do not necessarily indicate high risks to terrestrial ecological receptors at the onsite maximum-exposure location. The mercury TRV used to calculate the Hazard Quotients was the no-observed-adverse-effect level for methylmercury, which is highly toxic compared with the forms of mercury typically found in terrestrial environments. Mercury Hazard Quotients can be used to compare alternatives with confidence, but Hazard Quotients exceeding 1 should not be used as the basis to conclude that ecological resources at the onsite maximum-exposure location would be adversely impacted.

A potential adverse impact that could not be evaluated using the Hazard Quotients was the potential acidification of soil or water by deposition of the chemical COPCs nitrogen dioxide and sulfur dioxide. The deposition of nitrogen and sulfur dioxides in air emissions from site and WTP operations would be unlikely to acidify soil at Hanford. The Soil Survey for Benton County, Washington, describes the representative soil, the Quincy series, as ranging from mildly to moderately alkaline throughout (pH 7.8 to 8.4) and strongly effervescent in the lower part, indicating abundant calcium carbonate and acid-buffering capacity (Rasmussen 1971; NRCS 2008). The Quincy (Rupert) sand is derived from extensive alluvial and lacustrine flood deposits rather than from the basaltic rock in the area. The Burbank loamy sand, the second most widely distributed soil unit on the site, is very similar to the Quincy sand. The chemical properties table for Benton County does not indicate that the Quincy or Burbank soils are particularly saline. Soils in wetter regions of the western United States, especially soils derived from acidic parent materials, have little buffering capacity from calcium carbonate and other minerals because these minerals are leached out. In contrast, soils in arid regions such as Hanford tend to have a relatively high buffer capacity because soluble ions (particularly basic ions and associated minerals) tend to accumulate in the upper portion of the soil profile. With a pH (a measure of acidity/alkalinity) greater than 8 in the upper 20 centimeters (8 inches) according to the Natural Resources Conservation Service Soil Series Database and a reported soil pH of 7 for the 200 Area (Paragon Analytics 2003), soil acidification due to acid deposition from site and WTP emissions would not be a concern.

#### **P.2.2.2 Offsite Terrestrial Resources**

The results of the assessment of radioactive and chemical contaminant releases to air and subsequent deposition estimated for terrestrial receptors at the offsite maximum-exposure location under the various Tank Closure, FFTF Decommissioning, and Waste Management alternatives, as well as the alternative combinations, are summarized in Tables P-6, P-7, and P-8.

The maximum combined radiological Hazard Index from emissions under all alternatives was calculated to be 0.0000532 for the Great Basin pocket mouse under Tank Closure Alternative 6A, Option Case. Table P-6 presents the maximum Hazard Indices associated with air emissions calculated to reach the terrestrial receptors at the offsite maximum-exposure location (the Columbia River) under all alternatives. Exposure to radioactive COPCs from air emissions under all alternatives was below the 1-rad-per-day benchmark for soil-dwelling invertebrates and plants and the 0.1-rad-per-day benchmark for terrestrial wildlife receptors (i.e., Woodhouse's toad, mule deer, mourning dove, Great Basin pocket mouse, western meadowlark, coyote, and burrowing owl). Estimated hazards for the representative species indicated that no adverse effects are expected for offsite terrestrial receptors from exposure to radioactive COPCs from air emissions. Because the direct impacts of air exposure are expected to be small, any associated, potential indirect impacts on the ecosystem would be correspondingly minor.

**Table P-6. Long-Term Impacts of Radioactive COPC Air Deposition on Terrestrial Resources at the Offsite Maximum-Exposure Location: Maximum Hazard Indices by Receptor and Alternative**

Alternative	Maximum Hazard Index by Receptor								
	Plants	Soil Invertebrates	Woodhouse's Toad	Mule Deer	Mourning Dove	Great Basin Pocket Mouse	Western Meadowlark	Coyote	Burrowing Owl
<b>Tank Closure</b>									
1	1.16×10 <sup>-6</sup>	9.80×10 <sup>-6</sup>	1.16×10 <sup>-5</sup>	1.03×10 <sup>-5</sup>	1.40×10 <sup>-5</sup>	1.12×10 <sup>-5</sup>	1.37×10 <sup>-5</sup>	1.34×10 <sup>-5</sup>	1.21×10 <sup>-5</sup>
2A	1.08×10 <sup>-5</sup>	2.11×10 <sup>-5</sup>	1.77×10 <sup>-5</sup>	1.42×10 <sup>-5</sup>	3.42×10 <sup>-5</sup>	4.42×10 <sup>-5</sup>	2.45×10 <sup>-5</sup>	2.11×10 <sup>-5</sup>	2.92×10 <sup>-5</sup>
2B	1.03×10 <sup>-5</sup>	1.17×10 <sup>-5</sup>	8.67×10 <sup>-6</sup>	6.53×10 <sup>-6</sup>	2.27×10 <sup>-5</sup>	3.53×10 <sup>-5</sup>	1.33×10 <sup>-5</sup>	1.03×10 <sup>-5</sup>	1.95×10 <sup>-5</sup>
3A	1.04×10 <sup>-5</sup>	1.19×10 <sup>-5</sup>	1.43×10 <sup>-5</sup>	1.21×10 <sup>-5</sup>	2.85×10 <sup>-5</sup>	4.13×10 <sup>-5</sup>	1.90×10 <sup>-5</sup>	1.60×10 <sup>-5</sup>	2.54×10 <sup>-5</sup>
3B	9.55×10 <sup>-6</sup>	1.08×10 <sup>-5</sup>	6.12×10 <sup>-6</sup>	4.05×10 <sup>-6</sup>	1.96×10 <sup>-5</sup>	3.17×10 <sup>-5</sup>	1.05×10 <sup>-5</sup>	7.67×10 <sup>-6</sup>	1.66×10 <sup>-5</sup>
3C	1.05×10 <sup>-5</sup>	1.22×10 <sup>-5</sup>	1.49×10 <sup>-5</sup>	1.23×10 <sup>-5</sup>	2.90×10 <sup>-5</sup>	4.23×10 <sup>-5</sup>	1.93×10 <sup>-5</sup>	1.65×10 <sup>-5</sup>	2.60×10 <sup>-5</sup>
4	1.02×10 <sup>-5</sup>	1.16×10 <sup>-5</sup>	8.91×10 <sup>-6</sup>	6.71×10 <sup>-6</sup>	2.32×10 <sup>-5</sup>	3.60×10 <sup>-5</sup>	1.36×10 <sup>-5</sup>	1.06×10 <sup>-5</sup>	2.00×10 <sup>-5</sup>
5	9.65×10 <sup>-6</sup>	1.11×10 <sup>-5</sup>	1.11×10 <sup>-5</sup>	8.94×10 <sup>-6</sup>	2.47×10 <sup>-5</sup>	3.71×10 <sup>-5</sup>	1.55×10 <sup>-5</sup>	1.27×10 <sup>-5</sup>	2.17×10 <sup>-5</sup>
6A, Base Case	1.20×10 <sup>-5</sup>	1.50×10 <sup>-5</sup>	8.14×10 <sup>-6</sup>	4.79×10 <sup>-6</sup>	2.83×10 <sup>-5</sup>	4.73×10 <sup>-5</sup>	1.49×10 <sup>-5</sup>	1.05×10 <sup>-5</sup>	2.43×10 <sup>-5</sup>
6A, Option Case	1.32×10 <sup>-5</sup>	1.71×10 <sup>-5</sup>	9.20×10 <sup>-6</sup>	5.42×10 <sup>-6</sup>	3.17×10 <sup>-5</sup>	<b>5.32×10<sup>-5</sup></b>	1.68×10 <sup>-5</sup>	1.19×10 <sup>-5</sup>	2.73×10 <sup>-5</sup>
6B, Base Case	1.24×10 <sup>-5</sup>	1.56×10 <sup>-5</sup>	1.13×10 <sup>-5</sup>	7.86×10 <sup>-6</sup>	3.19×10 <sup>-5</sup>	5.15×10 <sup>-5</sup>	1.82×10 <sup>-5</sup>	1.38×10 <sup>-5</sup>	2.79×10 <sup>-5</sup>
6B, Option Case	1.28×10 <sup>-5</sup>	1.64×10 <sup>-5</sup>	1.16×10 <sup>-5</sup>	8.14×10 <sup>-6</sup>	3.25×10 <sup>-5</sup>	5.22×10 <sup>-5</sup>	1.86×10 <sup>-5</sup>	1.41×10 <sup>-5</sup>	2.83×10 <sup>-5</sup>
6C	9.88×10 <sup>-6</sup>	1.15×10 <sup>-5</sup>	8.67×10 <sup>-6</sup>	6.50×10 <sup>-6</sup>	2.26×10 <sup>-5</sup>	3.53×10 <sup>-5</sup>	1.32×10 <sup>-5</sup>	1.03×10 <sup>-5</sup>	1.95×10 <sup>-5</sup>
<b>FFTF Decommissioning</b>									
1	0	0	0	0	0	0	0	0	0
2, Hanford Option	1.33×10 <sup>-8</sup>	1.64×10 <sup>-8</sup>	1.58×10 <sup>-7</sup>	1.31×10 <sup>-7</sup>	2.18×10 <sup>-7</sup>	1.62×10 <sup>-7</sup>	2.07×10 <sup>-7</sup>	2.35×10 <sup>-7</sup>	2.19×10 <sup>-7</sup>
2, Idaho Option	5.52×10 <sup>-15</sup>	7.57×10 <sup>-14</sup>	6.72×10 <sup>-14</sup>	3.02×10 <sup>-14</sup>	2.23×10 <sup>-13</sup>	4.14×10 <sup>-13</sup>	1.28×10 <sup>-13</sup>	8.97×10 <sup>-14</sup>	2.16×10 <sup>-13</sup>
3, Hanford Option	1.33×10 <sup>-8</sup>	1.64×10 <sup>-8</sup>	1.58×10 <sup>-7</sup>	1.31×10 <sup>-7</sup>	2.18×10 <sup>-7</sup>	1.62×10 <sup>-7</sup>	2.07×10 <sup>-7</sup>	2.35×10 <sup>-7</sup>	2.19×10 <sup>-7</sup>
3, Idaho Option	0	0	0	0	0	0	0	0	0
<b>Waste Management</b>									
1	0	0	0	0	0	0	0	0	0
2, DG1	2.19×10 <sup>-13</sup>	2.23×10 <sup>-12</sup>	4.53×10 <sup>-15</sup>	1.52×10 <sup>-14</sup>	6.25×10 <sup>-14</sup>	3.23×10 <sup>-14</sup>	6.21×10 <sup>-15</sup>	5.10×10 <sup>-15</sup>	4.62×10 <sup>-15</sup>
2, DG2	2.19×10 <sup>-13</sup>	2.23×10 <sup>-12</sup>	4.53×10 <sup>-15</sup>	1.52×10 <sup>-14</sup>	6.25×10 <sup>-14</sup>	3.23×10 <sup>-14</sup>	6.21×10 <sup>-15</sup>	5.10×10 <sup>-15</sup>	4.62×10 <sup>-15</sup>
2, DG3	2.19×10 <sup>-13</sup>	2.23×10 <sup>-12</sup>	4.53×10 <sup>-15</sup>	1.52×10 <sup>-14</sup>	6.25×10 <sup>-14</sup>	3.23×10 <sup>-14</sup>	6.21×10 <sup>-15</sup>	5.10×10 <sup>-15</sup>	4.62×10 <sup>-15</sup>
3, DG1	2.19×10 <sup>-13</sup>	2.23×10 <sup>-12</sup>	4.53×10 <sup>-15</sup>	1.52×10 <sup>-14</sup>	6.25×10 <sup>-14</sup>	3.23×10 <sup>-14</sup>	6.21×10 <sup>-15</sup>	5.10×10 <sup>-15</sup>	4.62×10 <sup>-15</sup>
3, DG2	2.19×10 <sup>-13</sup>	2.23×10 <sup>-12</sup>	4.53×10 <sup>-15</sup>	1.52×10 <sup>-14</sup>	6.25×10 <sup>-14</sup>	3.23×10 <sup>-14</sup>	6.21×10 <sup>-15</sup>	5.10×10 <sup>-15</sup>	4.62×10 <sup>-15</sup>
3, DG3	2.19×10 <sup>-13</sup>	2.23×10 <sup>-12</sup>	4.53×10 <sup>-15</sup>	1.52×10 <sup>-14</sup>	6.25×10 <sup>-14</sup>	3.23×10 <sup>-14</sup>	6.21×10 <sup>-15</sup>	5.10×10 <sup>-15</sup>	4.62×10 <sup>-15</sup>
<b>Alternative Combination</b>									
1	1.16×10 <sup>-6</sup>	9.80×10 <sup>-6</sup>	1.16×10 <sup>-5</sup>	1.03×10 <sup>-5</sup>	1.40×10 <sup>-5</sup>	1.12×10 <sup>-5</sup>	1.37×10 <sup>-5</sup>	1.34×10 <sup>-5</sup>	1.21×10 <sup>-5</sup>
2	1.03×10 <sup>-5</sup>	1.17×10 <sup>-5</sup>	8.83×10 <sup>-6</sup>	6.66×10 <sup>-6</sup>	2.30×10 <sup>-5</sup>	3.55×10 <sup>-5</sup>	1.35×10 <sup>-5</sup>	1.05×10 <sup>-5</sup>	1.98×10 <sup>-5</sup>
3	1.25×10 <sup>-5</sup>	1.56×10 <sup>-5</sup>	1.15×10 <sup>-5</sup>	7.99×10 <sup>-6</sup>	3.21×10 <sup>-5</sup>	5.16×10 <sup>-5</sup>	1.84×10 <sup>-5</sup>	1.40×10 <sup>-5</sup>	2.81×10 <sup>-5</sup>

**Note:** The maximum Hazard Index is indicated by **bold** text. Hazard Index is unitless.

**Key:** COPC=constituent of potential concern; DG=Disposal Group; FFTF=Fast Flux Test Facility.

**Table P-7. Long-Term Impacts of Chemical COPC Air Deposition on Terrestrial Resources at the Offsite Maximum-Exposure Location: Maximum Risk Index by Alternative**

Alternative	Maximum Hazard Quotient	Chemical COPC	Receptor
<b>Tank Closure</b>			
1	4.20×10 <sup>-3</sup>	Xylene	Great Basin pocket mouse
2A	3.30×10 <sup>-1</sup>	Mercury	Western meadowlark
2B	3.60×10 <sup>-1</sup>	Mercury	Western meadowlark
3A	4.30×10 <sup>-1</sup>	Mercury	Western meadowlark
3B	2.45×10 <sup>-1</sup>	Mercury	Western meadowlark
3C	4.30×10 <sup>-1</sup>	Mercury	Western meadowlark
4	3.10×10 <sup>-1</sup>	Mercury	Western meadowlark
5	2.96×10 <sup>-1</sup>	Mercury	Western meadowlark
6A, Base Case	3.33×10 <sup>-1</sup>	Mercury	Western meadowlark
6A, Option Case	3.32×10 <sup>-1</sup>	Mercury	Western meadowlark
6B, Base Case	3.73×10 <sup>-1</sup>	Mercury	Western meadowlark
6B, Option Case	3.73×10 <sup>-1</sup>	Mercury	Western meadowlark
6C	3.73×10 <sup>-1</sup>	Mercury	Western meadowlark
<b>FFTF Decommissioning</b>			
1	2.41	Xylene	Great Basin pocket mouse
2, Hanford Option	8.69×10 <sup>-3</sup>	Xylene	Great Basin pocket mouse
2, Idaho Option	4.22×10 <sup>-3</sup>	Xylene	Great Basin pocket mouse
3, Hanford Option	8.75×10 <sup>-3</sup>	Xylene	Great Basin pocket mouse
3, Idaho Option	4.28×10 <sup>-3</sup>	Xylene	Great Basin pocket mouse
<b>Waste Management</b>			
1	3.53×10 <sup>-3</sup>	Xylene	Great Basin pocket mouse
2, DG1	1.01×10 <sup>-1</sup>	Xylene	Great Basin pocket mouse
2, DG2	3.95×10 <sup>-1</sup>	Xylene	Great Basin pocket mouse
2, DG3	5.32×10 <sup>-1</sup>	Xylene	Great Basin pocket mouse
3, DG1	9.85×10 <sup>-2</sup>	Xylene	Great Basin pocket mouse
3, DG2	3.93×10 <sup>-1</sup>	Xylene	Great Basin pocket mouse
3, DG3	5.30×10 <sup>-1</sup>	Xylene	Great Basin pocket mouse
<b>Alternative Combination</b>			
1	<b>2.42</b>	Xylene	Great Basin pocket mouse
2	3.60×10 <sup>-1</sup>	Mercury	Western meadowlark
3	5.73×10 <sup>-1</sup>	Xylene	Great Basin pocket mouse

**Note:** The maximum Hazard Quotient of all receptors is indicated by **bold** text. Risk indices are unitless.

**Key:** COPC=constituent of potential concern; DG=Disposal Group; FFTF=Fast Flux Test Facility.

**Table P-8. Long-Term Impacts of Chemical COPC Air Deposition on Terrestrial Resources at the Offsite Maximum-Exposure Location: Maximum Risk Index by Receptor**

Receptor	Alternative	Maximum Hazard Quotient	Chemical COPC
Plants	Alternative Combination 1	$5.34 \times 10^{-2}$	Toluene
Soil-dwelling invertebrates	Tank Closure Alternatives 3A, 3C	$4.26 \times 10^{-3}$	Mercury
Woodhouse's toad	Tank Closure Alternatives 3A, 3C	$2.97 \times 10^{-1}$	Mercury
Great Basin pocket mouse	Alternative Combination 1	2.42	Xylene
Coyote	Alternative Combination 1	$3.07 \times 10^{-1}$	Xylene
Mule deer	Waste Management Alternative 2, DG3	$1.16 \times 10^{-1}$	Formaldehyde
Western meadowlark	Tank Closure Alternatives 3A, 3C	$4.30 \times 10^{-1}$	Mercury
Mourning dove	Tank Closure Alternatives 3A, 3C	$3.55 \times 10^{-2}$	Mercury
Burrowing owl	Tank Closure Alternatives 3A, 3C	$2.99 \times 10^{-2}$	Mercury

**Note:** Risk indices are unitless.

**Key:** COPC=constituent of potential concern; DG=Disposal Group.

Exposures to chemicals from air emissions under all alternatives would exceed the Hazard Quotient criterion of 1 only for the Great Basin pocket mouse exposed to xylene under FFTF Decommissioning Alternative 1 and Alternative Combination 1, which includes FFTF Decommissioning Alternative 1 (see Table P-7). No other chemical COPCs had Hazard Quotients exceeding 1 for terrestrial receptors at the offsite maximum-exposure location. The maximum Hazard Quotient from emissions under all alternatives for all receptors was calculated to be 2.42 for the Great Basin pocket mouse exposed to xylene (see Table P-8). The highest Hazard Quotient for each alternative or alternative combination was either for the western meadowlark exposed to mercury or the Great Basin pocket mouse exposed to xylene (see Table P-7). Table P-8 summarizes the maximum Hazard Quotient for each receptor. Mercury had the highest Hazard Quotient for soil-dwelling invertebrates, the Woodhouse's toad, and the three bird species—mourning dove, western meadowlark, and burrowing owl (Tank Closure Alternatives 3A and 3C). Xylene had the highest Hazard Quotient for the Great Basin pocket mouse and the coyote (Alternative Combination 1). Toluene had the highest Hazard Quotient for plants (Alternative Combination 1), and formaldehyde the highest Hazard Quotient for the mule deer (Waste Management Alternative 2, Disposal Group 3).

Estimated hazards for the representative species indicate that no adverse effects are expected for offsite terrestrial receptors from exposure to chemicals from air emissions. The xylene Hazard Quotients above 1 are unlikely to indicate significant risk to small mammals for the reasons discussed for the onsite terrestrial maximum-exposure location, i.e., short environmental half-life, overestimated bioaccumulation, and conservative toxicological benchmarks (see Section P.2.2.1). Because the direct impacts of air exposure are expected to be small, any associated, potential indirect impacts on the ecosystem would be correspondingly minor.

As discussed in Section P.2.2.1, the deposition of nitrogen and sulfur dioxides in air emissions from the Tank Closure, FFTF Decommissioning, and Waste Management alternatives would be unlikely to acidify offsite soils because of the natural buffering capacity of area soils. Thus, soil acidification due to deposition of chemical COPCs from site and WTP emissions would not be a concern.

### **P.2.2.3      Uncertainties**

Uncertainty exists about the actual magnitude of future exposures and the threshold doses or benchmark concentration TRVs used to evaluate the long-term impact on terrestrial ecological resources of air releases. The uncertainties for chemical and radiological exposure estimates come from uncertainties in the source terms and transport models. Additional uncertainties are found in the BAFs and uptake factors,

which are linear models based on simplifying assumptions. The uncertainties for toxicity and radiological effects thresholds arise from extrapolating from laboratory experiments on test species to Hanford receptor species in natural environments, as well as uncertainty about the chemical to which ecological receptors would be exposed (e.g., chemical COPC breakdown products, which can have greater toxicity than the COPC itself). The lack of TRVs for some chemical COPCs and some receptors also results in uncertainties. TRVs for some chemical COPCs were not available for soil-dwelling invertebrates or the Woodhouse's toad, western meadowlark, mourning dove, and burrowing owl. As a result, there were uncertainties associated with the ecological risk evaluation. For example, it was not known whether these receptors would be more sensitive than mammals. The effects of chemicals deposited on microbial crusts also were not known. Together, these uncertainties produced limited underestimates of risk and moderate overestimates of risk for different combinations of receptors and chemical or radioactive COPCs. The effects of these uncertainties were unbiased with respect to the alternatives being evaluated in this *TC & WM EIS*; thus, the results presented above accurately reflect the relative impacts of alternatives on ecological resources. In addition, conservative exposure assumptions and TRVs mitigated these uncertainties and allowed for confidence in "no risk" conclusions.

### **P.2.3 Summary of Terrestrial Impacts**

Estimated radiation doses resulting from any of the alternatives were less than the 0.1-rad-per-day benchmark and did not exceed the 1-rad-per-day benchmark for terrestrial receptors at the on- and offsite maximum-exposure locations. All of the Hazard Indices associated with these alternatives were below 1. Estimated chemical doses resulting from any of the alternatives exceeded the Hazard Quotient criterion of 1 at the offsite terrestrial maximum-exposure location (the Columbia River) only for the Great Basin pocket mouse exposed to xylene under FFTF Decommissioning Alternative 1 and Alternative Combination 1, which includes FFTF Decommissioning Alternative 1. The low magnitude of the Hazard Quotients and the conservative exposure assumptions mean that long-term impacts on populations of small mammals under these alternatives would not be likely at the offsite maximum-exposure location. Although there were Hazard Quotients above 1 for mammals exposed to xylene and for plants, soil-dwelling invertebrates, lizards, mammals, and birds exposed to mercury at the onsite maximum-exposure location for many alternatives, the conservative exposure assumptions and toxicity benchmarks suggest that adverse impacts, while possible, would not be likely. Calculated risk indices for terrestrial resources from air releases were used in this *TC & WM EIS* to compare alternatives (Chapter 5) and evaluate cumulative impacts (Chapter 6).

## **P.3 IMPACTS ON COLUMBIA RIVER AQUATIC AND RIPARIAN RESOURCES RESULTING FROM CONTAMINANT RELEASES**

Ecological resources in the Columbia River and its riparian habitat would potentially be adversely impacted by two types of contaminant releases: air releases during site and WTP operations in the near-term future and groundwater releases in the distant future. The different actions involved in the different alternatives would result in different amounts and timing of air releases, different amounts of waste remaining in the tanks, and different waste forms disposed of at the site, thereby potentially contributing to future groundwater releases to the Columbia River. The focus was on long-term future impacts on the river because no additional fast-moving substances would be added to the tanks under any of the alternatives. Groundwater modeling for Hanford has shown that the discharge of fast-moving substances in the plumes has already peaked, and there is no evidence of adverse impacts on aquatic and riparian receptors (Bryce et al. 2002). Concentrations of radionuclides and chemicals resulting from deposition of airborne contaminants were predicted over the construction and operation periods associated with the alternatives, as described in Appendix G. Groundwater contaminated by leaching from the 200 Areas would eventually reach and discharge into the Hanford Reach of the Columbia River; discharges over 10,000 years were predicted, as described in Appendix O. These predicted release

concentrations were used to evaluate the long-term impacts on Columbia River aquatic and riparian ecological resources.

The potential for adverse effects on Columbia River aquatic and riparian ecological resources resulting from potential releases of radionuclides and chemicals through air emissions during waste handling and WTP operations, as well as future groundwater releases under the different alternatives, was evaluated using a quantitative risk assessment approach (63 FR 26846; EPA 1992, 1997). The general approach to this assessment is discussed in Section P.2.1. Impacts of sulfur and nitrogen oxide deposition on the water's pH were evaluated based on buffering capacity and predicted concentrations.

### **P.3.1 Impacts of Air Releases During Operations**

Potential adverse impacts on Columbia River aquatic and riparian ecological resources resulting from air releases of radionuclides or chemicals during WTP operations were evaluated for all alternatives. Under all alternatives, radionuclides and chemicals emitted to the air during WTP operations would potentially be transported away from the source to the Columbia River and to offsite terrestrial locations. The potential impacts on terrestrial ecological resources (i.e., terrestrial biota) at the offsite maximum-exposure location (the Columbia River) of contaminants released by air emission are discussed in Section P.2. The evaluation of potential adverse impacts on aquatic and riparian ecological resources (e.g., aquatic biota and their predators) at the Columbia River is described below.

#### **P.3.1.1 Methods**

The general approach for assessing potential adverse effects on aquatic and riparian ecological resources is discussed in Section P.2.1. The assumptions; receptors; exposure pathways and uptake mechanisms (routes); predicted air, soil, sediment, and surface-water concentrations; exposure model equations; and benchmarks used to model exposure for aquatic and riparian ecological resources potentially impacted by contaminant releases are described in the relevant sections below. The quantitative evaluations of long-term adverse impacts on aquatic and riparian resources of air releases, based on Hazard Quotients, Hazard Indices, and river water pH, are summarized and discussed in Section P.3.1.2. Impacts of sulfur and nitrogen oxide deposition on the pH were evaluated based on buffering capacity and predicted concentrations.

##### **P.3.1.1.1 Key Assumptions**

The following key assumptions were made in the evaluation of potential impacts on Columbia River aquatic and riparian resources resulting from exposure to radionuclides and chemicals released to air during closure operations:

- There would be no riparian soil contamination prior to tank closure activities.
- Soil contamination from air releases would not coincide with soil contamination from groundwater releases because material released to air during site and WTP operations would dissipate before slow-moving constituents discharge to riparian soil at the Columbia River.
- The concentrations of constituents in the tissues of fish preyed upon by predators (least weasel and bald eagle) would be in equilibrium with the concentrations in nearshore surface water.
- The concentrations of inorganic chemical and radioactive COPCs in Columbia River nearshore sediment would be equal to riparian soil concentrations.
- The concentrations of organic chemical COPCs in Columbia River sediment would be in equilibrium with concentrations in nearshore surface water.

These assumptions allowed a conservative assessment of the impact of air releases on ecological resources.

#### **P.3.1.1.2 Receptors and Exposure Pathways and Routes**

The receptors selected to represent the Columbia River aquatic and riparian ecological resources, including special status species (see Chapter 3, Section 3.2.7.4), are listed in Table P-2. These receptors were selected because they are among those expected to have higher exposures than those not selected from their group due to their higher ingestion rates per unit body weight for prey, water, and sediment or soil. Special status species are not expected to be more highly exposed or more sensitive to contaminants than the selected species. The selected representative receptors were sediment-dwelling benthic invertebrates, aquatic biota, including Woodhouse's toad tadpoles and salmonids, raccoon, spotted sandpiper, least weasel, and bald eagle. All were ECEM receptors except the spotted sandpiper, which was substituted for the common snipe because the spotted sandpiper has a more aquatic diet.

The exposure pathways evaluated in the ecological risk analysis for this *TC & WM EIS* are shown in Table P-2 for all ecological receptors. The exposure medium, exposure route, and receptor are indicated for each pathway evaluated in the analysis of impacts on aquatic and riparian resources of air releases.

#### **P.3.1.1.3 Predicted Sediment and Surface-Water Concentrations**

The riparian soil, sediment, and surface-water concentrations under Tank Closure Alternatives 1 through 6C; FFTF Decommissioning Alternatives 1, 2, and 3 (Hanford and Idaho Options); and Waste Management Alternatives 1, 2, and 3 were calculated from the modeled air deposition rates at the Columbia River (see Appendix G). The riparian soil concentrations resulting from air deposition would be cumulative and were calculated assuming deposition on the riparian shoreline and accumulation on the ground surface over the operations period. Sediment concentrations of inorganic chemical and radioactive COPCs would be the cumulative soil concentrations calculated as described in Section P.2.1. Sediment concentrations of organic chemical COPCs were calculated as the product of the maximum nearshore surface-water concentration, the organic carbon-partitioning coefficient ( $K_{oc}$ ) and the fraction of organic carbon content, which was conservatively assumed to be 0.04, four times greater than the ECEM value (DOE 1998). The maximum nearshore surface-water concentration ( $C_w$ ) and water-column surface-water concentration ( $C_{wc}$ ) were calculated assuming that the amount of material deposited on the water surface of the Hanford Reach on an annual basis is mixed into a 0.5-meter-deep (1.6-foot-deep) nearshore zone extending 40 meters (44 yards) into the river and throughout the water column. The resulting sediment and surface-water concentrations under Tank Closure Alternatives 1 through 6C; FFTF Decommissioning Alternatives 1, 2, and 3 (Hanford and Idaho Options); and Waste Management Alternatives 1, 2, and 3 were used as the source terms in the exposure model described below.

#### **P.3.1.1.4 Exposure Model Calculations**

The exposure model calculated external and internal doses from radioactive COPCs for all receptors and ingestion doses from chemical COPCs for wildlife receptors. To calculate internal doses for radioactive COPCs in receptors exposed by direct contact with sediment (benthic invertebrates) and surface water (aquatic biota, including salmonids) and to calculate the ingested doses for wildlife receptors exposed to chemical COPCs in these biota (spotted sandpipers, raccoons, least weasels, and bald eagles), the concentrations of radioactive and chemical COPCs in benthic invertebrates and aquatic biota were required.

For benthic invertebrates, the concentration of COPCs was calculated as follows:

$$C_a = C_{\text{sed}} \times BASF$$

For trophic-level-3 fish (salmonids), the concentration was calculated as follows:

$$C_a = C_w \times BCF_{\text{fish}} \times FCM_3 \times CF$$

where:

$C_a$	=	concentration in animal food, milligrams per kilogram wet weight or picocuries per gram wet weight
$C_{\text{sed}}$	=	sediment concentration, milligrams per kilogram dry sediment or picocuries per gram dry sediment
$BASF$	=	sediment-to-benthic invertebrate bioaccumulation factor, kilograms dry sediment per kilogram wet tissue
$C_w$	=	nearshore surface-water concentration, milligrams per liter or picocuries per liter
$BCF_{\text{fish}}$	=	water-to-fish bioconcentration factor, liters water per kilogram wet tissue
$FCM_3$	=	food chain multiplier for trophic-level-3 fish
$CF$	=	unit conversion factor, 1 kilogram per kilogram for chemical COPCs, 0.001 kilograms per gram for radioactive COPCs

Food chain multipliers are factors accounting for the accumulation and biomagnification in fish via the food web (EPA 1995).

#### **P.3.1.1.4.1 External Doses from Radionuclides**

External doses to all aquatic receptors would result from exposure to radioactive COPCs in soil, air, water, and sediment. External doses to Woodhouse's toad adults from radionuclides in soil and air are evaluated in Section P.2.2. Exposure of Woodhouse's toad tadpoles was evaluated along with aquatic biota and salmonids. Wildlife receptors (raccoon, spotted sandpiper, bald eagle, and least weasel) would be exposed externally to radionuclides in soil, air, and water. External radiation from soil, sediment, and water was modeled as described in *Methodology for Estimating Radiation Dose Rates to Freshwater Biota Exposed to Radionuclides in the Environment* (Blaylock, Frank, and O'Neal 1993). External radiation doses for aquatic biota, including Woodhouse's toad tadpoles and salmonids, raccoons, spotted sandpipers, benthic invertebrates, bald eagles, and least weasels were adjusted for the fraction of time the receptors were assumed to be immersed in water away from sediment, sufficiently near the water to receive external radiation, on nearshore soil, resting on sediment, and immersed in sediment (see Table P-2). Those fractions (based on professional judgment) were as follows:

- Aquatic biota: immersed in water, 0.9; resting on sediment, 0.1; and immersed in sediment, 0.
- Raccoon: near water, 0.083; above ground, 0.5; below ground, 0.5; resting on sediment, 0; and immersed in sediment, 0.
- Spotted sandpiper: near water, 0.5; above ground, 1; resting on sediment, 0; and immersed in sediment, 0.
- Benthic invertebrates: immersed in sediment, 0.9; immersed in water, 0.1; and resting on sediment, 0.
- Bald eagle: near water, 0.05; resting on sediment, 0; and immersed in sediment, 0.
- Least weasel: immersed in water, 0.2; above ground, 0.5; below ground, 0.5; resting on sediment, 0; and immersed in sediment, 0.

For this *TC & WM EIS*, aquatic biota and benthic invertebrates were assumed to spend their entire lives in water. Therefore, the fractions of time spent immersed in water ( $F_{imm}$ ), at the sediment-water interface ( $F_s$ ), and immersed in sediment ( $F_{in}$ ) sum to unity (1) for these receptors. For aquatic biota and benthic invertebrates,  $F_{imm}$  can be calculated by subtraction (i.e.,  $1 - F_s - F_{in}$ ).

The external doses (rad per day) to all aquatic receptors from water and sediment were calculated, respectively, as follows:

$$RD_{Ext-water, imm} = C_w \times DF_{water, imm}$$

and

$$RD_{Ext-sed} = C_{sed} \times DF_{sediment}$$

where:

$RD_{Ext-water, imm}$	=	external radiation dose from immersion in water, rad per day
$C_w$	=	total activity of radioactive COPC in water, picocuries per liter
$DF_{water, imm}$	=	factor for converting activity in water to external dose from water immersion
$RD_{Ext-sed}$	=	external radiation dose from sediment, rad per day
$C_{sed}$	=	activity of radionuclide in sediment, picocuries per gram
$DF_{sediment}$	=	factor for converting activity in sediment to external dose from sediment

The external dose factor for immersion in water ( $DF_{water, imm}$ ) was calculated as follows (Blaylock, Frank, and O'Neal 1993):

$$DF_{water, imm} = (F_{imm}) \times 0.001 \times CFa \times [(1 - \Phi_\beta) \times E_\beta n_\beta + (1 - \Phi_\gamma) \times E_\gamma n_\gamma]$$

where:

$F_{imm}$	=	fraction of time receptor spends immersed in water, unitless
0.001	=	factor for converting liters to grams
$CFa$	=	unit conversion factor, $5.11 \times 10^{-5}$ rad per day per picocurie per gram per MeV per disintegration
$\Phi_\beta$	=	absorbed fraction of energy from beta energy $E_\beta$
$E_\beta n_\beta$	=	average energy emitted as beta radiation, MeV per disintegration $\times$ proportion of disintegrations producing a beta particle
$\Phi_\gamma$	=	absorbed fraction of energy from gamma energy $E_\gamma$
$E_\gamma n_\gamma$	=	photon energy emitted during transition from a higher to a lower energy state, MeV $\times$ proportion of disintegrations producing gamma radiation

Values of  $F_{imm}$  are given in the first paragraph of this subsection. The calculation of exposure of ecological receptors to radioactive COPCs in sediment included the dose from the decay products, known as daughters. This conservative approach to calculating dose was adopted because sediment is likely to have a longer residence time than water and air, and radioactive COPCs and their daughters would remain longer in sediment than in soil; soil-loss processes are ignored in the calculation of dose from COPCs in soil. The activity of each of the daughter radionuclides equals the activity of the parent multiplied by the fraction of the decays in the immediately preceding generation that yield the daughter. Exposure factors for the daughter radionuclides were used to calculate the contribution of the daughters to the summed exposure from the parent and all daughter radionuclides for both external and internal radiation doses from radioactive COPCs in sediment.

The external dose factor for sediment ( $DF_{\text{sediment}}$ ) was calculated as follows (Blaylock, Frank, and O'Neal 1993):

$$DF_{\text{sediment}} = (0.5 \times F_s + F_{\text{in}}) \times CFa \times [(1 - \Phi_\beta) \times E_\beta n_\beta + (1 - \Phi_\gamma) \times E_\gamma n_\gamma]$$

where:

- 0.5 = factor accounting for assumption that a receptor at the sediment–water interface receives external radiation from sediment only from below, so the dose is only half of the dose from immersion
- $F_s$  = fraction of time receptor spends at the sediment–water interface, unitless
- $F_{\text{in}}$  = fraction of time receptor spends buried in sediment, unitless
- $CFa$  = unit conversion factor,  $5.11 \times 10^{-5}$  rad per day per picocurie per gram per MeV per disintegration
- $\Phi_\beta$  = absorbed fraction of energy from beta energy  $E_\beta$
- $E_\beta n_\beta$  = average energy emitted as beta radiation, MeV per disintegration  $\times$  proportion of disintegrations producing a beta particle
- $\Phi_\gamma$  = absorbed fraction of energy from gamma energy  $E_\gamma$
- $E_\gamma n_\gamma$  = photon energy emitted during transition from a higher to a lower energy state, MeV  $\times$  proportion of disintegrations producing gamma radiation

Values of  $F_s$  and  $F_{\text{in}}$  are given in the first paragraph of this subsection. To calculate external exposure to all aquatic receptors from radioactive COPCs in water and sediment,  $DF_{\text{water, imm}}$  and  $DF_{\text{sediment}}$  values were multiplied by the modeled activities of the corresponding radionuclides in surface water and the corresponding radionuclides and their daughters in sediment.

The external dose (rad per day) to all wildlife receptors from air (Eckerman and Ryman 1993) was calculated per the equations presented in Section P.2.1.4. To calculate external exposure to all aquatic receptors from radioactive COPCs in air, DCF values were multiplied by the modeled activities of the corresponding radionuclides in air.

The external dose (rad per day) for all wildlife receptors from proximity to water containing radioactive COPCs was calculated as follows (Eckerman and Ryman 1993):

$$RD_{\text{Ext-water, near}} = C_w \times DF_{\text{water, near}}$$

where:

- $RD_{\text{Ext-water, near}}$  = external radiation dose from proximity to water, rad per day
- $C_w$  = total activity of radioactive COPC in nearshore surface water, picocuries per liter
- $DF_{\text{water, near}}$  = factor for converting activity in water to external dose from water

The external dose factor for water ( $DF_{\text{water, near}}$ ) for wildlife receptors was calculated as follows (Blaylock, Frank, and O'Neal 1993):

$$DF_{\text{water, near}} = C_w \times F_{\text{near}} \times 0.001 \times CFa \times [(1 - \Phi_\gamma) \times E_\gamma n_\gamma]$$

where:

- $C_w$  = total activity of radioactive COPC in nearshore surface water, picocuries per liter
- $F_{\text{near}}$  = fraction of time receptor spends near the water, unitless
- 0.001 = factor for converting liters to grams

- $CFa$  = unit conversion factor,  $5.11 \times 10^{-5}$  rad per day per picocuries per gram per MeV per disintegration  
 $\Phi_\gamma$  = absorbed fraction of energy from gamma energy  $E_\gamma$   
 $E_\gamma n_\gamma$  = photon energy emitted during transition from a higher to a lower energy state, MeV  $\times$  proportion of disintegrations producing gamma radiation

To calculate external exposure to all aquatic receptors from radioactive COPCs in water,  $DF_{\text{water, near}}$  values were multiplied by the modeled total activities of the corresponding radionuclides in surface water.

#### P.3.1.1.4.2 Internal Doses from Radionuclides

Internal exposure to radionuclides was calculated from activity in tissues, rather than from daily ingestion, using the equations presented in Section P.2.1.4. The internal activities of radioactive COPCs were calculated by using BAFs and BCFs, along with radioactive COPC activities in sediment and water. For radionuclides in sediment, radiation by daughter radionuclides was also included in internal dose calculations. Decay energies and absorption fractions for gamma radiation for radioactive COPCs and daughter radionuclides came from Eckerman and Ryman (1993); Blaylock, Frank, and O'Neal (1993); and Sample et al. (1997).

The internal dose to aquatic receptors and wildlife receptors was calculated as follows (Sample et al. 1997):

$$RD_{\text{Int}} = C_n \times DF_{\text{Int}}$$

where:

$$DF_{\text{Int}} = CFa \times (QF \times E_\alpha n_\alpha \times \Phi_\alpha + E_\beta n_\beta \times \Phi_\beta + E_\gamma n_\gamma \times \Phi_\gamma)$$

and

- $RD_{\text{Int}}$  = internal radiation dose from ingestion of radioactive COPCs, rad per day  
 $C_n$  = activity of radionuclide in receptor tissue, picocuries per gram  
 $DF_{\text{Int}}$  = factor for converting radioactive COPC activity in tissue to internal dose  
 $CFa$  = unit conversion factor,  $5.11 \times 10^{-5}$  rad per day per picocuries per gram per MeV per disintegration  
 $QF$  = 5, quality factor for biological effect of alpha radiation (Kocher and Trabalka 2000), unitless  
 $E_\alpha n_\alpha$  = average energy emitted as alpha radiation, MeV per disintegration  $\times$  proportion of disintegrations producing an alpha particle  
 $\Phi_\alpha$  = absorbed fraction of energy from alpha energy  $E_\alpha$   
 $E_\beta n_\beta$  = average energy emitted as beta radiation, MeV per disintegration  $\times$  proportion of disintegrations producing a beta particle  
 $\Phi_\beta$  = absorbed fraction of energy from beta energy  $E_\beta$   
 $E_\gamma n_\gamma$  = photon energy emitted during transition from a higher to a lower energy state, MeV  $\times$  proportion of disintegrations producing gamma radiation  
 $\Phi_\gamma$  = absorbed fraction of energy from gamma energy  $E_\gamma$

To calculate internal exposure to all aquatic receptors from ingested radioactive COPCs,  $DF_{\text{Int}}$  values were multiplied by the modeled activities of the corresponding radionuclides in receptor tissues. For receptors ingesting sediment or prey exposed to sediment, only the fraction of tissue activity or concentration coming from sediment directly or indirectly through ingested prey was multiplied by the  $DF_{\text{Int}}$  values for daughters of radioactive COPCs.

Following the approach for terrestrial plants (see Section P.2.1.4), the concentration of carbon-14 in benthic invertebrates was calculated assuming that the ratio of carbon-14 to the natural carbon in tissue would be equal to the ratio of carbon-14 to the natural carbon in Columbia River nearshore surface water:

$$C_a = C_w \times 0.11/0.014$$

where:

- $C_a$  = concentration of carbon-14 in benthic invertebrates, picocuries per gram wet tissue
- $C_w$  = concentration of carbon-14 in nearshore surface water, picocuries per liter
- 0.11 = fraction of the total animal mass that is natural carbon, grams carbon per gram wet tissue
- 0.014 = concentration of natural carbon in Columbia River nearshore surface water, grams carbon per liter water

The concentration of natural carbon in Columbia River nearshore surface water was calculated from median alkalinity (57 milligrams calcium carbonate per liter) and pH (7.8) values for the Columbia River (Poston et al. 2007), as well as equilibrium constants for the aqueous carbonate solution,  $pK_1 = 6.3$  and  $pK_2 = 10.25$  (Stumm and Morgan 1970).

Likewise, the concentration of tritium in benthic invertebrates was calculated assuming that the specific activity of tritium in tissue would be equal to the specific activity in Columbia River nearshore surface water, as follows:

$$C_a = C_w \times 0.8/1,000$$

where:

- $C_a$  = concentration of tritium in benthic invertebrates, picocuries per gram
- $C_w$  = concentration of tritium in nearshore surface water, picocuries per liter
- 0.8 = fraction of animal mass that is water
- 1,000 = grams water per liter

The concentrations of carbon-14 and tritium in fish would be equal to those of benthic invertebrates. The concentrations of carbon-14 and tritium in wildlife receptors would be equal to the concentrations in their animal prey.

#### **P.3.1.1.4.3 Exposure Doses from Chemicals**

For aquatic and riparian receptors exposed to chemicals by multiple pathways (direct contact, ingestion, respiration) resulting from living in sediment or surface water, exposure was not calculated. The assessment of impacts for these receptors was made by comparing estimated sediment, sediment pore water, or surface-water concentrations to appropriate benchmark concentrations for these receptors (see Section P.3.1.1.5). Exposure was estimated only for wildlife receptors exposed to chemical and radioactive COPCs via ingestion. Inhalation was not included because there would be little to no resuspension of sediment or riparian soil into air. The ingestion ADD for chemical COPCs was compared with benchmark doses to characterize risk.

The ingestion doses to aquatic wildlife receptors from chemical COPCs in surface water and sediment were calculated as the sum of doses from ingesting water, sediment, and food as follows:

$$ADD_{total} = ADD_{water} + ADD_{sediment} + ADD_{food}$$

where:

- $ADD_{total}$  = total dose of chemical from ingestion of water, animal food, and sediment, milligrams per kilogram body weight per day  
 $ADD_{water}$  = dose of chemical from ingestion of water, milligrams per kilogram body weight per day  
 $ADD_{sediment}$  = dose of chemical from ingestion of sediment, milligrams per kilogram body weight per day  
 $ADD_{food}$  = dose of chemical from ingestion of animal food, milligrams per kilogram body weight per day

and

$$ADD_{water} = C_w \times IR_w \times CF$$

where:

- $C_w$  = nearshore surface-water concentration, milligrams per liter  
 $IR_w$  = ingestion rate of water by the receptor, liters per kilogram body weight per day  
 $CF$  = unit conversion factor, 1 for chemical COPCs

and

$$ADD_{sediment} = C_{sed} \times IR_s = C_{sed} \times IR_f \times SF$$

where:

- $C_{sed}$  = concentration in sediment, milligrams per kilogram dry sediment  
 $IR_s$  = ingestion rate of sediment by the receptor, kilograms dry sediment per kilogram body weight per day  
 $IR_f$  = daily food ingestion rate, kilograms wet weight per kilogram body weight per day  
 $SF$  = sediment ingested as a fraction of food ingested, kilograms dry sediment per kilogram wet weight food

and

$$ADD_{food} = C_a \times IR_a = C_a \times IR_f \times AF$$

where:

- $C_a$  = concentration of chemical COPC in animal food, milligrams per kilogram wet food  
 $IR_a$  = ingestion rate of animal food by the receptor, kilograms wet food per kilogram body weight per day  
 $IR_f$  = daily food ingestion rate, kilograms wet weight per kilogram body weight per day  
 $AF$  = animal fraction of diet: prey

Spotted sandpipers and raccoons were assumed to eat benthic invertebrates living in nearshore sediment and exposed to nearshore sediment pore water. Bald eagles and least weasels were assumed to eat fish, such as salmonids, exposed to nearshore surface water.

The area use and temporal use factors were assumed to equal 1 for conservatism, so they did not appear in the exposure equations.

#### P.3.1.1.5 Toxicological Benchmarks

The benchmark for combined internal and external exposure from all radionuclides is 0.1 rad per day for the spotted sandpiper, raccoon, least weasel, and bald eagle (IAEA 1992) and 1 rad per day for aquatic biota and benthic invertebrates (NCRP 1991). Chemical benchmarks for aquatic biota, including

Woodhouse's toad larval forms and salmonids, were surface-water concentrations (milligrams per liter); TRVs for benthic invertebrates exposed to water and sediment were sediment concentrations (milligrams per kilogram); and TRVs for wildlife receptors potentially impacted by chemicals released to the Columbia River via air emissions were doses (milligrams per kilogram per day). All TRVs were chemical-specific literature values from a variety of published sources (e.g., Jones, Suter, and Hull 1997; Sample, Opresko, and Suter 1996; Suter and Tsao 1996).

#### **P.3.1.1.6 Risk Indices**

As discussed in Section P.2.1, the long-term impacts on ecological resources of potential radionuclide and chemical releases were evaluated by comparing estimates of exposure for a given ecological receptor for a given chemical or radioactive COPC under each alternative to threshold exposures associated with a known level of adverse effect of the COPC on that type of receptor. The estimate of chemical exposure concentration under each alternative for sediment-dwelling (benthic) invertebrates was the predicted sediment concentration; for aquatic biota, including salmonids, it was the predicted surface-water concentration (see Appendix G). The methods for estimating exposure doses for aquatic and riparian receptors from predicted air, water, and sediment concentrations were defined in Section P.3.1.1.4. The exposure concentrations or doses associated with a known level of adverse effect were the TRVs (see Section P.3.1.1.5). A comparison of these two values was made by calculating a risk index, the dimensionless ratio of the exposure estimate (concentration or dose) to corresponding TRV (concentration or dose). These calculated risk indices, i.e., the Hazard Quotients for individual chemical COPCs and the Hazard Indices for all radioactive COPCs combined, were used to compare the long-term impacts of the *TC & WM EIS* alternatives (see Chapter 5) and to identify exposures posing little or no risk (Hazard Quotient or Hazard Index less than or equal to unity [1]).

The risk indices were calculated as follows:

For benthic invertebrates exposed to chemical COPCs in sediments,

$$HQ = C_{\text{sed}} / TRV$$

where:

$HQ$  = Hazard Quotient  
 $C_{\text{sed}}$  = concentration in sediment, milligrams per kilogram dry sediment  
 $TRV$  = toxicity reference value, milligrams per kilogram

For aquatic biota, including salmonids exposed to chemical COPCs in surface water,

$$HQ = C_w / TRV$$

where:

$HQ$  = Hazard Quotient  
 $C_w$  = nearshore surface-water concentration, milligrams per liter  
 $TRV$  = toxicity reference value, milligrams per liter

For wildlife receptors exposed to chemical COPCs in air, sediment and surface water,

$$HQ = ADD_{\text{total}} / TRV$$

where:

- $HQ$  = Hazard Quotient
- $ADD_{total}$  = total dose of chemical from ingestion of water, animal food, and sediment, milligrams per kilogram body weight per day
- $TRV$  = toxicity reference and value, milligrams per kilogram body weight per day

and for all receptors, the Hazard Index is the sum of external and internal doses from all radioactive COPCs divided by the TRV; that is,

$$HI = (RD_{Ext} + RD_{Int}) / TRV$$

where:

- $HI$  = Hazard Index
- $RD_{Ext}$  = external radiation dose from exposure to radioactive COPCs in air, soil, sediment, and/or water, rad per day
- $RD_{Int}$  = internal radiation dose from radioactive COPCs, rad per day
- $TRV$  = toxicity reference value, rad per day

Except where an exposure parameter or TRV was not available for a given receptor or COPC, the dose ( $ADD_{total}$ ) and Hazard Quotient for all chemical COPCs and the dose ( $RD_{Ext} + RD_{Int}$ ) summed over all radioactive COPCs and the Hazard Index were calculated for all aquatic and riparian receptors that potentially would be exposed at the Columbia River under all *TC & WM EIS* alternatives using predicted air, surface-water, and sediment concentrations resulting from air releases during operations. Tables with predicted air, surface-water, and sediment concentrations; input parameters; and calculations of dose and risk indices are available in *Calculating Risk Indices for Long-Term Impacts to Ecological Receptors – Releases to Air* (SAIC 2011a).

Radiological and chemical hazards estimated for potential aquatic receptors and terrestrial wildlife feeding in the Columbia River due to exposure to contaminants released to the air and subsequently deposited in the Columbia River are summarized below using maximum Hazard Quotients and Hazard Indices. Hazards due to discharge from groundwater for aquatic receptors and terrestrial wildlife feeding in the Columbia River are discussed in Section P.3.2.

### P.3.1.2 Results and Discussion

The results of the screening analysis for radioactive contaminant releases to air and subsequent deposition estimated for aquatic receptors and terrestrial wildlife feeding in the Columbia River under the various Tank Closure, FFTF Decommissioning, and Waste Management alternatives, as well as the alternative combinations, are summarized in Tables P–9, P–10, and P–11.

The maximum combined radiological Hazard Index from emissions under all alternatives was calculated to be 0.00134 for the spotted sandpiper under Tank Closure Alternative 6A, Option Case. Table P–9 presents the maximum Hazard Indices associated with air emissions calculated to reach the Columbia River under all alternatives. Exposure to radioactive COPCs from air emissions under all alternatives would be below the 1-rad-per-day benchmark for benthic invertebrates and aquatic biota, including salmonids, and the 0.1-rad-per-day benchmark for terrestrial wildlife receptors (i.e., the spotted sandpiper, raccoon, bald eagle, and least weasel). Estimated hazards for the representative species indicate that no adverse effects are expected for aquatic receptors and terrestrial wildlife feeding in the Columbia River from exposure to radioactive COPCs from air emissions. Because the direct impacts of air exposure are expected to be small, any associated, potentially indirect impacts on the ecosystem would be correspondingly minor.

**Table P-9. Long-Term Impacts of Radioactive COPC Air Deposition on Aquatic and Riparian Resources at the Columbia River: Hazard Indices by Receptor and Alternative**

Alternative	Hazard Index by Receptor					
	Benthic Invertebrates	Spotted Sandpiper	Raccoon	Bald Eagle	Least Weasel	Aquatic Biota/Salmonids
<b>Tank Closure</b>						
1	2.86×10 <sup>-4</sup>	1.04×10 <sup>-4</sup>	4.99×10 <sup>-5</sup>	1.24×10 <sup>-7</sup>	3.17×10 <sup>-6</sup>	6.57×10 <sup>-7</sup>
2A	4.91×10 <sup>-4</sup>	9.33×10 <sup>-4</sup>	4.49×10 <sup>-4</sup>	2.33×10 <sup>-5</sup>	4.67×10 <sup>-5</sup>	8.36×10 <sup>-6</sup>
2B	2.10×10 <sup>-4</sup>	8.41×10 <sup>-4</sup>	4.16×10 <sup>-4</sup>	4.40×10 <sup>-5</sup>	6.50×10 <sup>-5</sup>	9.97×10 <sup>-6</sup>
3A	2.11×10 <sup>-4</sup>	8.90×10 <sup>-4</sup>	4.60×10 <sup>-4</sup>	8.31×10 <sup>-5</sup>	1.03×10 <sup>-4</sup>	1.37×10 <sup>-5</sup>
3B	1.98×10 <sup>-4</sup>	7.87×10 <sup>-4</sup>	3.79×10 <sup>-4</sup>	2.26×10 <sup>-5</sup>	4.28×10 <sup>-5</sup>	7.50×10 <sup>-6</sup>
3C	2.12×10 <sup>-4</sup>	8.99×10 <sup>-4</sup>	4.64×10 <sup>-4</sup>	8.31×10 <sup>-5</sup>	1.04×10 <sup>-4</sup>	1.38×10 <sup>-5</sup>
4	2.10×10 <sup>-4</sup>	8.50×10 <sup>-4</sup>	4.16×10 <sup>-4</sup>	3.75×10 <sup>-5</sup>	5.79×10 <sup>-5</sup>	9.19×10 <sup>-6</sup>
5	1.99×10 <sup>-4</sup>	8.35×10 <sup>-4</sup>	4.20×10 <sup>-4</sup>	5.70×10 <sup>-5</sup>	7.72×10 <sup>-5</sup>	1.10×10 <sup>-5</sup>
6A, Base Case	2.77×10 <sup>-4</sup>	1.19×10 <sup>-3</sup>	5.69×10 <sup>-4</sup>	1.74×10 <sup>-5</sup>	3.88×10 <sup>-5</sup>	8.69×10 <sup>-6</sup>
6A, Option Case	3.10×10 <sup>-4</sup>	<b>1.34×10<sup>-3</sup></b>	6.38×10 <sup>-4</sup>	1.76×10 <sup>-5</sup>	3.94×10 <sup>-5</sup>	9.32×10 <sup>-6</sup>
6B, Base Case	2.85×10 <sup>-4</sup>	1.25×10 <sup>-3</sup>	6.11×10 <sup>-4</sup>	4.47×10 <sup>-5</sup>	6.71×10 <sup>-5</sup>	1.17×10 <sup>-5</sup>
6B, Option Case	2.93×10 <sup>-4</sup>	1.27×10 <sup>-3</sup>	6.17×10 <sup>-4</sup>	4.48×10 <sup>-5</sup>	6.72×10 <sup>-5</sup>	1.18×10 <sup>-5</sup>
6C	2.06×10 <sup>-4</sup>	8.40×10 <sup>-4</sup>	4.15×10 <sup>-4</sup>	4.39×10 <sup>-5</sup>	6.49×10 <sup>-5</sup>	9.89×10 <sup>-6</sup>
<b>FFTF Decommissioning</b>						
1	0	0	0	0	0	0
2, Hanford Option	1.54×10 <sup>-7</sup>	1.09×10 <sup>-6</sup>	5.98×10 <sup>-7</sup>	2.42×10 <sup>-8</sup>	1.19×10 <sup>-7</sup>	9.60×10 <sup>-9</sup>
2, Idaho Option	1.40×10 <sup>-12</sup>	1.05×10 <sup>-11</sup>	5.02×10 <sup>-12</sup>	3.39×10 <sup>-13</sup>	9.70×10 <sup>-13</sup>	1.15×10 <sup>-13</sup>
3, Hanford Option	1.54×10 <sup>-7</sup>	1.09×10 <sup>-6</sup>	5.97×10 <sup>-7</sup>	2.42×10 <sup>-8</sup>	1.19×10 <sup>-7</sup>	9.60×10 <sup>-9</sup>
3, Idaho Option	0	0	0	0	0	0
<b>Waste Management</b>						
1	0	0	0	0	0	0
2, DG1	1.09×10 <sup>-11</sup>	8.34×10 <sup>-13</sup>	3.62×10 <sup>-13</sup>	9.10×10 <sup>-16</sup>	5.06×10 <sup>-15</sup>	5.33×10 <sup>-14</sup>
2, DG2	1.09×10 <sup>-11</sup>	8.34×10 <sup>-13</sup>	3.62×10 <sup>-13</sup>	9.10×10 <sup>-16</sup>	5.06×10 <sup>-15</sup>	5.33×10 <sup>-14</sup>
2, DG3	1.09×10 <sup>-11</sup>	8.34×10 <sup>-13</sup>	3.62×10 <sup>-13</sup>	9.10×10 <sup>-16</sup>	5.06×10 <sup>-15</sup>	5.33×10 <sup>-14</sup>
3, DG1	1.09×10 <sup>-11</sup>	8.34×10 <sup>-13</sup>	3.62×10 <sup>-13</sup>	9.10×10 <sup>-16</sup>	5.06×10 <sup>-15</sup>	5.33×10 <sup>-14</sup>
3, DG2	1.09×10 <sup>-11</sup>	8.34×10 <sup>-13</sup>	3.62×10 <sup>-13</sup>	9.10×10 <sup>-16</sup>	5.06×10 <sup>-15</sup>	5.33×10 <sup>-14</sup>
3, DG3	1.09×10 <sup>-11</sup>	8.34×10 <sup>-13</sup>	3.62×10 <sup>-13</sup>	9.10×10 <sup>-16</sup>	5.06×10 <sup>-15</sup>	5.33×10 <sup>-14</sup>
<b>Alternative Combination</b>						
1	2.86×10 <sup>-4</sup>	1.04×10 <sup>-4</sup>	4.99×10 <sup>-5</sup>	1.24×10 <sup>-7</sup>	3.17×10 <sup>-6</sup>	6.57×10 <sup>-7</sup>
2	2.11×10 <sup>-4</sup>	8.42×10 <sup>-4</sup>	4.16×10 <sup>-4</sup>	4.40×10 <sup>-5</sup>	6.51×10 <sup>-5</sup>	9.98×10 <sup>-6</sup>
3	2.85×10 <sup>-4</sup>	1.25×10 <sup>-3</sup>	6.11×10 <sup>-4</sup>	4.47×10 <sup>-5</sup>	6.72×10 <sup>-5</sup>	1.18×10 <sup>-5</sup>

**Note:** The maximum Hazard Index is indicated by **bold** text. Hazard Index is unitless.

**Key:** COPC=constituent of potential concern; DG=Disposal Group; FFTF=Fast Flux Test Facility.

No receptor exposed to chemical COPCs deposited in the Columbia River as a result of air emissions under the various Tank Closure, FFTF Decommissioning, and Waste Management alternatives had a screening Hazard Quotient exceeding 1 (see Table P-10). The highest Hazard Quotient was 0.508 for the spotted sandpiper exposed to mercury in nearshore surface water under Tank Closure Alternatives 3A and 3C. Hazard Quotients for such terrestrial mammals as the raccoon and least weasel, as well as piscivorous birds, which feed in the Columbia River on benthic invertebrates and salmonids, respectively,

did not exceed 0.1 (see Table P–11). Given the conservative exposure assumptions and toxicological benchmarks, ecological receptors in the Hanford Reach would be unlikely to be at unacceptable risk due to the deposition of chemical COPCs emitted to the air under any alternative.

**Table P–10. Long-Term Impacts of Chemical COPC Air Deposition on Aquatic and Riparian Resources at the Columbia River: Maximum Risk Index by Alternative**

Alternative	Maximum Hazard Quotient	Chemical COPC	Receptor
<b>Tank Closure</b>			
1	4.35×10 <sup>-2</sup>	Ammonia	Aquatic Biota/Salmonids
2A	3.90×10 <sup>-1</sup>	Mercury	Spotted sandpiper
2B	4.25×10 <sup>-1</sup>	Mercury	Spotted sandpiper
3A	<b>5.08×10<sup>-1</sup></b>	Mercury	Spotted sandpiper
3B	2.89×10 <sup>-1</sup>	Mercury	Spotted sandpiper
3C	<b>5.08×10<sup>-1</sup></b>	Mercury	Spotted sandpiper
4	3.66×10 <sup>-1</sup>	Mercury	Spotted sandpiper
5	3.50×10 <sup>-1</sup>	Mercury	Spotted sandpiper
6A, Base Case	3.93×10 <sup>-1</sup>	Mercury	Spotted sandpiper
6A, Option Case	3.92×10 <sup>-1</sup>	Mercury	Spotted sandpiper
6B, Base Case	4.41×10 <sup>-1</sup>	Mercury	Spotted sandpiper
6B, Option Case	4.40×10 <sup>-1</sup>	Mercury	Spotted sandpiper
6C	4.40×10 <sup>-1</sup>	Mercury	Spotted sandpiper
<b>FFTF Decommissioning</b>			
1	6.89×10 <sup>-2</sup>	Benzene	Aquatic Biota/Salmonids
2, Hanford Option	4.15×10 <sup>-2</sup>	Ammonia	Aquatic Biota/Salmonids
2, Idaho Option	9.33×10 <sup>-3</sup>	Benzene	Aquatic Biota/Salmonids
3, Hanford Option	4.10×10 <sup>-2</sup>	Ammonia	Aquatic Biota/Salmonids
3, Idaho Option	4.82×10 <sup>-3</sup>	Benzene	Aquatic Biota/Salmonids
<b>Waste Management</b>			
1	6.97×10 <sup>-3</sup>	Benzene	Aquatic Biota/Salmonids
2, DG1	1.24×10 <sup>-1</sup>	Benzene	Aquatic Biota/Salmonids
2, DG2	4.01×10 <sup>-1</sup>	Benzene	Aquatic Biota/Salmonids
2, DG3	4.01×10 <sup>-1</sup>	Benzene	Aquatic Biota/Salmonids
3, DG1	1.20×10 <sup>-1</sup>	Benzene	Aquatic Biota/Salmonids
3, DG2	3.96×10 <sup>-1</sup>	Benzene	Aquatic Biota/Salmonids
3, DG3	3.96×10 <sup>-1</sup>	Benzene	Aquatic Biota/Salmonids
<b>Alternative Combination</b>			
1	8.51×10 <sup>-2</sup>	Benzene	Aquatic Biota/Salmonids
2	4.25×10 <sup>-1</sup>	Mercury	Spotted sandpiper
3	4.61×10 <sup>-1</sup>	Benzene	Aquatic Biota/Salmonids

**Note:** The maximum Hazard Quotient of all receptors is indicated by **bold** text. Risk indices are unitless.

**Key:** COPC=constituent of potential concern; DG=Disposal Group; FFTF=Fast Flux Test Facility.

**Table P–11. Long-Term Impacts of Chemical COPC Air Deposition on Aquatic and Riparian Resources at the Columbia River: Maximum Risk Index by Receptor**

Receptor	Alternative	Maximum Hazard Quotient	Chemical COPC
Benthic invertebrates	Tank Closure Alternative 2A	$6.83 \times 10^{-2}$	Ammonia
Aquatic biota/salmonids	Alternative Combination 3	$4.61 \times 10^{-1}$	Benzene
Spotted sandpiper	Tank Closure Alternatives 3A, 3C	$5.08 \times 10^{-1}$	Mercury
Raccoon	Tank Closure Alternatives 3A, 3C	$4.31 \times 10^{-2}$	Mercury
Least weasel	Tank Closure Alternative 6B, Base Case Alternative Combination 3	$2.38 \times 10^{-2}$	Mercury
Bald eagle	Tank Closure Alternative 6B, Base Case Alternative Combination 3	$4.16 \times 10^{-2}$	Mercury

**Note:** Risk indices are unitless.

**Key:** COPC=constituent of potential concern.

As was the case for Hanford soils, the buffering capacity of the Hanford Reach would be sufficient to maintain the pH within the National Ambient Water Quality Criteria acceptable range for aquatic life (pH = 5.0–9.0) and Washington Ambient Surface-water Quality Standards for the Hanford Reach (pH = 6.5–8.5) despite deposition of nitrogen and sulfur dioxides from air emissions under the various Tank Closure, FFTF Decommissioning, and Waste Management alternatives. Two weak acids (sulfurous acid and nitrous acid) and a strong acid (nitric acid) potentially result from the dissolution of nitrogen and sulfur dioxides in river water. According to the *Hanford Site Environmental Report for Calendar Year 2010 (Including Some Early 2011 Information)* (Poston, Duncan, and Dirkes 2011), the Hanford Reach has a reported alkalinity ranging from 52 to 64 milligrams calcium carbonate per liter and a pH between 7.4 and 8.3. An alkalinity of 52 milligrams calcium carbonate per liter would keep the pH at or above 7.4, given the addition of 0.041 milligrams nitrogen dioxide per liter and 0.00016 milligrams sulfur dioxide per liter (Alternative Combination 3), the maximum predicted nearshore surface-water concentrations. The resulting pH would not fall outside the permissible range of pH for the Hanford Reach (6.5–8.5), and the estimated change in the pH would not exceed the maximum allowable 0.5 induced variation limit (Poston, Duncan, and Dirkes 2011). The pH of the Hanford Reach is thus potentially lowered only slightly by the deposition of nitrogen and sulfur dioxides released into the air under all *TC & WM EIS* alternatives, and aquatic biota are unlikely to be adversely impacted by pH changes.

### **P.3.1.3      Uncertainties**

Uncertainty exists about the actual magnitude of future exposures and the threshold doses or benchmark concentration TRVs used to evaluate the long-term impact on aquatic and riparian ecological resources of air releases. The uncertainties regarding the chemical and radiological exposure estimates come from uncertainties in the source terms and transport models. Additional uncertainties are found in the BAFs and uptake factors, which are linear models based on simplifying assumptions. The uncertainties for toxicity and radiological effects thresholds arise from extrapolations from laboratory experiments on test species to Hanford receptor species in natural environments, as well as uncertainty about the chemical to which ecological receptors would be exposed (e.g., chemical COPC breakdown products, which can be more toxic than the COPC itself). The lack of TRVs for some chemical COPCs and some receptors also resulted in uncertainties. Combined, these uncertainties produced limited underestimates of risk and moderate overestimates of risk for different combinations of receptors and chemical or radioactive COPCs. The effects of these uncertainties were unbiased with respect to the alternatives being evaluated in this *TC & WM EIS*; thus, the results presented above accurately reflect the relative impacts of the alternatives on ecological resources. In addition, the use of conservative exposure assumptions and TRVs mitigated these uncertainties, providing confidence in the “no risk” conclusions.

### **P.3.2 Impacts of Groundwater Releases**

The potential for adverse effects on Columbia River aquatic and riparian resources from potential releases of radionuclides and chemicals to groundwater under the different Tank Closure, FTF Decommissioning, and Waste Management alternatives was evaluated using a quantitative risk assessment approach (63 FR 26846; EPA 1992, 1997). Groundwater contamination in the distant future (up to 10,000 years) would be possible under all alternatives because some waste would be generated and disposed of on site or contaminated soil would be left in place under all alternatives. Radionuclides and chemicals potentially would be transported to the Columbia River and its riparian habitat. The potential for adverse impacts on aquatic and riparian resources at the Columbia River is described below.

#### **P.3.2.1 Methods**

The general approach for assessing potential adverse effects on aquatic and riparian ecological resources was discussed in Section P.2.1. The assumptions, receptors, exposure pathways and uptake mechanisms (routes); predicted sediment and surface-water concentrations; exposure model equations; and benchmarks used to model exposure for aquatic and riparian ecological resources potentially impacted by contaminant releases are described in the relevant sections below. The quantitative evaluations of long-term adverse impacts on aquatic and riparian resources of groundwater releases, based on Hazard Quotients, Hazard Indices, and river-water nitrate concentrations, are summarized and discussed in Section P.3.2.2. The impact of nitrate discharge on the eutrophication of surface water was evaluated based on ambient and predicted concentrations.

##### **P.3.2.1.1 Key Assumptions**

The following key assumptions were made in the evaluation of potential impacts on Columbia River aquatic and riparian resources of exposure to radionuclides and chemicals through groundwater releases:

- Exposure of riparian vegetation and soil-dwelling biota to seep water was inconsequential because groundwater discharges at discrete points along the shore and either discharges underwater or flows only a short distance—less than 5 meters (16.4 feet)—through the riparian zone before entering the river.
- Concentrations in groundwater at the Columbia River overestimated concentrations of seep and sediment pore water because Columbia River water mixes with those waters to varying degrees.
- Groundwater flux was assumed to be approximately 1 cubic meter (35.3 cubic feet, or 264 gallons) per second because the river flux is approximately 3,000 times greater than the flux from groundwater, and the flux of the Columbia River is approximately 3,300 cubic meters (116,540 cubic feet, or 871,761 gallons) per second (Bryce et al. 2002).
- The tissue concentrations in fish preyed upon by predators (least weasel and bald eagle) would be in equilibrium with nearshore surface-water concentrations.
- Surface-water and sediment contamination from groundwater releases would not coincide with soil contamination from air releases because material released to air during site and WTP operations would dissipate before slow-moving constituents discharge to the Columbia River.

##### **P.3.2.1.2 Receptors and Exposure Pathways and Routes**

The receptors selected to represent the Columbia River aquatic and riparian ecological resources potentially exposed to groundwater releases, including special status species (Chapter 3, Section 3.2.7.4), are listed in Table P-2. These receptors were selected because they are expected to have higher

exposures than those not selected from their group due to their higher ingestion rates per unit body weight for prey, water, and sediment or soil. Special status species were not expected to be more highly exposed or more sensitive to contaminants than the selected species. The selected representative receptors were benthic invertebrates; muskrat; spotted sandpiper; raccoon; bald eagle; least weasel; and aquatic biota, including salmonids. All were ECEM receptors, except the spotted sandpiper, which was substituted for the common snipe because the spotted sandpiper has a more aquatic diet. The muskrat was added as a receptor exposed primarily to groundwater discharging at seeps along the river because of its relatively high water-ingestion rate and small size compared with other mammals such as the mule deer or coyote. For this evaluation, the muskrat was assumed to be exposed by ingestion of only seep water to assess the importance of this pathway.

The exposure pathways evaluated in the ecological risk analysis for this *TC & WM EIS* are shown in Table P-2 for all ecological receptors. The exposure medium, exposure route, and receptor are indicated for each pathway evaluated in the analysis of impacts on aquatic and riparian resources of releases to groundwater.

#### **P.3.2.1.3 Predicted Seep, Sediment, and Surface-Water Concentrations**

Tank Closure Alternatives 1 through 6C; FFTF Decommissioning Alternatives 1 and 2; and Waste Management Alternatives 1, 2, and 3 (Disposal Groups 1, 2, and 3) have groundwater modeling results for 10,000 years. Separate groundwater modeling results do not exist for FFTF Decommissioning Alternative 3 because it would not result in a release to groundwater. The previously mentioned alternatives would potentially impact seep, sediment pore water, sediment, and surface water. The concentrations were calculated from the modeled groundwater concentrations at the Columbia River resulting from the varying radioactive and chemical COPC inventories in place under the different alternatives (see Appendix O).

Seep and sediment pore-water concentrations were equal to the modeled peak annual average groundwater concentration at the Columbia River. Seep concentrations were used to assess potential impacts on wildlife receptors drinking water in the riparian zone. Peak annual average nearshore surface-water concentrations were used to estimate adverse impacts on aquatic biota (e.g., periphyton, plankton, larval mayflies, juvenile salmonids, and lower-trophic-level fish). Sediment concentrations for nonpolar hydrophobic organic compounds were calculated assuming equilibrium partitioning between sediment and sediment pore water. Sediment and sediment pore-water concentrations were used to assess potential impacts on sediment-dwelling biota and their predators. Nearshore surface-water concentrations used to estimate body burdens in fish (e.g., salmonids) and dose to predators of fish were calculated assuming that the groundwater would be mixed throughout a 0.5-meter-deep (1.6-foot-deep), 40-meter-wide (131-foot-wide) shallow zone along the facility side of the river. With a reported maximum velocity of 0.25 meters (0.8 feet) per second in the nearshore environment of redds (USGS 2000), the nearshore flux was estimated as 5 cubic meters (177 cubic feet, or 1,321 gallons) per second. The flux of groundwater into the river over this reach was one three-thousandth of the flux of the Columbia River in the Hanford Reach, approximately 1 cubic meter (35.3 cubic feet, or 264 gallons) per second (Bryce et al. 2002). The groundwater (seep and sediment pore water), sediment, and nearshore surface-water concentrations under Tank Closure Alternatives 1 through 6C; FFTF Decommissioning Alternatives 1 and 2 (Hanford and Idaho Options); and Waste Management Alternatives 1, 2, and 3 (Disposal Groups 1, 2, and 3) were used as the source terms in the exposure model described in the following subsections.

#### **P.3.2.1.4 Exposure Model Calculations**

The exposure model calculated ingestion doses from chemicals for wildlife receptors, as well as external and internal doses from radionuclides for all receptors, using the equations for  $RD_{\text{Ext-water, imm}}$ ,

$RD_{\text{Ext-water near}}$ ,  $RD_{\text{Ext-sed}}$ , and  $RD_{\text{Int}}$  presented in Section P.3.1.1. There was no external dose to receptors from air for radionuclides released to the groundwater and discharged to the Columbia River.

Exposure was not calculated for aquatic and riparian receptors exposed to chemicals by multiple pathways (direct contact, ingestion, respiration) resulting from living in sediment or surface water. The assessment of impacts on aquatic and sediment-dwelling biota was made by comparing estimated sediment or nearshore surface-water concentrations to appropriate benchmark concentrations for these receptors (see Section P.3.2.1.5).

### P.3.2.1.5 Toxicological Benchmarks

The benchmark for combined internal and external exposure from all radionuclides was 0.1 rad per day for the muskrat (IAEA 1992). Radiological and chemical benchmarks for the other receptors were the same as those in Section P.3.1.1.5.

### P.3.2.1.6 Risk Indices

As discussed in Section P.2.1, the long-term impacts of potential radionuclide and chemical releases on ecological resources were evaluated by comparing estimated ecological receptor exposures to given chemical or radioactive COPCs under each alternative to the threshold COPC exposures associated with known adverse effects on those receptors. The estimate of chemical exposure concentration under each alternative for sediment-dwelling (benthic) invertebrates was the predicted sediment concentration; for aquatic biota, including salmonids, it was the predicted nearshore surface-water concentration (see Appendix O). The methods for estimating exposure doses to aquatic and riparian receptors from the predicted groundwater concentrations and discharge at the Columbia River were defined in Section P.3.1.1.4. The exposure concentrations or doses associated with a known level of adverse effect were the TRVs (see Section P.3.1.1.5). A comparison of the estimated and threshold COPC exposures was made by calculating a risk index, the dimensionless ratio of the exposure estimate (concentration or dose) to corresponding TRV (concentration or dose). These calculated risk indices, i.e., the Hazard Quotients for individual chemical COPCs and the Hazard Indices for all radioactive COPCs combined, were used to compare the *TC & WM EIS* alternatives (see Chapter 5) and to identify exposures posing little or no risk (Hazard Quotient or Hazard Index less than or equal to unity [1]).

The risk indices were calculated as follows:

For benthic invertebrates exposed to chemical COPCs in sediment:

$$HQ = C_{\text{sed}} / TRV$$

where:

$HQ$	=	Hazard Quotient
$C_{\text{sed}}$	=	concentration in sediment, milligrams per kilogram dry sediment
$TRV$	=	toxicity reference value, milligrams per kilogram

For aquatic biota, including salmonids, exposed to chemical COPCS in nearshore surface water:

$$HQ = C_w / TRV$$

where:

$HQ$	=	Hazard Quotient
$C_w$	=	nearshore surface-water concentration, milligrams per liter
$TRV$	=	toxicity reference value, milligrams per liter

For wildlife receptors exposed to chemical COPCs in groundwater, sediment, and nearshore surface water:

$$HQ = ADD_{\text{total}} / TRV$$

where:

- $HQ$  = Hazard Quotient
- $ADD_{\text{total}}$  = total dose of chemical from ingestion of water, animal food, and sediment, milligrams per kilogram body weight per day
- $TRV$  = toxicity reference value, milligrams per kilogram body weight per day

For all receptors, the Hazard Index is the sum of external and internal doses from all radioactive COPCs divided by the TRV, as follows:

$$HI = (RD_{\text{Ext}} + RD_{\text{Int}}) / TRV$$

where:

- $HI$  = Hazard Index
- $RD_{\text{Ext}}$  = external radiation dose from exposure to all radioactive COPCs in air, soil, sediment, and/or water, rad per day
- $RD_{\text{Int}}$  = internal radiation dose from all radioactive COPCs, rad per day
- $TRV$  = toxicity reference value, rad per day

Except where an exposure parameter or TRV was not available for a given receptor or COPC, the dose ( $ADD_{\text{total}}$ ) and Hazard Quotient for all chemical COPCs and the dose ( $RD_{\text{Ext}} + RD_{\text{Int}}$ ) summed over all radioactive COPCs and the Hazard Index were calculated for all aquatic and riparian receptors potentially exposed at the Columbia River under all *TC & WM EIS* alternatives using predicted groundwater, seep, nearshore surface-water, and sediment concentrations resulting from releases to groundwater. Tables with predicted surface-water and sediment concentrations, input parameters, and calculations of dose and risk indices are available in *Calculating Risk Indices for Long-Term Impacts to Ecological Receptors – Releases to Groundwater* (SAIC 2011b).

### **P.3.2.2 Results and Discussion**

The results of the screening analysis for radioactive and chemical contaminant releases to groundwater due to site and WTP operations and subsequent discharge to the Columbia River estimated for aquatic receptors and riparian wildlife feeding in the Columbia River under the various Tank Closure, FFTF Decommissioning, and Waste Management alternatives are summarized in Tables P-12, P-13, and P-14.

**Table P–12. Long-Term Impacts of Radioactive COPC Groundwater Discharge on Aquatic and Riparian Resources at the Columbia River: Hazard Indices by Receptor and Alternative**

Alternative	Hazard Index by Receptor						
	Benthic Invertebrates	Muskrat	Spotted Sandpiper	Raccoon	Bald Eagle	Least Weasel	Aquatic Biota/Salmonids
<b>Tank Closure</b>							
1	$2.03 \times 10^{-3}$	$4.11 \times 10^{-5}$	$6.76 \times 10^{-4}$	$3.08 \times 10^{-4}$	$5.51 \times 10^{-4}$	$1.56 \times 10^{-3}$	$2.81 \times 10^{-4}$
2A	$9.53 \times 10^{-4}$	$3.38 \times 10^{-5}$	$3.33 \times 10^{-4}$	$1.60 \times 10^{-4}$	$2.22 \times 10^{-4}$	$6.29 \times 10^{-4}$	$1.11 \times 10^{-4}$
2B, 3A, 3B, 3C, 6C	$4.43 \times 10^{-4}$	$3.38 \times 10^{-5}$	$2.07 \times 10^{-4}$	$1.05 \times 10^{-4}$	$2.23 \times 10^{-4}$	$6.31 \times 10^{-4}$	$1.09 \times 10^{-4}$
4	$4.28 \times 10^{-4}$	$3.38 \times 10^{-5}$	$2.04 \times 10^{-4}$	$1.04 \times 10^{-4}$	$2.23 \times 10^{-4}$	$6.31 \times 10^{-4}$	$1.09 \times 10^{-4}$
5	$4.35 \times 10^{-4}$	$3.38 \times 10^{-5}$	$2.05 \times 10^{-4}$	$1.05 \times 10^{-4}$	$2.23 \times 10^{-4}$	$6.31 \times 10^{-4}$	$1.09 \times 10^{-4}$
6A, Base Case	$2.72 \times 10^{-4}$	$3.38 \times 10^{-5}$	$1.66 \times 10^{-4}$	$8.75 \times 10^{-5}$	$2.23 \times 10^{-4}$	$6.30 \times 10^{-4}$	$1.08 \times 10^{-4}$
6A, Option Case	$9.92 \times 10^{-6}$	$3.41 \times 10^{-5}$	$1.04 \times 10^{-4}$	$6.09 \times 10^{-5}$	$2.23 \times 10^{-4}$	$6.31 \times 10^{-4}$	$1.07 \times 10^{-4}$
6B, Base Case	$2.72 \times 10^{-4}$	$3.38 \times 10^{-5}$	$1.66 \times 10^{-4}$	$8.75 \times 10^{-5}$	$2.23 \times 10^{-4}$	$6.30 \times 10^{-4}$	$1.08 \times 10^{-4}$
6B, Option Case	$1.01 \times 10^{-5}$	$3.45 \times 10^{-5}$	$1.06 \times 10^{-4}$	$6.21 \times 10^{-5}$	$2.23 \times 10^{-4}$	$6.30 \times 10^{-4}$	$1.07 \times 10^{-4}$
<b>FFTF Decommissioning</b>							
1	$2.20 \times 10^{-7}$	$2.80 \times 10^{-7}$	$2.77 \times 10^{-6}$	$1.23 \times 10^{-6}$	$2.36 \times 10^{-6}$	$6.76 \times 10^{-6}$	$1.11 \times 10^{-6}$
2	$2.32 \times 10^{-7}$	$2.94 \times 10^{-7}$	$2.92 \times 10^{-6}$	$1.30 \times 10^{-6}$	$2.36 \times 10^{-6}$	$6.75 \times 10^{-6}$	$1.10 \times 10^{-6}$
3	$8.78 \times 10^{-14}$	$1.11 \times 10^{-13}$	$1.10 \times 10^{-12}$	$4.91 \times 10^{-13}$	$5.48 \times 10^{-13}$	$1.58 \times 10^{-12}$	$2.48 \times 10^{-13}$
<b>Waste Management</b>							
1	$8.74 \times 10^{-9}$	$1.07 \times 10^{-8}$	$1.08 \times 10^{-7}$	$4.81 \times 10^{-8}$	$4.47 \times 10^{-8}$	$1.29 \times 10^{-7}$	$1.99 \times 10^{-8}$
2, DG1, SG1-A	$2.78 \times 10^{-6}$	$3.32 \times 10^{-6}$	$3.39 \times 10^{-5}$	$1.51 \times 10^{-5}$	$3.05 \times 10^{-5}$	$8.72 \times 10^{-5}$	$1.43 \times 10^{-5}$
2, DG1, SG1-B	$4.37 \times 10^{-6}$	$5.35 \times 10^{-6}$	$5.39 \times 10^{-5}$	$2.40 \times 10^{-5}$	$5.30 \times 10^{-5}$	$1.51 \times 10^{-4}$	$2.50 \times 10^{-5}$
2, DG1, SG1-C	$6.41 \times 10^{-6}$	$7.95 \times 10^{-6}$	$7.97 \times 10^{-5}$	$3.54 \times 10^{-5}$	$6.52 \times 10^{-5}$	$1.87 \times 10^{-4}$	$3.05 \times 10^{-5}$
2, DG1, SG1-D	$3.54 \times 10^{-6}$	$4.28 \times 10^{-6}$	$4.34 \times 10^{-5}$	$1.93 \times 10^{-5}$	$3.93 \times 10^{-5}$	$1.12 \times 10^{-4}$	$1.85 \times 10^{-5}$
2, DG1, SG1-E	$8.25 \times 10^{-6}$	$1.03 \times 10^{-5}$	$1.03 \times 10^{-4}$	$4.57 \times 10^{-5}$	$7.95 \times 10^{-5}$	$2.28 \times 10^{-4}$	$3.70 \times 10^{-5}$
2, DG1, SG1-F	$4.03 \times 10^{-6}$	$4.91 \times 10^{-6}$	$4.96 \times 10^{-5}$	$2.21 \times 10^{-5}$	$4.90 \times 10^{-5}$	$1.40 \times 10^{-4}$	$2.31 \times 10^{-5}$
2, DG1, SG1-G	$2.79 \times 10^{-6}$	$3.34 \times 10^{-6}$	$3.41 \times 10^{-5}$	$1.52 \times 10^{-5}$	$3.04 \times 10^{-5}$	$8.68 \times 10^{-5}$	$1.43 \times 10^{-5}$
2, DG2, SG2-A	$2.72 \times 10^{-6}$	$3.28 \times 10^{-6}$	$3.33 \times 10^{-5}$	$1.48 \times 10^{-5}$	$3.05 \times 10^{-5}$	$8.71 \times 10^{-5}$	$1.43 \times 10^{-5}$
2, DG2, SG2-B, Base Case	$2.75 \times 10^{-6}$	$3.32 \times 10^{-6}$	$3.37 \times 10^{-5}$	$1.50 \times 10^{-5}$	$3.00 \times 10^{-5}$	$8.59 \times 10^{-5}$	$1.41 \times 10^{-5}$
2, DG2, SG2-B, Option Case	$2.77 \times 10^{-6}$	$3.33 \times 10^{-6}$	$3.39 \times 10^{-5}$	$1.51 \times 10^{-5}$	$3.03 \times 10^{-5}$	$8.67 \times 10^{-5}$	$1.42 \times 10^{-5}$
2, DG3, Base Case	$2.70 \times 10^{-6}$	$3.25 \times 10^{-6}$	$3.31 \times 10^{-5}$	$1.47 \times 10^{-5}$	$3.13 \times 10^{-5}$	$8.94 \times 10^{-5}$	$1.47 \times 10^{-5}$
2, DG3, Option Case	$2.72 \times 10^{-6}$	$3.28 \times 10^{-6}$	$3.33 \times 10^{-5}$	$1.48 \times 10^{-5}$	$3.16 \times 10^{-5}$	$9.03 \times 10^{-5}$	$1.49 \times 10^{-5}$
3, DG1, SG1-A	$1.22 \times 10^{-5}$	$1.47 \times 10^{-5}$	$1.49 \times 10^{-4}$	$6.63 \times 10^{-5}$	$6.56 \times 10^{-5}$	$1.89 \times 10^{-4}$	$2.94 \times 10^{-5}$
3, DG1, SG1-B	$1.22 \times 10^{-5}$	$1.47 \times 10^{-5}$	$1.49 \times 10^{-4}$	$6.63 \times 10^{-5}$	$6.56 \times 10^{-5}$	$1.89 \times 10^{-4}$	$2.94 \times 10^{-5}$
3, DG1, SG1-C	$1.22 \times 10^{-5}$	$1.47 \times 10^{-5}$	$1.49 \times 10^{-4}$	$6.63 \times 10^{-5}$	$6.56 \times 10^{-5}$	$1.89 \times 10^{-4}$	$2.94 \times 10^{-5}$
3, DG1, SG1-D	$1.22 \times 10^{-5}$	$1.47 \times 10^{-5}$	$1.49 \times 10^{-4}$	$6.63 \times 10^{-5}$	$6.56 \times 10^{-5}$	$1.89 \times 10^{-4}$	$2.94 \times 10^{-5}$
3, DG1, SG1-E	$1.22 \times 10^{-5}$	$1.47 \times 10^{-5}$	$1.49 \times 10^{-4}$	$6.63 \times 10^{-5}$	$6.92 \times 10^{-5}$	$1.99 \times 10^{-4}$	$3.11 \times 10^{-5}$
3, DG1, SG1-F	$1.22 \times 10^{-5}$	$1.47 \times 10^{-5}$	$1.49 \times 10^{-4}$	$6.63 \times 10^{-5}$	$6.51 \times 10^{-5}$	$1.88 \times 10^{-4}$	$2.92 \times 10^{-5}$
3, DG1, SG1-G	$1.22 \times 10^{-5}$	$1.47 \times 10^{-5}$	$1.49 \times 10^{-4}$	$6.63 \times 10^{-5}$	$6.56 \times 10^{-5}$	$1.89 \times 10^{-4}$	$2.94 \times 10^{-5}$
3, DG2, SG2-A	$1.22 \times 10^{-5}$	$1.47 \times 10^{-5}$	$1.49 \times 10^{-4}$	$6.63 \times 10^{-5}$	$6.51 \times 10^{-5}$	$1.88 \times 10^{-4}$	$2.92 \times 10^{-5}$
3, DG2, SG2-B, Base Case	$1.22 \times 10^{-5}$	$1.47 \times 10^{-5}$	$1.49 \times 10^{-4}$	$6.64 \times 10^{-5}$	$7.15 \times 10^{-5}$	$2.06 \times 10^{-4}$	$3.22 \times 10^{-5}$

**Table P-12. Long-Term Impacts of Radioactive COPC Groundwater Discharge on Aquatic and Riparian Resources at the Columbia River: Hazard Indices by Receptor and Alternative (continued)**

Alternative	Hazard Index by Receptor						
	Benthic Invertebrates	Muskrat	Spotted Sandpiper	Raccoon	Bald Eagle	Least Weasel	Aquatic Biota/Salmonids
<b>Waste Management (continued)</b>							
3, DG2, SG2-B, Option Case	1.22×10 <sup>-5</sup>	1.47×10 <sup>-5</sup>	1.49×10 <sup>-4</sup>	6.63×10 <sup>-5</sup>	7.54×10 <sup>-5</sup>	2.17×10 <sup>-4</sup>	3.42×10 <sup>-5</sup>
3, DG3, Base Case	1.22×10 <sup>-5</sup>	1.47×10 <sup>-5</sup>	1.49×10 <sup>-4</sup>	6.63×10 <sup>-5</sup>	7.13×10 <sup>-5</sup>	2.05×10 <sup>-4</sup>	3.22×10 <sup>-5</sup>
3, DG3, Option Case	1.22×10 <sup>-5</sup>	1.47×10 <sup>-5</sup>	1.49×10 <sup>-4</sup>	6.63×10 <sup>-5</sup>	7.52×10 <sup>-5</sup>	2.16×10 <sup>-4</sup>	3.41×10 <sup>-5</sup>
<b>Alternative Combination</b>							
1	<b>2.03×10<sup>-3</sup></b>	4.11×10 <sup>-5</sup>	6.76×10 <sup>-4</sup>	3.08×10 <sup>-4</sup>	5.52×10 <sup>-4</sup>	1.56×10 <sup>-3</sup>	2.81×10 <sup>-4</sup>
2	4.43×10 <sup>-4</sup>	3.38×10 <sup>-5</sup>	2.07×10 <sup>-4</sup>	1.05×10 <sup>-4</sup>	2.23×10 <sup>-4</sup>	6.31×10 <sup>-4</sup>	1.09×10 <sup>-4</sup>
3	2.72×10 <sup>-4</sup>	3.38×10 <sup>-5</sup>	1.66×10 <sup>-4</sup>	8.75×10 <sup>-5</sup>	2.23×10 <sup>-4</sup>	6.30×10 <sup>-4</sup>	1.08×10 <sup>-4</sup>

**Note:** The maximum Hazard Index is indicated by **bold** text. Hazard Index is unitless.

**Key:** COPC=constituent of potential concern; DG=Disposal Group; FFTF=Fast Flux Test Facility; SG=Subgroup.

**Table P-13. Long-Term Impacts of Radioactive and Chemical COPC Groundwater Discharge on Aquatic and Riparian Resources at the Columbia River: Maximum Risk Index by Alternative**

Alternative	Maximum Hazard Quotient or Hazard Index	Chemical or Radioactive COPC	Receptor
<b>Tank Closure</b>			
1	4.32×10 <sup>1</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2A	4.31×10 <sup>1</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2B, 3A, 3B, 3C, 6C	4.31×10 <sup>1</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
4	4.31×10 <sup>1</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
5	4.31×10 <sup>1</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
6A, Base Case	4.31×10 <sup>1</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
6A, Option Case	4.44×10 <sup>1</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
6B, Base Case	4.31×10 <sup>1</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
6B, Option Case	<b>4.45×10<sup>1</sup></b>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
<b>FFTF Decommissioning</b>			
1	2.91×10 <sup>-2</sup>	Uranium	Raccoon
2	6.75×10 <sup>-6</sup>	All radionuclides	Least Weasel
3	1.58×10 <sup>-12</sup>	All radionuclides	Least Weasel
<b>Waste Management</b>			
1	3.14×10 <sup>-3</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2, DG1, SG1-A	2.05×10 <sup>-2</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2, DG1, SG1-B	1.66×10 <sup>-2</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2, DG1, SG1-C	2.90	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2, DG1, SG1-D	1.83×10 <sup>-1</sup>	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2, DG1, SG1-E	1.63	Chromium <sup>a</sup>	Aquatic Biota/Salmonids

**Table P-13. Long-Term Impacts of Radioactive and Chemical COPC Groundwater Discharge on Aquatic and Riparian Resources at the Columbia River: Maximum Risk Index by Alternative (continued)**

Alternative	Maximum Hazard Quotient or Hazard Index	Chemical or Radioactive COPC	Receptor
<b>Waste Management (continued)</b>			
2, DG1, SG1-F	2.55	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2, DG1, SG1-G	$2.07 \times 10^{-2}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2, DG2, SG2-A	$2.04 \times 10^{-2}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2, DG2, SG2-B, Base Case	$1.03 \times 10^{-1}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2, DG2, SG2-B, Option Case	$9.72 \times 10^{-1}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2, DG3, Base Case	$1.04 \times 10^{-1}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2, DG3, Option Case	$9.60 \times 10^{-1}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG1, SG1-A	$2.25 \times 10^{-2}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG1, SG1-B	$2.25 \times 10^{-2}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG1, SG1-C	2.90	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG1, SG1-D	$1.83 \times 10^{-1}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG1, SG1-E	1.63	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG1, SG1-F	2.54	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG1, SG1-G	$2.25 \times 10^{-2}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG2, SG2-A	$1.85 \times 10^{-2}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG2, SG2-B, Base Case	$1.09 \times 10^{-1}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG2, SG2-B, Option Case	$9.77 \times 10^{-1}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG3, Base Case	$1.10 \times 10^{-1}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3, DG3, Option Case	$9.66 \times 10^{-1}$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
<b>Alternative Combination</b>			
1	$4.32 \times 10^1$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
2	$4.31 \times 10^1$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids
3	$4.31 \times 10^1$	Chromium <sup>a</sup>	Aquatic Biota/Salmonids

<sup>a</sup> For purposes of long-term impacts, it was assumed that this is hexavalent chromium.

**Note:** The maximum Hazard Quotient or Hazard Index is indicated by **bold** text. Hazard Quotient and Hazard Index are unitless.

**Key:** COPC=constituent of potential concern; DG=Disposal Group; FFTF=Fast Flux Test Facility; SG=Subgroup.

**Table P-14. Long-Term Impacts of Chemical COPC Groundwater Discharge on  
Aquatic and Riparian Resources at the Columbia River:  
Maximum Risk Index by Receptor**

Receptor	Alternative	Maximum Hazard Quotient	Chemical COPC
Benthic invertebrates	Tank Closure Alternative 1 Alternative Combination 1	$1.69 \times 10^{-1}$	Chromium <sup>a</sup>
Aquatic biota/Salmonids	Tank Closure Alternative 6B, Option Case	$4.45 \times 10^1$	Chromium <sup>a</sup>
Muskrat	Tank Closure Alternatives 2B; 3A; 3B; 3C; 6C; 4; 5; 6A, Base Case; 6B, Base Case Alternative Combinations 2, 3	$1.43 \times 10^{-2}$	Nitrate
Spotted sandpiper	Tank Closure Alternative 1 Alternative Combination 1	1.15	Chromium <sup>a</sup>
Raccoon	Tank Closure Alternative 1 Alternative Combination 1	$1.39 \times 10^{-1}$	Chromium <sup>a</sup>
Least weasel	Tank Closure Alternatives 2B, 3A, 3B, 3C, 6C Alternative Combination 2	1.37	Nitrate
Bald eagle	Tank Closure Alternative 1 Alternative Combination 1	$3.71 \times 10^{-2}$	Chromium <sup>a</sup>

<sup>a</sup> For purposes of long-term impacts, it was assumed that this is hexavalent chromium.

**Key:** COPC=constituent of potential concern.

The maximum combined radionuclide Hazard Index from groundwater discharge under all of the alternatives was calculated to be 0.002 for benthic invertebrates under Tank Closure Alternative 1. Table P-12 presents the Hazard Indices associated with groundwater discharge to the Columbia River under all of the alternatives. Exposure to radioactive COPCs from groundwater discharge under all of the alternatives was below the 0.1-rad-per-day benchmark for wildlife receptors (i.e., muskrat, spotted sandpiper, raccoon, bald eagle, least weasel) and the 1-rad-per-day benchmark for benthic invertebrates and aquatic biota, including salmonids. Estimated hazards for the representative species indicated that no adverse effects are expected for aquatic receptors and terrestrial wildlife feeding in the Columbia River from exposure to radioactive COPCs from groundwater discharge. Because the direct impacts of groundwater discharge are expected to be small, any associated potential indirect impacts on the ecosystem would be correspondingly minor.

Exposure to chemical COPCs discharged into the Columbia River as a result of releases to groundwater under the various Tank Closure, FFTF Decommissioning, and Waste Management alternatives exceeded the Hazard Quotient criterion of 1 under all Tank Closure alternatives; Waste Management Alternative 1; and Waste Management Alternatives 2 and 3, Disposal Group 1, Subgroups 1-C, 1-E, and 1-F. In all cases, the maximum Hazard Quotient was for aquatic biota, including salmonids, exposed to chromium, assuming it was in hexavalent form (see Table P-13). The highest Hazard Quotient was 44.5 for aquatic biota, including salmonids exposed to hexavalent chromium in nearshore surface water under Tank Closure Alternative 6B, Option Case (see Table P-14). Hazard Quotients for terrestrial predators (i.e., spotted sandpiper, raccoon, bald eagle, and least weasel) feeding on Columbia River benthic invertebrates and aquatic biota, including salmonids, did not exceed the Hazard Quotient of 1.37 for nitrate. Only chromium and nitrate had Hazard Quotients exceeding 1 for aquatic and riparian receptors at the Columbia River.

The chromium Hazard Quotients above 1 did not necessarily indicate high risk to aquatic biota, including salmonids, at the Columbia River. The TRV for hexavalent chromium used to calculate salmonid Hazard

Quotients was the sensitive-species-test-effect concentration affecting 20 percent of the test population (EC<sub>20</sub>). Hexavalent chromium is highly toxic compared with the trivalent form of chromium, which is more likely to occur in oxygenated aquatic environments. Hexavalent chromium Hazard Quotients can be used to compare alternatives, but they should not be used as the sole basis for concluding that ecological resources at the Columbia River would be adversely impacted.

Given the magnitude of the Hazard Quotients and the conservative exposure assumptions and toxicological benchmarks, aquatic biota and sediment-dwelling biota in the Hanford Reach and their terrestrial predators would be unlikely to be at unacceptable risk due to the discharge of chemical COPCs in groundwater under any alternative. The modeled concentrations in nearshore surface water and sediment overestimated risk due to the conservative model assumptions, namely that all groundwater discharge occurs in the 40-meter (131-foot) nearshore zone, when in reality groundwater would likely discharge throughout the riverbed and thus be highly diluted. The model also assumed that nearshore sediment would be in equilibrium with discharging groundwater, which ignored the likely movement of surface water into the uppermost sediment layer where benthic organisms are found.

Nitrate in discharging groundwater under Tank Closure alternatives could potentially contribute to eutrophication in nearshore surface water of the Hanford Reach. Dissolved concentrations of nitrite and nitrate as nitrogen in surface water at the Richland Pump House immediately downstream of Hanford did not exceed 1.0 milligram per liter during 2006 and 2010 (Poston et al. 2007; Poston, Duncan, and Dirkes 2011). Modeled maximum nitrate concentrations in Columbia River nearshore surface water ranged from 0 milligrams per liter (FFTF Decommissioning alternatives) to 3.18 milligrams per liter (e.g., Alternative Combinations 2 and 3). Only the Tank Closure alternatives and, as a result, the alternative combinations, have predicted maximum nearshore surface-water concentrations exceeding ambient concentrations. Whether increased nitrate inputs would actually result in eutrophication depends on the amount of available phosphorus.

### **P.3.2.3      Uncertainties**

Uncertainty exists about the actual magnitude of future exposures and the threshold doses or benchmark concentration TRVs used to evaluate the long-term impacts on aquatic and riparian ecological resources of groundwater releases. The uncertainties for chemical and radiological exposure estimates result from uncertainties in the source terms and transport models. Additional uncertainties were found in the BAFs and uptake factors, which were linear models based on simplifying assumptions. The uncertainties for toxicity and radiological-effects thresholds arose from extrapolating from laboratory experiments on test species to Hanford receptor species in natural environments and uncertainty about the chemical form to which ecological receptors would be exposed, e.g., hexavalent or trivalent chromium. The lack of TRVs for some chemical COPCs and some receptors resulted in uncertainties. Combined, these uncertainties produced limited underestimates of risk and moderate overestimates of risk for different combinations of receptors and chemical or radioactive COPCs. Conservative exposure assumptions and TRVs mitigated these uncertainties, allowing confidence in “no risk” conclusions. There were large uncertainties about the impact of nitrate in groundwater releases on potential eutrophication in the Columbia River. The effects of these uncertainties on the risk indices and nitrate impacts on eutrophication were unbiased with respect to the alternatives being evaluated in this *TC & WM EIS*; thus, the results accurately reflect the relative impacts of alternatives on ecological resources.

### **P.3.3      Summary of Aquatic Impacts**

Estimated radiation doses resulting from air deposition and groundwater discharge for any of the alternatives were less than the 0.1-rad-per-day and 1-rad-per-day benchmarks for ecological receptors exposed to radioactive COPCs at the Columbia River. All Hazard Indices for radioactive COPCs associated with these alternatives were below 1. Only estimated exposures of aquatic biota to hexavalent chromium in nearshore surface water under all Tank Closure alternatives; Waste Management

Alternatives 2 and 3, Disposal Group 1, Subgroups 1-C, 1-E, and 1-F; and all three alternative combinations, as well as of least weasel to nitrate under all Tank Closure alternatives, exceeded the Hazard Quotient criterion of 1 at the Columbia River. Based on the conservative nature of the exposure assumptions, the estimated Hazard Indices and Hazard Quotients for the representative receptors indicated that no adverse effects of radioactive or chemical COPCs in air and groundwater releases to the Columbia River under the various alternatives evaluated are expected. No long-term impacts are expected on water pH of additional nitrogen and sulfur dioxides resulting from air emission and deposition in the Hanford Reach. The potential impact on aquatic biota in the Hanford Reach of nitrate in groundwater discharge is uncertain. Calculated risk indices for aquatic and riparian resources from air and groundwater releases were used in this *TC & WM EIS* to compare alternatives (Chapter 5) and evaluate cumulative impacts (Chapter 6).

#### **P.4 REFERENCES**

ANL (Argonne National Laboratory), 1999, *Risk/Impact Technical Report for the Hanford Groundwater/Vadose Zone Integration Project*, DOE/CH/CRE-7-1999, U.S. Department of Energy Center for Risk Excellence, December.

Baes C.F., III, R.D. Sharp, A.L. Sjoreen, and R.W. Shor, 1984, *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture*, ORNL-5786, Oak Ridge National Laboratory, Oak Ridge, Tennessee, September.

Baker, D.A., and J.K. Soldat, 1992, *Methods for Estimating Doses to Organisms from Radioactive Materials Released into the Aquatic Environment*, PNL-8150, Pacific Northwest Laboratory, Richland, Washington, June.

Blaylock, B.G., M.L. Frank, and B.R. O'Neal, 1993, *Methodology for Estimating Radiation Dose Rates to Freshwater Biota Exposed to Radionuclides in the Environment*, ES/ER/TM-78, U.S. Department of Energy, Office of Environmental Management, Oak Ridge, Tennessee, September.

Bryce, R.W., C.T. Kincaid, P.W. Eslinger, and L.F. Morasch, eds., 2002, *An Initial Assessment of Hanford Impact Performed with the System Assessment Capability*, PNNL-14027, Pacific Northwest National Laboratory, Richland, Washington, September.

DOE (U.S. Department of Energy), 1995, *Hanford Site Risk Assessment Methodology*, DOE/RL-91-45, Rev. 3, Richland Operations Office, Richland, Washington, May.

DOE (U.S. Department of Energy), 1998, *Screening Assessment and Requirements for a Comprehensive Assessment, Columbia River Comprehensive Impact Assessment*, DOE/RL-96-16, Rev. 1, Pacific Northwest National Laboratory and CRCIA Management Team, March.

DOE and Ecology (U.S. Department of Energy, Richland Operations Office, Richland, Washington, and Washington State Department of Ecology, Olympia, Washington), 1996, *Tank Waste Remediation System, Hanford Site, Richland, Washington, Final Environmental Impact Statement*, DOE/EIS-0189, Washington, August.

Eckerman, K.F., and J.C. Ryman, 1993, *External Exposure to Radionuclides in Air, Water, and Soil*, Federal Guidance Report No. 12, EPA-402-R-93-081, U.S. Environmental Protection Agency, Office of Radiation and Indoor Air, Washington, D.C., September.

Efroymson, R.A., M.E. Will, and G.W. Suter II, 1997, *Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process: 1997 Revision*, ES/ER/TM-126/R2, U.S. Department of Energy, Office of Environmental Management, Oak Ridge, Tennessee, November.

Efroymson, R.A., M.E. Will, G.W. Suter II, and A.C. Wooten, 1997, *Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants: 1997 Revision*, ES/ER/TM-85/R3, U.S. Department of Energy, Office of Environmental Management, Oak Ridge, Tennessee, November.

EPA (U.S. Environmental Protection Agency), 1992, *Framework for Ecological Risk Assessment*, EPA/630/R-92/001, Risk Assessment Forum, Washington, D.C., February.

EPA (U.S. Environmental Protection Agency), 1993, *Wildlife Exposure Factors Handbook*, EPA/600/R-93/187, Office of Research and Development, Office of Health and Environmental Assessment, Washington, D.C., December.

EPA (U.S. Environmental Protection Agency), 1995, *Great Lakes Water Quality Initiative Technical Support Document for the Procedure to Determine Bioaccumulation Factors*, EPA-820-B-95-005, Office of Water, Washington, D.C., March.

EPA (U.S. Environmental Protection Agency), 1997, *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments, Interim Final*, EPA 540-R-97-006, Office of Solid Waste and Emergency Response, Washington, D.C., June.

EPA (U.S. Environmental Protection Agency), 1999, *Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, Vol. 3, Peer Review Draft, EPA530-D-99-001C, Office of Solid Waste and Emergency Response, Washington, D.C., August.

EPA (U.S. Environmental Protection Agency), 2000, *Ecological Soil Screening Level Guidance-Draft*. Exhibit 1-3, "Evaluation of Dermal Contact and Inhalation Exposure Pathways for the Purpose of Setting Eco-SSLs," Washington, D.C., June 27.

EPA (U.S. Environmental Protection Agency), 2005 *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA530-R-05-006, Office of Solid Waste and Emergency Response, Washington, D.C., September.

EPA (U.S. Environmental Protection Agency), 2009, *Integrated Risk Information System*, accessed through: <http://www.epa.gov/iris/> on March 5.

Eslinger, P.W., C. Arimescu, B.A. Kanyid, and T.B. Miley, 2002, *User Instructions for the Systems Assessment Capability, Rev. 0, Computer Codes: Vol. 2, Impact Modules*, PNNL-13932, Pacific Northwest National Laboratory, Richland, Washington, June.

Howard, P.H., R.S. Boethling, W.F. Jarvis, W.M. Meylan, and E.M. Michalenko, 1991, *Handbook of Environmental Degradation Rates*, Lewis Publishers, Inc., Chelsea, Michigan.

IAEA (International Atomic Energy Agency), 1992, *Effects of Ionizing Radiation on Plants and Animals at Levels Implied by Current Radiation Protection Standards*, Technical Reports Series No. 332, Vienna, Austria, March.

Jones, D.S., G.W. Suter II, and R.N. Hull, 1997, *Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1997 Revision*, ES/ER/TM-95/R4, U.S. Department of Energy, Office of Environmental Management, Oak Ridge, Tennessee, November.

Kocher, D.C., and J.R. Trabalka, 2000, "On the Application of a Radiation Weighting Factor for Alpha Particles in Protection of Non-Human Biota," *Health Physics Society Journal*, Vol. 79, No. 4, pp. 407-411, October.

NCRP (National Council on Radiation Protection and Measurements), 1991, *Effects of Ionizing Radiation on Aquatic Organisms*, NCRP Report No. 109, Bethesda, Maryland, August 30.

NRC (U.S. Nuclear Regulatory Commission), 1977, *Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I*, Regulatory Guide 1.109, Rev. 1, Office of Standard Development, Washington, D.C., October.

NRCS (Natural Resources Conservation Service), 2008, *Official Soil Series Descriptions (OSD) with Series Extent Mapping Capabilities: Quincy Series*, accessed through <http://ortho.ftw.nrcs.usda.gov/osd/osd.html>, November.

Paragon Analytics (Paragon Analytics, Inc.), 2003, *Inorganics Case Narrative, Washington State Dept. of Ecology WTP/TOC on Soil Samples*, Order Number - 0307058, Fort Collins, Colorado, August 5.

Poston, T.M., J.P. Duncan, and R.L. Dirkes, eds., 2011, *Hanford Site Environmental Report for Calendar Year 2010 (Including Some Early 2011 Information)*, PNNL-20548, Pacific Northwest National Laboratory, Richland, Washington, September.

Poston, T.M., R.W. Hanf, J.P. Duncan, and R.L. Dirkes, eds. 2007, *Hanford Site Environmental Report for Calendar Year 2006 (Including Some Early 2007 Information)*, PNNL-16623, Pacific Northwest National Laboratory, Richland, Washington, September.

Rasmussen, J.J., 1971, *Soil Survey of Benton County Area, Washington*, U.S. Department of Agriculture, Soil Conservation Service, July.

SAIC (Science Applications International Corporation), 2011a, *Calculating Risk Indices for Long-Term Impacts to Ecological Receptors – Releases to Air*, Germantown, Maryland.

SAIC (Science Applications International Corporation), 2011b, *Calculating Risk Indices for Long-Term Impacts to Ecological Receptors – Releases to Groundwater*, Germantown, Maryland.

Sample, B.E., D.M. Opresko, and G.W. Suter II, 1996, *Toxicological Benchmarks for Wildlife: 1996 Revision*, ES/ER/TM-86/R3, U.S. Department of Energy, Office of Environmental Management, Oak Ridge, Tennessee, June.

Sample, B.E., M.S. Aplin, R.A. Efroymsen, G.W. Suter II, and C.J.E. Welsh, 1997, *Methods and Tools for Estimation of the Exposure of Terrestrial Wildlife to Contaminants*, ORNL/TM-13391, Oak Ridge National Laboratory, Oak Ridge, Tennessee, October.

Stumm, W., and J.J. Morgan, 1970, *Aquatic Chemistry: An Introduction Emphasizing Chemical Equilibria in Natural Waters*, Wiley-Interscience, New York, New York.

Suter, G.W., II, 1993, *Ecological Risk Assessment*, Lewis Publishers, Boca Raton, Florida.

Suter, G.W., II, and C.L. Tsao, 1996, *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision*, ES/ER/TM-96/R2, U.S. Department of Energy, Office of Environmental Management, Oak Ridge, Tennessee, June.

USFS, NPS, and USFWS (U.S. Forest Service, National Park Service, and U.S. Fish and Wildlife Service), 2000, *Federal Land Managers' Air Quality Related Values Workgroup (FLAG), Phase I Report*, December.

USGS (U.S. Geological Survey), 2000, *Use of Rearing Habitat by Juvenile Fall Chinook Salmon in the Hanford Reach of the Columbia River, Washington*, accessed through <http://biology.usgs.gov/wfrc/crrlhome/rearing.htm>, October 13.

Zach, R., 1985, "Contribution of Inhalation by Food Animals to Man's Ingestion Dose," *Health Physics*, Vol. 49, No. 5, pp. 737–745, November.

**Federal Register**

63 FR 26846, U.S. Environmental Protection Agency, 1998, "Guidelines for Ecological Risk Assessment," May 14.

**United States Code**

16 U.S.C. 1531 et seq., Endangered Species Act of 1973, as amended.

**U.S. Department of Energy Standards**

DOE Standard 1153-2002, *A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota*, July 2002.

