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# Environmental Risk Assessment Work Plan for the Hanford Tank Waste Treatment and Immobilization Plant

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# Notice

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# History Sheet

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# Acronyms and Abbreviations

| | |
|---|---|
| ADAF | age dependent adjustment factors |
| ADD | average daily dose |
| AE | absorption efficiency |
| AEGL | acute exposure guideline level |
| AHQ | acute hazard quotient |
| AIEC | acute inhalation exposure criterion |
| APCS | air pollution control system |
| AR | arylhydrocarbon receptor |
| AREC | acute radionuclide exposure criterion |
| AREL | acute reference exposure level |
| ATSDR | Agency for Toxic Substances and Disease Registry |
| AUF | area use factor |
| BAF | bioaccumulation factor |
| BCF | bioconcentration factor |
| BEF | bioaccumulation equivalency factor |
| BIF | boiler and industrial furnace |
| BLM | US Bureau of Land Management |
| CalEPA | California Environmental Protection Agency |
| CAS | Chemical Abstract Services |
| CDE | committed dose equivalent |
| CFR | *Code of Federal Regulations* |
| CLUP | *Comprehensive Land-Use Plan* |
| COPC | chemical of potential concern |
| CSEFH | Child-Specific Exposure Factors Handbook |

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|---|---|
| CSF | cancer slope factor |
| CSM | conceptual site model |
| CTUIR | Confederated Tribes of the Umatilla Indian Reservation |
| DCF | dose conversion factor |
| DD | daily dose |
| DEM | digital elevation model |
| DNA | deoxyribonucleic acid |
| DOE | US Department of Energy |
| DQO | data quality objective |
| DST | double-shell tank |
| DW | dry weight |
| DWP | Dangerous Waste Permit |
| EC | exposure concentration |
| ECF | elevation correction factor |
| Eco-SSL | Ecological Soil Screening Level |
| Ecology | Washington State Department of Ecology |
| EDL | estimated detection limit |
| EFH | Exposure Factors Handbook |
| EIS | Environmental Impact Statement |
| EPA | US Environmental Protection Agency |
| EPC | exposure point concentration |
| ERA | ecological risk assessment |
| ERPG | Emergency Response Planning Guideline |
| ESB | Equilibrium Partitioning Sediment Benchmark |
| ESQ | ecological screening quotient |
| ESU | evolutionarily significant unit |

| | |
|---|---|
| ETF | Effluent Treatment Facility |
| FCM | food chain multiplier |
| FCV | final chronic value |
| FEALE | Fitzner-Eberhardt Arid Lands Ecology Reserve |
| FGR | Federal Guidance Report |
| FR | *Federal Register* |
| FRA | final risk assessment |
| FW | fresh weight |
| FWS | US Fish and Wildlife Service |
| GAF | gastrointestinal absorption factor |
| HAB | Hanford Advisory Board |
| HEAST | Health Effects Assessment Summary Table |
| HEME | high-efficiency mist eliminator |
| HEPA | high-efficiency particulate air |
| HHRA | human health risk assessment |
| HHRAP | Human Health Risk Assessment Protocol |
| HI | hazard index |
| HLW | high-level waste |
| HOP | high-level waste melter process system |
| HQ | hazard quotient |
| IAEA | International Atomic Energy Agency |
| IHLW | immobilized high-level waste |
| ILAW | immobilized low-activity waste |
| ILCR | incremental lifetime cancer risk |
| IRIS | Integrated Risk Information System |
| ISCST3 | Industrial Source Complex Short Term Model, Version 3 |

| | |
|---|---|
| ISMS | integrated safety management system |
| IWAQM | Interagency Workgroup on Air Quality Modeling |
| IX | ion exchange |
| LADD | lifetime average daily dose |
| LAW | low-activity waste |
| LERF | Liquid Effluent Retention Facility |
| LOAEL | lowest observed adverse effect level |
| LOP | low activity waste primary offgas process system |
| LVP | low activity waste secondary offgas/vessel vent process system |
| MDL | method detection limit |
| MM5 | Mesoscale Model 5 |
| MSA | Mission Support Alliance |
| MSL | mean sea level |
| MTCA | *Model Toxics Control Act* |
| MW | molecular weight |
| NAAQS | National Ambient Air Quality Standard |
| NAWQC | National Ambient Water Quality Criteria |
| NCEA | National Center for Environmental Assessment |
| NCR | Nuclear Regulatory Commission |
| NOAA | National Oceanic and Atmospheric Administration |
| NOAEL | no observed adverse effect level |
| ORP | Office of River Protection |
| OSHA | Occupational Safety and Health Administration |
| OSWER | Office of Solid Waste and Emergency Response |
| PAH | polycyclic aromatic hydrocarbons |
| PCB | polychlorinated biphenyl |

| | |
|---|---|
| PCDD | polychlorinated dibenzo-p-dioxin |
| PCDF | polychlorinated dibenzofuran |
| pCi | picocurie |
| PEF | particulate emission factor |
| PIC | product of incomplete combustion |
| PJM | pulse jet mixer |
| PJV | pulse jet ventilation system |
| PNNL | Pacific Northwest National Laboratory |
| PRA | pre-demonstration test risk assessment |
| PSD | prevention of significant deterioration |
| PT | Pretreatment (Facility) |
| PVP | pretreatment vessel vent process system |
| PVV | pretreatment process vessel vent extraction |
| PWD | plant wash and disposal system |
| QF | quality factor |
| RAGS | EPA's *Risk Assessment Guidance for Superfund* |
| RAWP | risk assessment work plan |
| RCRA | Resource Conservation and Recovery Act |
| RDL | reliable detection limit |
| RCF | root concentration factor |
| RF | risk factor |
| RfC | reference concentration |
| RfD | reference dose |
| RFD | reverse flow diverter |
| RME | reasonable maximum exposure |
| ROD | Record of Decision |

| | |
|---|---|
| ROPC | radionuclide of potential concern |
| RPF | relative potency factor |
| RSL | Regional Screening Level |
| SBS | submerged bed scrubber |
| SCR | selective catalytic reduction |
| SCV | secondary chronic value |
| SF | slope factor |
| SFr | soil or sediment ingestion fraction |
| SLERA | screening-level ecological risk assessment |
| SLERAP | Screening-Level Ecological Risk Assessment Protocol |
| SLRA | screening-level risk assessment |
| SSFM | steady-state flowsheet model |
| SVOC | semivolatile organic compound |
| T&E | threatened and endangered |
| TAP | toxic air pollutant |
| TC&WM | Tank Closure and Waste Management |
| TCDD | tetrachlorodibenzo-p-dioxin |
| TCDF | tetrachlorodibenzofuran |
| TCE | trichloroethylene |
| TEDF | Treated Effluent Disposal Facility |
| TEEL | temporary emergency exposure limit |
| TEF | toxicity equivalency factor |
| TEQ | toxic equivalency |
| TIC | tentatively identified compound |
| TLP | treated low activity waste evaporation process system |
| TOE | total organic emissions |

| | |
|---|---|
| TRU | transuranic |
| TRV | toxicity reference value |
| TSS | total suspended solids |
| TUF | temporal use factor |
| TWRS | Tank Waste Remediation System |
| UHC | underlying hazardous constituent |
| UNSCEAR | United Nations Scientific Committee on the Effects of Atomic Radiation |
| UR | unit risk |
| URF | unit risk factor |
| USGS | US Geological Survey |
| USLE | Universal Soil Loss Equation |
| UTS | Universal Treatment Standards |
| VOC | volatile organic compound |
| WAC | waste acceptance criteria |
| WESP | wet electrostatic precipitator |
| WHO | World Health Organization |
| WTP | Hanford Tank Waste Treatment and Immobilization Plant |
| YN | Yakama Indian Nation |

# Glossary

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|---|---|
| Abated emissions | Emissions that have passed through WTP process mechanisms or air pollution control equipment to reduce the potential for public exposure consistent with applicable air permitting requirements. |
| Abiotic | Non-living; used to describe air, soil, sediment, and water to which receptors may be exposed. |
| Anadromous | Describing fish that spend most of their adult lives in salt water and migrate to freshwater rivers and lakes to reproduce. |
| Animal fraction ($F_A$) | Fraction of a receptor's diet that is animal or prey (unitless). |
| Benthic | Having to do with sediment at the bottom of a stream, pond, river, or lake. |
| Bioaccumulation factor (*BAF*) | Uptake factor for direct and indirect transfer of chemicals from abiotic medium and food to an organism, expressed as the ratio of the concentration of a chemical in an organism and the concentration of the chemical in an abiotic medium that is a direct source of the chemical for the organism and which the organism's food is also exposed. |
| Bioaccumulation factor for an animal product | The ratio of the chemical concentration in fresh weight tissue to the daily intake of the chemical by the animal. |
| Bioconcentration factor (*BCF*) | Uptake factor for direct transfer of chemicals from abiotic medium only to an organism, expressed as the ratio of the concentration of a chemical in an organism and the concentration of the chemical in an abiotic medium that is a direct source of the chemical for the organism. |
| Biomagnification factor (*BMF*) | The ratio of the concentration of a chemical in a consumer and the concentration of the chemical in its food. |
| CALPUFF | An air dispersion model. This model handles winds more realistically than the ISCST3 model. |
| Cancer Slope Factor (*CSF*) | Plausible upper-bound estimate (for chemicals) and central estimate (for radionuclides) of the probability of a cancer response per unit intake over a lifetime. |
| Carnivore | An animal that eats other animals. |

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| Conservation of mass | The conservation of mass is a fundamental concept of physics. Within a defined system, the amount of mass remains constant (that is, mass cannot be created from nothing). For this discussion, the defined system is the release of chemical emissions from the WTP, subsequent deposition to soil, and uptake into biological organisms. |
| Conservative | Used in the RAWP to refer to conditions that implicitly or explicitly overestimate exposure. In some cases the word "conservative" is used to refer to procedures that result in higher risks than would have been calculated by explicitly using methods in the guidance. |
| Default | A predetermined numerical value that is used in place of a missing value. |
| Dose | The amount of a chemical taken in by an organism. |
| Driver | A COPC or ROPC that contributes 10 % or more of the threshold incremental lifetime cancer risk for human risk, or 10 % or more of the threshold hazard index for human or ecological risk. |
| $EC_{20}$ | The lowest chronic exposure that would reduce population recruitment by 20%. |
| Exposure duration (*ED*) | Time period (typically in years) over which a receptor is exposed. |
| Estimated Exposure Level (*EEL*) | The mass of constituent per mass of media [communities] or mass daily dose constituent ingested per mass body weight-day [class-specific guilds] of ecological receptors. EEL is the same as the daily dose (DD). |
| Exposure frequency (*EF*) | Number of days each year during which a receptor is exposed. |
| Exposure time (*ET*) | Number of hours per exposure event in which a receptor is exposed. |
| Exposure Point Concentration (EPC) | A concentration to which a receptor is exposure via an inhalation, ingestion, or adsorption pathway. |
| Ecological screening quotient (*ESQ*) | The ratio of the constituent estimated exposure level (EEL) and the toxicity reference value (TRV). An ESQ value in excess of one is indicative of a potential risk issue. |
| Feed | For the animals included in this discussion (cattle, wild game, swine, poultry, and wildfowl), feed may include forage, grain, or silage. |

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| Flue | A pipe or tube conveying air and other abated air contaminants to the environment. Each WTP facility has several flues dedicated to process and vessel ventilation, pulse jet mixer, process cell, and building exhaust. Flues from each of the three main WTP facilities are bundled together into a structural steel lattice that is referred to as the facility's stack or effective stack. |
| Food chain | A sequence of discrete feeding relationships between different species populations or groups of similar organisms. |
| Food chain multiplier (*FCM*) | A food chain multiplier is the ratio of a bioaccumulation factor to the bioconcentration factor. A food chain multiplier is used to estimate the concentration of a chemical in a predator eating prey. Ratios of FCMs are used in wildlife exposure modeling for a predator to account for the increase in the concentration of a chemical as it moves from an animal to its predator. |
| Forage | Primarily pasture grass and hay, including wild vegetation, exposed to wet and dry deposition, air-to-plant transfer, and root uptake of contaminants. |
| Fugitive emissions | Emissions of air contaminants that do not reasonably pass through a stack, chimney, vent, or other functionally-equivalent opening. As discussed in Section 3, all process vessels and piping are located inside the Pretreatment, LAW vitrification, or HLW Vitrification buildings. Each building has a separate HVAC system that handles emissions from the process equipment (including leaks or spills). Process piping between buildings is double lined and therefore does not contribute to fugitive emissions. However, the building ventilation system abates only particles or aerosols; vapors effectively pass through the building filters and are emitted. These emissions are addressed as unabated emissions. |
| Grain | Barley, wheat, or similar protected seed product (domestic and wild), limited to exposure to contaminants solely through root uptake. |
| Hanford offsite maximum | Location of maximum predicted ground-level concentrations of both airborne and deposited emissions outside the Hanford Site boundary. This location will have the highest modeled exposures on land that DOE does not control. |
| Herbivore | An animal that eats primarily plant material. |
| Home Range (*HR*) | The area an ecological receptor occupies for breeding, hunting, and or grazing. |
| Human Health Risk Assessment Protocol (HHRAP) | The human health risk assessment guidance document (EPA. 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, Final, EPA/530/R-05/006). |
| Insectivore | An animal that eats primarily insects and other invertebrates. |

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| ISCST3 | Industrial Source Complex Short-Term Model; an earlier air dispersion model used by the WTP, now replaced by CALPUFF. |
| Joule-heated | Heated by passing an electric current directly through the material. |
| LD50 | The median lethal dose of a substance, or the amount required to kill 50% of a given test population. |
| LOAEL | Lowest observed adverse effect level; the lowest dose of a toxic chemical that caused an observable adverse effect in a toxicity test on the endpoint being measured; if the range of doses tested did not include a dose low enough to cause a NOAEL (no observed adverse effect level), it is not possible to determine how close the LOAEL is to a no adverse effect level dose. |
| Mass density | The weight of material in a unit area given a specified soil depth. |
| Mass-limited uptake factor | An uptake factor that results in 100 % of an available chemical being transferred into a biological receptor but no more. |
| mGy | MilliGray, a unit of absorbed radiation equal to 0.001 Joule/kg. |
| NOAEL | No observed adverse effect level; the highest dose of a toxic chemical that did not cause any observable adverse effect in a toxicity test on the endpoint being measured; if the range of doses tested did not include a dose high enough to cause a LOAEL, it is not possible to determine how close the NOAEL is to an adverse effect level. |
| Omnivore | An animal that eats both plants and animals. |
| Onsite ground maximum | Location of maximum predicted ground-level concentrations of both airborne and deposited emissions on the Hanford Site. This location will have the highest modeled exposures for current workers on the Hanford Site and for ecological receptors. |
| Planktivorous | Describing fish that eat plankton. |
| Plant fraction ($F_P$) | Fraction of a receptor's diet that is plant (unitless). |
| Plausible | Describing exposure scenarios for receptors that currently exist, or may reasonably be expected to exist in the future, at a given location (for example, a future resident at the Hanford offsite maximum location). Exposure parameters for plausible scenarios are conservative. |

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| Product of incomplete combustion (PIC) | A chemical produced when combustion of an organic COPC does not completely convert the COPC to carbon dioxide and oxides of nitrogen, sulfur, and/or any other element that makes up the chemical structure of the COPC. |
| Quality Factor *(QF)* | A factor that describes the relative biological activity (i.e., quality) of alpha radiation compared to gamma radiation. |
| rad | A unit of absorbed radiation equal to 0.01 Joule/kg. |
| Regression | A mathematical method that determines how closely an equation fits a series of data points. Regression can be used to derive a generalized equation from a number of observed values, for example, the equations to calculate bioaccumulation factors from $\log_{10} K_{ow}$ values. |
| Root Concentration Factor (*RCF*) | A factor used to calculate the belowground transfer of a chemical from the soil to a root vegetable. |
| Sensitive species $EC_{20}$ | A benchmark calculated from chronic toxicity test data that is intended to allow no more than a 20 % reduction in weight or number of offspring in 95 % of species. |
| Silage | Vegetation and livestock feed that has been stored and fermented, exposed to wet and dry deposition, air-to-plant transfer, and root uptake of contaminants. Silage is limited to consumption by domestic livestock. |
| SLERAP | Screening-level ecological risk assessment protocol; the ecological risk assessment guidance document (EPA. 1999. *Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities,* Peer Review Draft, EPA 530-D-99-001A). |
| Soil pore water | Water in the interstitial spaces between the mineral and organic particles of soil. |
| Stack | A structural steel lattice that supports individual flues from each of the WTP's three main facilities. Stack parameters important to air dispersion modeling including flow rate, temperature and velocity can be calculated arithmetically as the weighted average of the combined flues within the steel lattice. |
| Steady state | The condition where the value of a variable does not change through time. |
| Surrogate | A chemical with known bioaccumulation or toxicity factors which are used in lieu of those factors for a COPC for which the factors are not known. The surrogate is sufficiently chemically similar to the COPC that the COPC is expected to have similar bioaccumulation or toxicity factors to those of the surrogate. |

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| T&E species | Plant and animal species that have been designated by law as threatened or endangered. |
| Target analyte | An analyte that is expected to occur in WTP airborne emissions and can readily be identified and quantified by chemical analytical methods that will be used at the WTP. |
| Temporal-Use Factor (*TUF*) | Fraction of time each year that an ecological receptor is in the vicinity of the exposure location during which it forages or resides at the exposure location. The TUF will be assumed to be 1 for all receptors. |
| Tentatively identified compound (TIC) | A compound that is detected in environmental samples that is not a target analyte. TICs are identified generally as a result of using mass spectrometry techniques. When a TIC is identified, it can be definitively identified by analyzing an authentic standard of the putative unknown. |
| Tilled soil | Soil evenly mixed down to a depth of 20 cm. |
| Toxicity Equivalance Factor | The ratio of toxicity of a polychlorinated dibenzodioxin, dibenzofuran, and biphenyl COPC to the toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin. |
| Toxicity Reference Values (*TRV*) | A threshold value used to characterize ecological risk based upon a hierarchy of toxicological data. |
| Unabated emissions | Emissions of air contaminants that would result if all abatement control equipment in the WTP did not exist, but operations were otherwise normal. There will be no unabated emissions from the process flues at the WTP facility at any time. There is the possibility that a fraction of waste being treated by the WTP will leak or spill from ancillary equipment such as valves, equipment seals, and connectors and enter WTP's process cells. Process cell ventilation will pass through HEPA filtration prior to discharge to the environment, however a portion of the organics that are vapor phase will not be abated by HEPA filters. These emissions are referred to as process cell organic emissions and have been estimated and included along with the abated process emissions for purposes of this risk assessment. |
| Untilled soil | Soil evenly mixed down to a depth of 2 cm. |
| Uptake factor | The ratio of a chemical concentration in one environmental medium to its concentration in another. |
| Vapor Partitioning Coefficient ($F_v$) | Fraction of the COPC or ROPC air concentration in vapor phase (unitless). Particle phase constituents are defined as having an $F_v < 0.05$ (that is, having a vapor concentration of less than 5 percent). Vapor phase constituents have an $F_v = 1$. A subset of organic COPCs falls into the category of particle-bound ($0.05 < F_v < 1$). Particle-bound constituents have a portion of vapor condensed onto the particle surface. |

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| Wetland | An area whose soil is saturated with water; saturation causes low oxygen concentrations in the soil and results in the growth of plants specialized to live with low oxygen levels. |
| Worst-case | Describing exposure scenarios for receptors that are not reasonably expected to exist now or in the future at the specified location (for example, a future resident at the onsite ground maximum location). Exposure parameters for worst-case scenarios are conservative. |

# Symbol Glossary

| | |
|---|---|
| $A$ | area of exposure for an ecological receptor |
| $a$ | empirical intercept coefficient |
| $A_{beef}$ | concentration of constituent in beef |
| $AbF$ | absorbed fraction of energy from radiation |
| $ACF$ | area correction factor for an infinite slab |
| $A_{chicken}$ | concentration of constituent in chicken |
| $ADAF$ | age dependent adjustment factor for cancer slope factor for mutagenic chemicals |
| $ADD$ | average daily dose: for evaluating exposure to noncarcinogenic COPCs, the intake is referred to as average daily dose ($ADD$) |
| $ADD_{infant}$ | average daily dose of constituents from breast milk |
| $A_{egg}$ | concentration of constituent in eggs |
| $A_{fowl}$ | concentration of a constituent in wildfowl |
| $A_{game}$ | concentration of a constituent in wild game |
| $AHQ$ | acute hazard quotient |
| $A_I$ | area of impervious watershed receiving deposition |
| $A_i$ | concentration of constituent in animal product $i$ |
| $AIEC$ | acute inhalation exposure criteria |
| $A_L$ | total watershed area receiving deposition |
| $A_{milk}$ | concentration of constituent in milk |
| $A_{pork}$ | concentration of constituent in pork |
| $AREC_E$ | acute radionuclide exposure criteria for external gamma |
| $AREC_I$ | acute radionuclide exposure criteria for inhalation |
| $AREC_M$ | acute radionuclide exposure criteria for ROPC $i$ corrected for the presence of multiple ROPCs |

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|---|---|
| $AREC_R$ | total acute radionuclide exposure criteria |
| $AT_C$ | averaging time is the period over which exposure is averaged. The averaging time for carcinogens is based on a lifetime exposure of 70 years |
| $AT_{infant}$ | infant averaging time |
| $AT_N$ | averaging time is the period over which exposure is averaged. For noncarcinogens, $AT_N$ is equal in length to the exposure duration (*ED*) |
| *AUF* | area use factor; the portion or percent of an ecological receptor's home range area used by the receptor |
| $A_w$ | average annual water body surface area |
| *b* | empirical slope coefficient |
| $Ba_{beef}$ | biotransfer factor for beef, expressed as the ratio of the chemical concentration in fresh weight tissue to the daily intake of the chemical by beef cattle |
| $Ba_{chicken}$ | biotransfer factor for chickens, expressed as the ratio of the chemical concentration in fresh weight tissue to the chemical intake from the feed by chickens |
| $Ba_{deer}$ | biotransfer factor for wild game animals |
| $Ba_{egg}$ | biotransfer factor for eggs, expressed as the ratio of the chemical concentration in fresh weight tissue to the chemical intake from the feed by chickens |
| *BAF* | bioaccumulation factor |
| $BAF_{fish}$ | bioaccumulation factor for a constituent in fish |
| $BAF_{organs}$ | organ-specific bioaccumulation factor for a constituent in fish |
| $Ba_{milk}$ | biotransfer factor for milk, expressed as the ratio of the chemical concentration in fresh weight tissue to the daily intake of the chemical by milk cows |
| $Ba_{pork}$ | biotransfer factor for pork, expressed as the ratio of the chemical concentration in fresh weight tissue to the daily intake of the chemical by swine |
| *BCF* | uptake factor for direct transfer of chemicals from abiotic medium only to an organism, expressed as the ratio of the concentration of a chemical in an organism and the concentration of the chemical in an abiotic medium that is a direct source of the chemical for the organism |
| $BCF_A$ | generic notation for animal bioconcentration factor, used to calculate the transfer of a constituent from the exposure media to the animal |
| $BCF_{fish}$ | fish bioconcentration factor, used to calculate the transfer of a constituent from surface water to a fish |

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|---|---|
| $BCF_i$ | generic notation for the media-to-animal or media-to-plant bioconcentration factor for constituent *i* in the exposure media |
| $BCF_{inv}$ | aquatic invertebrate bioconcentration factor, used to calculate the transfer of a constituent from surface water to an aquatic invertebrate |
| $BCF_P$ | plant bioconcentration factor, used to calculate the transfer of a constituent from plant to an animal |
| $BCF_r$ | soil-to-plant bioconcentration factor, used to calculate the transfer of a constituent from soil, sediment, or water to a plant |
| $BCF_S$ | soil bioconcentration factor, used to calculate the transfer of a constituent from soil or sediment to a plant or animal |
| $BCF_W$ | water bioconcentration factor, used to calculate the transfer of a constituent from water to a plant or animal |
| $BD$ | soil bulk density |
| $BEF$ | the ratio of bioaccumulation of a polychlorinated dibenzodioxin, dibenzofuran, or biphenyl COPC to the bioaccumulation of 2,3,7,8-tetrachlorodibenzo-p-dioxin |
| $Br$ | soil-to-plant uptake factor; uptake is through roots or root hairs |
| $BR$ | breathing rate of standard man |
| $Br_{ag}$ | soil-to-plant uptake factor for aboveground plants, accounting for the uptake from soil and the subsequent transport of chemicals through the roots to the aboveground parts of a plant |
| $Br_{rootveg}$ | soil-to-plant uptake factor for chemicals in root vegetables, accounting for the uptake from soil to the belowground root vegetable or produce |
| $Bs$ | soil bioavailability factor |
| $BSAF_{fish}$ | biota-to-sediment accumulation factor for fish |
| $Bv_{ag}$ | constituent air-to-plant biotransfer factor for aboveground plant |
| $BW$ | total body weight of a receptor (adult or child) |
| $BW_{infant}$ | infant body weight |
| $C$ | USLE cover management factor |
| $C_A$ | concentration of a COPC or ROPC in the tissue of an animal receptor resulting from ingestion of contaminated soil, sediment, water, and food |
| $C_a$ | concentration of a COPC or ROPC in air resulting from WTP airborne emissions |

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|---|---|
| $C_{A(C\text{-}14)}$ | concentration of carbon-14 in the surrounding air |
| $C_{A(H\text{-}3)}$ | concentration of tritium in the surrounding air |
| $C_{acute}$ | one-hour acute air concentration |
| $C_{AP}$ | concentration of a COPC or ROPC in aquatic plants resulting from uptake of WTP airborne emissions directly and from water |
| $C_{BS}$ | bed sediment concentration |
| $CDE$ | committed dose equivalent |
| $CDE+D$ | committed dose equivalent for radionuclide *i* and its daughter products |
| $CDE_i$ | committed dose equivalent for radionuclide *i* |
| $C_{dw}$ | dissolved-phase water concentration of constituent |
| $C_{fish}$ | concentration of constituent in fish |
| $C_{forage}$ | modeled concentration in forage |
| $C_{grain}$ | modeled concentration in grain |
| $C_i^{GRAV}$ | stack concentration of $i^{th}$ identified nonvolatile COPC |
| $C_i^{SVOC}$ | stack concentration of $i^{th}$ identified semivolatile COPC |
| $C_i^{VOC}$ | stack concentration of $i^{th}$ identified volatile COPC |
| $C_M$ | generic notation for the concentration of a COPC or ROPC in exposure media such as air, soil, water and sediment |
| $C_{milkfat}$ | concentration of constituent in milk fat of breast milk for a specific exposure scenario of a nursing mother |
| COPC | chemical of potential concern |
| $C_p$ | concentration of a COPC or ROPC in plants resulting from uptake of WTP airborne emissions directly and from soil |
| $C_{pw}$ | concentration of a COPC or ROPC in soil pore water resulting from deposition of WTP airborne emissions |
| $CR_{forage}$ | consumption rate of forage by a receptor (quantity consumed per day) |
| $CR_{grain}$ | consumption rate of grain by a receptor (quantity consumed per day) |
| $CR_i$ | consumption rate of animal product *i* |

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| $CR_{silage}$ | consumption rate of silage by a receptor (quantity consumed per day) |
| $CR_{soil}$ | consumption rate of soil by a receptor (quantity consumed per day) |
| $Cs$ | concentration in soil |
| $Cs_2$ | soil concentration at the 2 cm soil depth (untilled soil) |
| $Cs_{15}$ | soil concentration at the 15 cm soil depth (root zone) |
| $Cs_{20}$ | soil concentration at the 20 cm soil depth (tilled soil) |
| $C_{sed}$ | constituent concentration in bed sediment |
| $CSF$ | Cancer Slope Factor: plausible upper-bound estimate (for chemicals) and central estimate (for radionuclides) of the probability of a cancer response per unit intake over a lifetime |
| $CSF_{adj}$ | adjusted cancer slope factor for 2 cm depth |
| $CSF_{dermal}$ | dermal cancer slope factor |
| $CSF_{food}$ | radionuclide-specific food ingestion cancer slope factor |
| $CSF_i$ | constituent-specific external pathway cancer slope factor for pathway *i* |
| $CSF_{o_i}$ | radionuclide-specific oral (ingestion) pathway cancer slope factor for pathway *i* |
| $CSF_{water}$ | radionuclide-specific water ingestion cancer slope factor |
| $C_{silage}$ | modeled concentration in silage |
| $C_{soil}$ | modeled concentration in soil |
| $Cs_{tD}$ | maximum soil concentration; occurs at the time emissions cease |
| $C_{TP}$ | concentration of a COPC or ROPC in terrestrial plants resulting from uptake of WTP airborne emissions directly and from soil and air |
| $C_{V(C\text{-}14)}$ | concentration of carbon-14 in vegetation |
| $C_{V(H\text{-}3)}$ | concentration of tritium in vegetation |
| $C_{wctot}$ | total constituent concentration in the water column |
| $C_{wtot}$ | total water body constituent concentration, including the water column and bed sediment |

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| $Cyp$ | unitized yearly average air concentration from particles $Cyp$, from the air dispersion modeling, is stack-specific (subscript 1 for 1 micron particle size [$Cyp_1$], subscript 2.5 for 2.5 micron particle size [$Cyp_{2.5}$]) |
| $Cyv$ | unitized yearly average air concentration from vapor $Cyv$, from the air dispersion modeling, is stack-specific |
| $d_{bs}$ | depth of the upper benthic sediment layer |
| $DCF$ | dose conversion factor, a multiplier used to convert the concentration of an ROPC in air, soil, or water to the external radiation dose absorbed by a receptor |
| $DD$ | daily dose; the amount of a chemical taken in by an organism per unit body weight each day (mg/kg of receptor body weight per day) |
| $DD_{belowgrd}$ | external dose from exposure to belowground soil |
| $DD_{E_{air}}$ | external radiation dose from airborne ROPCs surrounding the receptor |
| $DD_{E_{sed}}$ | external radiation dose from ROPCs in sediment due to immersion in sediment and/or sediment/water interface |
| $DD_{E_{soil}}$ | external radiation dose from ROPCs in soil to a receptor that either is immersed in soil or is on or near the surface of the soil |
| $DD_{E_{water}}$ | external radiation dose from ROPCs in water; receptor either is immersed in water or is on or near the surface of the water |
| $DD_I$ | internal radiation dose to a receptor due to the receptor's own tissue concentration of ROPCs |
| $DD_{imm}$ | external radiation dose from ROPCs in water to a receptor that is immersed in water |
| $DD_{near}$ | external radiation dose from ROPCs in water to a receptor that is above but near the surface of the water |
| $DF_{abovegrd}$ | factor for converting activity of radionuclide in soil to external dose from exposure to aboveground from untilled soil |
| $DF_{air}$ | factor for converting activity of the ROPC in air to external dose from air |
| $DF_{imm}$ | factor for converting activity of radionuclide in water to external dose from immersion water |
| $DF_{near}$ | factor for converting activity of radionuclide in water to external dose from exposure near water, or due to surface water |
| $DF_{s/w}$ | factor for converting activity of radionuclide in sediment/water interface to external dose from exposure to the sediment/water interface |

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| $DF_{sed}$ | factor for converting activity of radionuclide in sediment contact and immersion to external dose from exposure to the sediment |
| $DF_{sed,imm}$ | factor for converting activity of radionuclide in sediment to external dose from exposure due to burial in sediment |
| $DF_{water}$ | factor for converting activity of radionuclide in water to external dose from exposure to water |
| $DL$ | dose limit of 0.1 rem (100 mrem) |
| $Ds$ | deposition term to soil |
| $d_{wc}$ | average annual depth of the water column |
| $Dydp$ | unitized yearly average dry deposition from particle phase from the air dispersion modeling, is stack-specific (subscript 1 for 1 micron particle size [$Dydp_1$], subscript 2.5 for 2.5 micron particle size [$Dydp_{2.5}$]) |
| $Dydv$ | unitized yearly average dry deposition from vapor phase from the air dispersion modeling, is stack-specific (subscript 1 for 1 micron particle size [$Dydv_1$], subscript 2.5 for 2.5 micron particle size [$Dydv_{2.5}$]) |
| $Dywp$ | unitized yearly average wet deposition from particle phase from the air dispersion modeling, is stack-specific (subscript 1 for 1 micron particle size [$Dywp_1$], subscript 2.5 for 2.5 micron particle size [$Dywp_{2.5}$]) |
| $Dywv$ | unitized yearly average wet deposition from vapor phase from the air dispersion modeling, is stack-specific (subscript 1 for 1 micron particle size [$Dywv_1$], subscript 2.5 for 2.5 micron particle size [$Dywv_{2.5}$]) |
| $E$ | average energy emitted as radiation (MeV per disintegration) × proportion of disintegrations producing radiation (MeV per disintegration) |
| $E_v$ | evapotranspiration |
| $e$ | Euler's number, exponential function (base of the natural logarithm, unitless), $e = \sum_{i=0}^{\infty} \frac{1}{i!} \approx 2.718282$ |
| $EC_{inh}$ | exposure concentration of constituents through inhalation of emissions |
| $EC_{inh_{sweatlodge}}$ | exposure concentration of constituents through inhalation in the sweat lodge |
| $ED$ | exposure duration: time period (typically in years) over which a receptor is exposed |
| $ED_{infant}$ | infant exposure duration |

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| *EEL* | Estimated Exposure Level: the mass of constituent per mass of media [communities] or mass daily dose constituent ingested per mass body weight-day [class-specific guilds] of ecological receptors *EEL* is the same as the daily dose (*DD*) |
| *EF* | exposure frequency: number of days each year during which a receptor is exposed |
| $EF_{infant}$ | infant exposure exposure frequency |
| *ESQ* | ecological screening quotient: the ratio of the constituent estimated exposure level (*EEL*) and the toxicity reference value (*TRV*) An *ESQ* value in excess of one is indicative of a potential risk issue |
| *ET* | exposure time: number of hours per exposure event in which a receptor is exposed |
| $ET_i$ | receptor-specific exposure time fraction indoors |
| $ET_o$ | receptor-specific exposure time fraction outdoors |
| $f_1$ | fraction of ingested Constituent that is stored in fat |
| $f_2$ | fraction of mother's weight that is fat |
| $f_3$ | fraction of breast milk that is fat |
| $f_4$ | fraction of ingested constituent that is absorbed |
| $F_A$ | animal fraction; fraction of a receptor's diet that is animal or prey |
| $f_{bs}$ | fraction of total water body constituent concentration in the benthic sediment |
| *FCM* | food chain multiplier: A food chain multiplier is the ratio of a bioaccumulation factor to the bioconcentration factor A food chain multiplier is used to estimate the concentration of a chemical in a predator eating prey. Ratios of *FCM*s are used in wildlife exposure modeling for a predator to account for the increase in the concentration of a chemical as it moves from an animal to its predator |
| $F_{fish}$ | fraction of ingested fish that is contaminated |
| $F_{forage}$ | fraction of forage grown on contaminated soil and ingested by the animals |
| $F_{grain}$ | fraction of grain grown on contaminated soil and ingested by domestic livestock |
| $F_i$ | fraction of ingested soil that is contaminated |
| $F_i$ | fraction of media *i* that is contaminated for a given exposure pathway |
| $f_i$ | decay probability of radionuclide *i* |
| $F_{imm}$ | fraction of time receptor spends immersed in water |

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| $f_{lipid}$ | fish lipid content |
| $F_{near}$ | fraction of time receptor spends near or swimming on the surface of the water |
| $f_{oc}$ | fraction of the dry mass of soil consisting of organic carbon, for example, particle-bound, dissolved, or emulsified organic chemicals and decaying plant and animal material |
| $F_P$ | plant fraction: fraction of a receptor's diet that is plant |
| $F_{s/w}$ | fraction of time receptor spends at the sediment-water interface |
| $F_{sed}$ | fraction of time receptor spends buried in sediment |
| $F_{TOE}^{GRAV}$ | total organic emission factor for the nonvolatile fraction |
| $F_{TOE}^{SVOC}$ | total organic emission factor for the volatile fraction |
| $F_{TOE}^{SVOC}$ | total organic emission factor for the semivolatile fraction |
| $F_v$ | vapor partitioning coefficient: fraction of the constituent air concentration in vapor phase |
| $Fw$ | fraction of constituent wet deposition that adheres to plant surfaces |
| $f_{wc}$ | fraction of the total water body constituent concentration in the water column |
| $g/s$ | grams per second |
| $h$ | biological half-life of COPC or effective half-life of ROPC |
| $h_b$ | biological half-life of ROPC |
| $HI$ | hazard index for a specific exposure pathway |
| $HQ$ | hazard quotient |
| $HR$ | home range: the area where an ecological receptor for breeding, hunting, and or grazing |
| $h_r$ | radiological half-life of ROPC |
| $Humidity$ | absolute humidity of the atmosphere |
| $I$ | irrigation |
| $I_{ag}$ | intake of constituent through ingestion of produce |
| $I_d$ | intake of COPCs from adult dermal absorption within the sweat lodge |

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| $I_{d,l}$ | intake of COPCs from adult dermal absorption of condensate within the sweat lodge |
| $I_{d,v}$ | intake of COPCs from adult dermal absorption of vapors within the sweat lodge |
| $I_{dw}$ | intake of constituents from drinking water |
| $I_{fish}$ | intake of constituent from fish |
| $I_i$ | intake of constituent from animal product *i* |
| $I_{inh_{sweatlodge}}$ | intake of constituent via inhalation in the sweat lodge |
| $I_{inhsoil}$ | intake of constituent through inhalation of resuspended soil |
| $I_{ira}$ | external exposure to radiation from ROPCs in air |
| $I_{irs}$ | external exposure to radiation from ROPCs in soil |
| $I_{soil}$ | intake of constituent due to soil ingestion |
| $ILCR$ | incremental lifetime cancer risk |
| $ILCR_e$ | incremental lifetime cancer risk from external pathways |
| $ILCR_{e_{air}}$ | incremental lifetime cancer risk due to external exposure to radionuclides in air |
| $ILCR_{e_{dermal}}$ | incremental lifetime cancer risk due to external pathways |
| $ILCR_{e_{soil}}$ | incremental lifetime cancer risk due to external exposure to radionuclides in soil |
| $ILCR_o$ | incremental lifetime cancer risk from oral (ingestion) pathways |
| $IR$ | inhalation rate |
| $IR_F$ | daily food ingestion rate |
| $IR_{infant}$ | daily infant ingestion rate of breast milk |
| $IR_W$ | daily water ingestion rate |
| $K$ | USLE erodibility factor |
| $Kd_{bs}$ | bed sediment/sediment pore water partition coefficient |
| $Kd_s$ | soil-water partitioning coefficient |
| $Kd_{sw}$ | suspended sediments/surface water partition coefficient |

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| $K_G$ | gas-phase transfer coefficient |
| $K_{oc}$ | soil organic carbon-water partitioning coefficient (concentration of chemical in soil, expressed as soil carbon, relative to its solubility in water) |
| $K_{ow}$ | octanol/water partitioning coefficient (ratio of the solute concentration in the water-saturated octanol phase to the solute concentration in the octanol-saturated water phase) |
| $kp$ | plant surface loss coefficient |
| $Kp$ | permeability constant |
| $ks$ | overall soil loss constant due to all processes |
| $k_{wt}$ | overall total water body constituent dissipation rate constant |
| $LADD$ | lifetime average daily dose  For evaluating exposure to carcinogenic compounds, the intake is referred to as lifetime average daily dose |
| $LADD_d$ | lifetime average daily dose, or intake, $I_d$, from dermal absorption of condensate and vapors within the sweat lodge |
| $LADD_i$ | lifetime average daily dose, or intake, $I$, of constituent via ingestion pathway $i$ (dermal absorption, external air and soil exposure) |
| $LADD_{ira}$ | lifetime average daily dose, or intake, $I_{ira}$, from radiation from ROPCs in air |
| $LADD_{irs}$ | lifetime average daily dose to radiation from ROPCs in soil, $I_{irs}$, from ROPCs in soil |
| $LADD_o$ | lifetime average daily dose, or intake, of constituent via ingestion |
| $LADD_{o_i}$ | lifetime average daily dose, or intake, of constituent via ingestion pathway $i$ (water ingestion, food ingestion, and soil ingestion, as applicable) |
| $L_{DEP}$ | total (wet and dry) particle-phase and total (wet and dry) vapor-phase direct deposition load to water body |
| $L_{DIF}$ | vapor-phase dry deposition diffusion load to water body |
| $L_E$ | soil erosion load to the surface water body |
| $LOAEL$ | lowest observed adverse effect level |
| $L_R$ | runoff load from pervious surfaces |
| $L_{RI}$ | runoff load from impervious surfaces |
| $LS$ | USLE length-slope factor |

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| $L_T$ | total constituent load to the water body, including deposition, runoff, and erosion |
| $m$ | maternal intake of constituents from all adult exposures |
| $MF$ | metabolism factor |
| $MF$ | modifying factor |
| $MW_w$ | molar weight of water |
| $n$ | number of samples in the data set |
| $NOAEL$ | no observed adverse effect level |
| $OC_{sed}$ | fraction of organic carbon in bottom sediment |
| $P$ | precipitation |
| $p$ | ratio of the total annual release time to the total annual time during which photosynthesis occurs |
| $Pd$ | concentration of constituent in aboveground plant due to direct (wet and dry) deposition |
| $Pd_{(Hg)}$ | concentration of total mercury in aboveground produce due to direct (wet and dry) deposition |
| $Pd_{(Hg^{2+})}$ | concentration of divalent mercury in aboveground produce due to direct (wet and dry) deposition |
| $Pd_{(MHg)}$ | concentration of methyl mercury in aboveground produce due to direct (wet and dry) deposition |
| $P_{domestic\ grain}$ | concentration of a constituent in grain that is ingested by a chicken |
| $PEF$ | particulate emission factor |
| $PF$ | USLE supporting practice factor |
| $P_{forage}$ | concentration of a constituent in forage that is ingested by the wild game |
| $P_i$ | concentration of constituent in plant type *i* that is ingested by livestock or game |
| $Pr_{ag}$ | concentration of constituent in aboveground plant due to root uptake |
| $Pr_{bg}$ | concentration of constituent in belowground plant due to root uptake |
| $Pv$ | concentration of constituent in aboveground plant due to air-to-plant transfer |
| $Pv_{(Hg)}$ | concentration of total mercury in aboveground plant due to air-to-plant transfer |

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| $Pv_{(Hg^{2+})}$ | concentration of divalent mercury in aboveground plant due to air-to-plant transfer |
| $Pv_{(MHg)}$ | concentration of methyl mercury in aboveground plant due to air-to-plant transfer |
| $P_{wild\ grain}$ | concentration of a constituent in grain that is ingested by a wildfowl |
| $Q$ | constituent-specific emission rate obtained from calculations after the air dispersion modeling |
| $QF$ | quality factor: a factor that describes the relative biological activity (ie, quality) of alpha radiation compared to gamma radiation |
| $Qp_{forage(deer)}$ | quantity of forage eaten by the wild game animals per day |
| $Qp_{forage(i)}$ | amount of forage eaten by animal product *i* per day |
| $Qp_{grain(chicken)}$ | quantity of grain eaten by the wildfowl per day |
| $Qp_{grain(i)}$ | amount of grain eaten by animal product *i* per day |
| $Qp_{i(beef)}$ | quantity of plant type *i* eaten by the beef cattle per day |
| $Qp_{i(chicken)}$ | quantity of plant type *i* eaten by the chicken per day |
| $Qp_{i(milk)}$ | quantity of plant type *i* eaten by the dairy cattle per day |
| $Qp_{i(pork)}$ | quantity of plant type *i* eaten by the pork per day |
| $Qp_{silage(i)}$ | amount of silage eaten by animal product *i* per day |
| $Qs_{soil(beef)}$ | quantity of soil ingested by the beef cattle per day |
| $Qs_{soil(chicken)}$ | quantity of soil ingested by the chicken per day |
| $Qs_{soil(milk)}$ | quantity of soil ingested by the dairy cattle per day |
| $Qs_{soil(pork)}$ | quantity of soil ingested by the pork per day |
| $R$ | ideal gas law constant, $R$ = 0.08205746 L·atm/mol·°K |
| $r$ | radius of sweat lodge |
| $\rho_a$ | density of air |
| $RCF$ | root concentration factor: used to calculate the belowground transfer of a chemical from the soil to a root vegetable |
| $RF$ | USLE rainfall (or erosivity) factor |
| $RF_{air}$ | radionuclide-specific risk coefficient for morbidity for pathway *i* (external air) |

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| *RfC* | reference concentration |
| *RfD* | reference dose |
| $Risk_{lifetime}$ | combined lifetime risk due to exposure to a specific mutagenic chemical via a specific exposure pathway or exposure media |
| *RO* | average annual surface runoff from pervious areas |
| *Rp* | interception fraction of the edible portion of plant for aboveground produce |
| $\rho_w$ | density of liquid water at sweat lodge temperature |
| *s* | estimate of standard deviation |
| *SA* | body surface area available for contact |
| *Se* | shielding factor |
| *SFr* | soil (or sediment) fraction is the ratio of the soil (or sediment) ingestion rate to the sum of the plant and animal ingestion rates |
| *SPv* | plant bioconcentration factor, used to calculate the transfer of a chemical from soil to a soil-dwelling plant |
| *t* | student t-test value for data set |
| $T_1$ | the time at the start of exposure |
| $T_2$ | the time at the end of exposure |
| *tD* | time period over which deposition occurs (time period of WTP operation) |
| *TEF* | the ratio of toxicity of a polychlorinated dibenzodioxin, dibenzofuran, or biphenyl COPC to the toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin or the ratio of toxicity of a polyaromatic hydrocarbon COPC to that of benzo(a)pyrene |
| *TOE* | TOE factor: total organic emissions divided by the sum of the identified organics |
| $TO_{TOTAL}^{GRAV}$ | total nonvolatile organic emission |
| $TO_{TOTAL}^{SVOC}$ | total volatile organic emission |
| $TO_{TOTAL}^{SVOC}$ | total semivolatile organic emission |
| *Tp* | length of plant exposure to deposition per harvest of edible portion of plant for aboveground produce |
| *TRV* | toxicity reference values: a threshold value used to characterize ecological risk based upon a hierarchy of toxicological data |

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| $T_{sl}$ | temperature of the sweat lodge |
| *TSS* | total suspended solids concentration |
| $T_{wk}$ | water body temperature |
| $T_{wk}$ | water body temperature |
| *u* | current velocity |
| *UCL* | upper confidence limit |
| *UF* | uncertainty factor |
| *URF* | inhalation unit risk factor |
| $Vf_x$ | average annual volumetric flow rate through the water body |
| $VG_{ag}$ | empirical correction factor for the aboveground plant due to air-to-plant transfer |
| $VG_{rootveg}$ | empirical correction factor for belowground plants |
| $V_W$ | volume of water in the sweat lodge |
| *WP* | aquatic plant concentration factor, used to calculate the transfer of a chemical from surface water to an aquatic plant |
| *x* | arithmetic mean (average) of stack gas concentrations |
| *Yp* | site-specific and plant-type-specific yield or standing crop biomass of the edible portion of the plant for aboveground produce (productivity) |
| $Z_s$ | soil mixing zone depth (*s* = depth in centimeters) |
| $\theta$ | temperature correction factor |
| $\theta_{bs}$ | bed sediment porosity |
| $\theta_{sw}$ | soil volumetric water content |

# Executive Summary

The purpose of this work plan is to provide the concepts, methods, and data to be used in an environmental risk assessment. The intent of this environmental risk assessment is to evaluate the human health and ecological risk from the potential airborne emissions. It is important that people and the environment are not harmed because potential exposures are overlooked or underestimated, but it is also important to maximize the ability of the facility to dispose of the tank waste and to protect against the potential leakage from these tanks into the nearby Hanford site soil, groundwater, and ultimately, the Columbia River. A balance of these goals will result from the interactive process of reviewing and improving this work plan and subsequent documents that will contain the actual environmental risk assessments. Indeed, each stage of this work will benefit from interactions with regulatory agencies, American Indian tribes, and the public to assure public health and to protect the environment. These interactions are expected to be in the form of questions and comments about methods and data, and other inputs.

Hanford tank wastes consist of approximately 54 million US gallons of highly radioactive and mixed dangerous wastes that are managed by the US Department of Energy. The wastes consist of solids (sludge), liquids (supernatant), and salt cake (dried salts that will dissolve in water to form supernatant). The term low-activity waste (LAW) generally refers to the supernatant portion, while high-level waste (HLW) usually refers to the solids. These wastes are stored in underground holding tanks and will be pumped to the WTP. At the WTP, wastes will be pretreated and immobilized using a technology called *vitrification*. Vitrification is a thermal process that converts the waste materials into durable glass. The vitrified wastes and secondary wastes resulting from the WTP processes will then be transferred to designated treatment, storage, or disposal units. The WTP is scheduled to be in operation for up to 40 years. During the pretreatment and vitrification of the various types of wastes, some airborne emissions will be created. Various engineered devices will control the nature and amounts of these emissions, but there will still be material in the form of vapors and small particles that will be released via three tall stacks into the environment around the WTP.

Once the vapors and particulates leave the facility stacks, they will be carried by air currents and deposited on the surface of soil and vegetation around the WTP and on the surface of the Columbia River. An air-dispersion model named CALPUFF will be used to calculate how the emitted chemicals and radionuclides will be dispersed. Some of the material will enter terrestrial and aquatic food chains, and people and animals can ingest the food that contains small amounts of material from the emissions. The work plan contains details about these processes; pathways and exposures are defined in very explicit ways so that a complete and quantitative risk assessment can be conducted. The work plan presents a thorough explanation of these exposures via various pathways to a variety of receptors for over 400 different chemicals and radionuclides.

The environmental risk assessment will define and evaluate risks, or the potential for harm, to human and ecological receptors within various distances from the WTP. For example, the air-dispersion model will model exposure depositions and concentrations within 50 kilometers of the WTP. The area within 50 kilometers is predominantly located within Benton County in Washington State, and includes parts of Franklin, Grant, Yakima, and Kittitas counties. The Tri-Cities, composed of the cities of Richland, Kennewick, and Pasco, are adjacent to the southern edge of the Hanford Site. The Tri-Cities area

contains a population of approximately 220,000, the majority of whom reside between 30 kilometers and 50 kilometers from the WTP Site (Census Bureau 2009[1]). There are no permanent residents on the Hanford Site, but there are workers. American Indian tribes have treaty rights to resources on the Hanford Site, and the environmental risk assessment will evaluate potential risks from food gathering and social activities. A variety of ecological receptors inhabit the Hanford Site. They include terrestrial and aquatic plants (the basis of the food chains); terrestrial, aquatic, and sediment-dwelling animals; mammals and birds that eat the terrestrial and aquatic plants and animals; and aquatic biota in the Columbia River. Thus, Hanford Site-specific human and ecological receptors will be evaluated in the risk assessments. There will be two types of risk assessments: one focusing on humans (the human health risk assessment) and the other focusing on plants and animals in the environment (the ecological risk assessment).

The human health risk assessment includes four fundamental steps: (1) data evaluation, (2) exposure assessment, (3) toxicity assessment, and (4) risk characterization. These steps, as well as the collection of considerable amounts of data and associated estimation methods, are specified by the Washington State Department of Ecology and the US Environmental Protection Agency. The data evaluation step focuses on the selection of the chemicals and radionuclides of potential concern and the quantification of emissions; both of these are described fully in the work plan. Exposure assessment, the second step, deals with estimating the type, extent, and magnitude of potential exposures. The types of human receptors that will be used to calculate quantitative estimates of risk are also established at this step. These receptors are the following: worker, resident (both adult and child), resident subsistence farmer (both adult and child), resident subsistence fisher (both adult and child), American Indian subsistence resident (both adult and child), nursing infant of all adult receptors, and a person who has an acute or short-term exposure. The geographical locations where the people live and work and the exposure pathways are explained in the work plan. The third step is a toxicity assessment, which involves evaluating the potential of the various chemicals and radionuclides to cause adverse health effects in exposed individuals. The toxicity assessment will consider the potential cancer and noncancer effects associated with contaminant exposures. Risk characterization, the fourth step, involves evaluating the exposure and the toxicity information together to estimate the potential for various humans under various conditions to experience adverse effects (cancer and noncancer) as a result of being exposed to the media contaminated by emissions from the WTP. Risks are presented as potential incremental lifetime cancer risk, or noncancer hazard quotients and hazard indices. The information will be presented for each chemical and radionuclide, each pathway, each set of exposures, and each receptor. In turn, these risk values will be compared to risk thresholds. Thus, various comparisons will provide information in order to better understand and make decisions about the protection of human health.

The ecological risk assessment includes the same fundamental steps as the human health risk assessment, although the first step is called problem formulation instead of data evaluation. As described above for the human health risk assessment, these four steps follow a logical order, with additional methodical substeps. Just as is the case for human health risk methods and data, the methods and the data for the ecological risk assessment have been specified by regulatory agencies such as the Washington State Department of Ecology and the US Environmental Protection Agency. As in the case of human health, where Hanford Site-specific human receptors are being evaluated, Hanford Site-specific vegetation and animals are also being evaluated. These receptors are organized into two types according to the habitat type in which they live: (1) the land or terrestrial habitats around the WTP site, and (2) the aquatic

[1] Census Bureau. 2009. "Washington QuickLinks, Annual Estimates of the Resident Population for Incorporated Places in Washington", listed alphabetically: April 1, 2000 to July 1, 2008, release date July 1, 2009, US Census Bureau, Systems Support Division. Available at http://quickfacts.census.gov/qfd/states/53000lk.html.

habitats of the Columbia River. For the terrestrial habitats, the following receptors will be used to quantify potential risk: plants, soil invertebrates, herbivorous mammals and birds, omnivorous mammals and birds, and carnivorous mammals and birds. For the Columbia River, the following aquatic receptors will be used: plants, sediment-dwelling invertebrates, fish (including salmon) and other aquatic organisms, herbivorous waterfowl, shorebirds, and fish-eating mammals and birds. There is abundant information about the feeding habits of these organisms; there is also considerable toxicity data. A quantitative characterization will be provided for a variety of chemicals and radionuclides, assessing many pathways in a variety of geographical places and many exposures to a variety of ecological receptors. The ecological risk assessment calculates exposure and effects ratios. These ratios, called hazard quotients and hazard indices, are in turn compared to thresholds. This information will assist users in making informed decisions about the protection of the environment.

Various types and degrees of uncertainty are introduced into the human health and ecological risk assessments at every step of the process. This uncertainty occurs because risk assessment is a complex process, requiring integration of source information, estimates of fate and transport in variable environments, exposure assessment, and effects assessment. Uncertainty is inherent even when the most accurate, up-to-date, and appropriate models are used. Throughout the risk assessments, an effort is made to overestimate, rather than underestimate, the exposures and risks in order to compensate for these uncertainties. The work plan explains how an uncertainty assessment will be used to place the risk estimates in proper perspective to allow informed risk management decisions.

In summary, chemical and radionuclide contaminants present in underground tanks at the Hanford Site need to be retrieved and treated before they leak into the nearby soil and groundwater, and possibly into the Columbia River. The WTP processes to pretreat and vitrify the contents of underground tanks will help to solve this potential problem. Emissions are expected from these waste treatment processes, and this work plan shows the models and scientific data that will be used to characterize how separate chemicals and radionuclides may move through the air, soil, surface water, sediment, and food chains around the WTP in the Hanford Site environment. These airborne releases could potentially expose a variety of human and ecological receptors to chemicals and radionuclides.

This work plan will benefit from inputs from regulatory agencies, American Indian tribes, and the public. After inputs are incorporated, the work plan methods and data will be implemented. Computations will follow, and risk predictions will be compared to appropriate thresholds. These findings will be put into proper perspective using an uncertainty assessment to allow fully informed risk management decisions. These decisions will focus on protecting human health, plants, and animals while operating the WTP successfully.

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# 1 Introduction

This risk assessment work plan (RAWP) presents the risk assessment protocol for evaluating potential risks to human health and ecological resources from the Hanford Tank Waste Treatment and Immobilization Plant (WTP) at the Hanford Site. This RAWP is required to satisfy, in part, requirements established by condition III.10.C.11 of the WTP Dangerous Waste Permit (WA7890008967, herein referred to as the DWP), as well as addressing provisions in Washington Administrative Code (WAC) 173-303-680 and applicable Code of Federal Regulations (CFR) incorporated by reference.

The Hanford Site is located in southeastern Washington State, is owned by the US government, and is managed by the US Department of Energy (DOE), US Bureau of Land Management (BLM), US Fish and Wildlife Service (FWS), and Washington State Department of Fish and Wildlife. The WTP will include two waste vitrification facilities and a pretreatment facility, and will be built in the 200 East Area of the Hanford Site.

This work plan establishes the methods for conducting the screening-level risk assessment (SLRA) to estimate potential risks to human health and ecological resources associated with airborne releases resulting from processing Hanford tank waste into a stable, glassified form. Airborne releases are the only viable pathway for receptor exposure; therefore, the US Environmental Protection Agency (EPA) guidance for airborne releases is being used (see Section 2). Other releases, such as releases to water and non-dangerous solid waste disposal, are permitted through appropriate regulatory programs. Throughout the risk assessment process, the intent is to provide data to help assess the potential impacts to people living or working on or near the Hanford Site as well as for plants and animals.

The risk assessment, in conjunction with the other portions of the DWP will serve to:

- Establish operating conditions for the facilities
- Identify feed constituents that need to be controlled to stay below acceptable risk thresholds
- Identify monitoring of WTP components required to verify permit compliance

The limits and monitoring requirements established as a result of the risk assessment process are not the only inputs required for control and operation of the WTP. Other inputs will include the following:

- Equipment control limits and monitoring established as a result of experience with operations from similar DOE vitrification facilities, including the West Valley Demonstration Project in West Valley, New York, and the Defense Waste Processing Facility at the Savannah River Site in Aiken, South Carolina
- Control limits and monitoring recommendations of equipment vendors
- Control limits and monitoring required by other permits, approvals, and authorizations (e.g., air permits)

This RAWP contains a brief statement of the risk assessment approach (see Section 2) and a description of the WTP (see Section 3). Sections 4 through 8 present the key components of the human health and ecological SLRA protocol, as noted below:

- Section 4: Identification of constituents of potential concern
- Section 5: Quantification of airborne emissions
- Section 6: Modeling of the airborne emissions and other environmental pathways
- Section 7: Screening human health risk assessment
- Section 8: Screening ecological risk assessment

Section 9 presents the relationship of the risk assessment to the WTP, including the process to establish risk-based emissions limits, if needed. Section 10 describes how uncertainty will be handled in the SLRA. References are provided in Section 11 and are followed by four supplemental reports providing details of the constituents of potential concern (Supplement 1), details of the emissions estimate (Supplement 2), details of the WTP process cell emissions (Supplement 3), and chemical-specific physical/chemical and toxicity data for human health and ecological resources (Supplement 4). The fifth supplement (Supplement 5) provides information on air dispersion modeling. The public, American Indian tribes, and regulatory agencies are being invited to comment on this work plan and on subsequent documents to obtain their input to the decision-making process.

## 1.1 References

WAC 173-303-680. *Miscellaneous units*, Washington Administrative Code, effective 01 January 2005.

WA7890008967. *Hanford Facility Resource Conservation and Recovery Act (RCRA) Permit, Dangerous Waste Portion for the Treatment, Storage, and Disposal of Dangerous Waste, Part III, Operating Unit 10, (Waste Treatment and Immobilization Plant).*

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# 2 Risk Assessment Approach

This section describes the overall SLRA scope and approach (shown in Figure 2-1) that will be used to establish operating conditions for cold commissioning (nonradioactive waste testing), as well as processing of mixed wastes at the WTP.

The primary regulatory guidance followed for this risk assessment is found in the *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA 2005) and the *Screening-Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA 1999).

## 2.1 Scope of the Screening-Level Risk Assessment

The SLRA will evaluate exposure and risks to potential human and ecological receptors within a 50-km radius of the WTP. Section 7 includes additional discussion of the human receptors, and Section 8 provides additional details of the ecological receptors.

The area within the 50-km radius of the WTP is located predominantly within Benton County in Washington State, with smaller portions located in Franklin, Grant, Yakima, and Kittitas counties. The Tri-Cities (i.e., the combined cities of Richland, Kennewick, and Pasco) are adjacent to the southern edge of the Hanford Site. The Tri-Cities area contains a population of approximately 220,000 (US Census 2009), the majority of which reside between 30 km and 50 km from the WTP site. The population outside the Tri-Cities, but within 50 km of the WTP site, is sparse. There are no permanent residences on the Hanford Site. American Indian tribes have treaty rights to resources on the Hanford Site, and the SLRA includes potential risks from food gathering and other tribal-specific activities, as well as from inhalation and external exposures to WTP emissions (for more information, see Section 7.1).

A variety of ecological receptors inhabit the Hanford Site. They include terrestrial and aquatic plants; terrestrial, aquatic, and sediment-dwelling invertebrates; mammals and birds that eat terrestrial plants and animals; fish and other aquatic biota; and mammals and birds that eat fish and other aquatic biota. These ecological receptors are discussed in more detail in Section 8.1.

The SLRA (specifically, the pre-demonstration test risk assessment [PRA] and the final risk assessment [FRA]) will address the potential operating life of the WTP. The current WTP DWP (WA7890008967) covers projected operations of the WTP. The SLRA assumes that the facility will operate at maximum capacity for its entire design life (40 years from the start of the facility operations). Risks from the waste in the Hanford double-shell tank system, as well as cumulative risks from the Hanford Site, are outside the scope of the SLRA.

## 2.2 Screening-Level Risk Assessment Process

The major components of the SLRA process for airborne emissions are as follows (Figure 2-1):

- **Work plan for the SLRA** – This work plan is submitted to comply with conditions of the DWP. The work plan establishes the methods for the future implementation of the SLRA. The PRA and FRA are subparts of the SLRA, as described in this work plan.

- **Pre-demonstration test risk assessment** – The PRA will be performed before performance-demonstration testing of the WTP. The PRA will estimate human health and ecological risk based on engineering estimates of emissions from WTP units.
- **Final risk assessment** – The FRA will be conducted following collection of data from performance demonstration testing of WTP units. The FRA is conducted using an approach similar to the PRA. However, estimated emission rates will be supplemented with the results of the environmental performance demonstration tests.

Participants in the SLRA process are:

- DOE
- Washington State Department of Ecology (Ecology)
- EPA, Region 10
- Yakima Indian Nation (YN)
- Confederated Tribes of the Umatilla Indian Reservation (CTUIR)
- Nez Perce Tribe
- Hanford Advisory Board (HAB)
- The general public
- Bechtel National, Inc.

All participants are welcome to make contributions to the development of this work plan.

The SLRA must serve several purposes, including (1) identifying any potential risks to human health or ecological resources that may result from emissions from the WTP; (2) providing the information necessary to determine what, if any, additional permit conditions are necessary for the operation of the WTP to be protective of human health and ecological resources; and (3) providing risk information to Ecology, EPA, DOE, American Indian tribes, and the public. For these reasons, the overall approach for the SLRA is to identify potential risks associated with both plausible and worst-case scenarios as defined in the following:

- The plausible exposure scenarios represent more realistic assumptions regarding the location of potential human and ecological receptors. The exposure scenarios reflect anticipated WTP operations and the continuation of current uses of the surrounding land and habitats, and make reasonable assumptions about future land uses while still using upper-bound estimates of exposure pathways and activity patterns.
- The worst-case exposure scenarios represent worst-case assumptions regarding the location of human and ecological receptors, exposure pathways, and activity patterns (e.g., subsistence fishing). The receptor locations used in the worst-case scenario are considered hypothetical, because assumed activities (e.g., residence, subsistence farming) do not currently occur in the worst-case Hanford Site locations nor are they expected. These hypothetical worst-case scenarios are discussed in the uncertainty assessment (Section 10).

The exposure scenarios are intended to provide a better understanding of the range of potential risks to a variety of human and ecological receptors representing conservative exposures at locations typical of the Hanford Site area under a variety of land use conditions, both now and in the future.

Both the plausible and worst-case exposure scenarios will incorporate conservative assumptions regarding human and ecological exposures. This approach is consistent with the *EPA Risk Characterization Program* (EPA 1995), which directs the "use of several descriptors, rather than a single description, to enable the EPA to present a fuller picture of risk that corresponds to the range of different exposure conditions encountered by various individuals and populations."

The general technical process for the SLRA is provided in Figure 2-2. This process starts with the estimation of air concentration of various chemicals and radionuclides, moves to an estimation of airborne deposition, and continues to predictions of movement in soil, surface water, and food. Next, exposure to humans, plants, and animals will be estimated to complete the risk characterization.

Requirements and assumptions for the FRA will be influenced by the results of the PRA, as well as from data collected during environmental performance demonstration tests. The FRA will include estimated emissions based on engineering calculations (pretreatment system emissions and vapor-phase organic emissions from WTP process cells) and environmental performance demonstration tests for the low-activity waste (LAW) and high-level waste (HLW) vitrification systems. Based on the results of the environmental performance demonstration tests, the FRA may involve running new models, modeling additional chemicals, or changing model parameters. Information that will require updating in the FRA, as specified in the DWP, will include:

- Toxicity data current at the time of the submittal
- Compounds newly identified, or updated emissions-data from current waste characterization and emission testing
- Air modeling updated to include stack-gas parameters based on most current emissions testing and current WTP unit design
- Physical/transport properties of constituents current at the time of the submittal
- Process description based on current WTP unit design
- Emissions data and all supporting calculations based on current WTP unit design
- Update of receptor locations based on land use or land use zoning changes, if any

The performance demonstration testing of melter units presents unique challenges that differ from incineration-type combustion units, which are used as a starting point for developing test plans. Comparisons are made between the systems used to control melter emissions and those used for flame combustion units, as well as comparisons to the quantities and concentrations of constituents of potential concern (COPC) fed to melter units to those of other, flame-type combustion units. For the performance demonstration test to be predictive of the melter-offgas system's ability to control emissions and demonstrate that human health and environmental protection standards established by the SLRA are met, it will be necessary to take these differences into account.

The SLRA process is iterative. It includes review of the PRA findings and revision of risk assessment assumptions, WTP engineering design, and operation of the FRA. Results of the PRA and FRA will be used to confirm that the emissions from WTP do not pose an unacceptable risk to human health or the environment. If unacceptable risks are identified, they can be mitigated through operating limits. Input from Ecology, EPA, American Indian tribes, and the public will be included at each step of the process. The graphic description of the process provided in Figure 2-1 identifies points for this input.

The PRA modeling results will be used to formulate FRA approaches; thus, the PRA is an important first step and the primary emphasis of this work plan.

## 2.3 References

Census Bureau. 2009. "Washington QuickLinks, Population Estimates: Places in Washington listed alphabetically: Population Estimates for July 1, 2008", release date July 1, 2009, US Census Bureau, Systems Support Division, accessed January 6, 2010. Available at http://quickfacts.census.gov/qfd/states/53000lk.html.

EPA. 1995. *EPA Risk Characterization Program*, memorandum from Carol Browner, Administrator, to EPA staff, Office of the Administrator, 21 March 1995. US Environmental Protection Agency, Washington, DC.

EPA. 1999. *Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, Peer Review Draft, EPA 530-D-99-001A. Office of Solid Waste and Emergency Response, US Environmental Protection Agency, Washington, DC.

EPA. 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, Final, EPA/530/R-05/006. US Environmental Protection Agency, Washington, DC.

WA7890008967. *Hanford Facility Resource Conservation and Recovery Act (RCRA) Permit, Dangerous Waste Portion for the Treatment, Storage, and Disposal of Dangerous Waste, Part III, Operating Unit 10, (Waste Treatment and Immobilization Plant).*

**Figure 2-1 Overview of Screening-Level Risk Assessment Process for WTP Air Emissions**

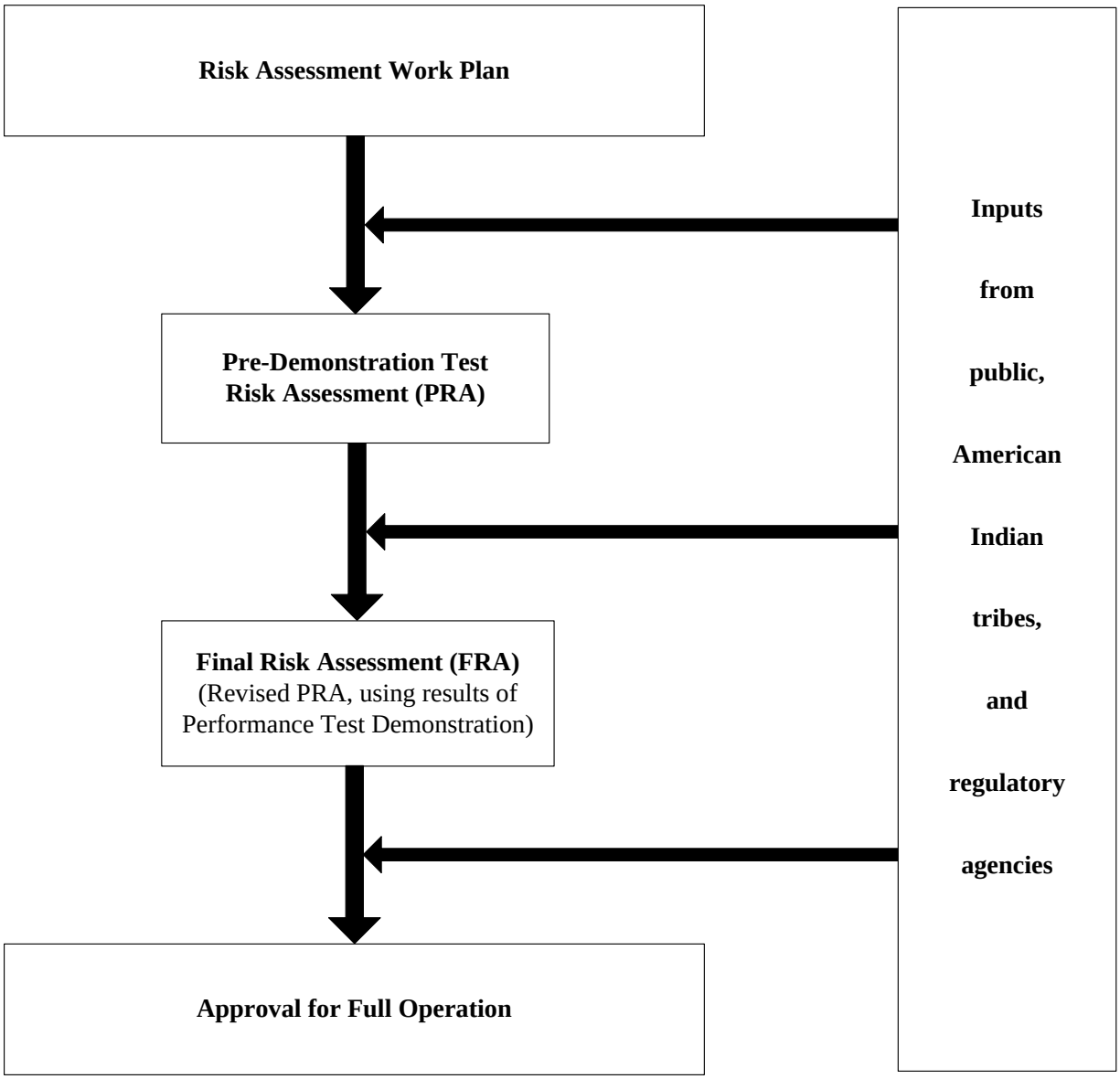
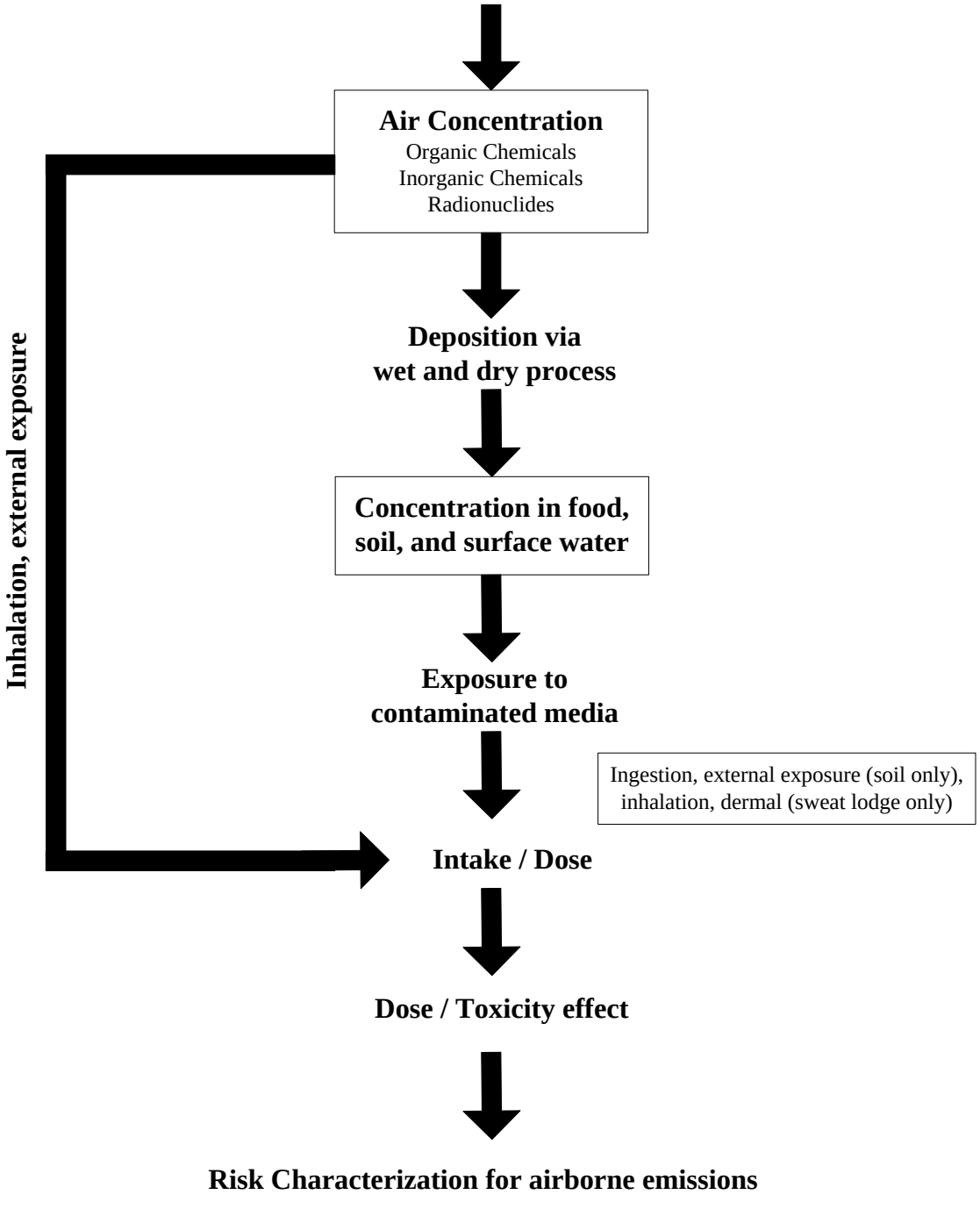


**Figure 2-2 Overview of Fate and Transport of Airborne Emissions During the PRA and FRA for the WTP**



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# 3 WTP Facility/Process Description

Hanford tank waste consists of approximately 54 million US gallons of highly radioactive and mixed hazardous wastes stored in underground storage tanks at DOE's Hanford Site. The WTP is being constructed to treat mixed wastes from underground storage tanks. After the tank waste is received from the Hanford double-shell tank system, it will be pretreated and then immobilized using a process called *vitrification*. Vitrification is a thermal process that converts the waste materials into a durable glass. The vitrified wastes and secondary wastes resulting from the WTP processes will be transferred to permitted treatment, storage, or disposal units for disposition. Offgas generated by the pretreatment and vitrification processes will be treated in independent offgas treatment systems. This section provides an overview of the mixed waste treatment processes that will be used in the WTP. Readers should consult Chapter 4 of the Hanford Facility Resource Conservation and Recovery Act (RCRA) Permit, Part III, Operating Unit Group 10, (permit number WA7890008967) for additional details on WTP engineering and waste treatment processes.

## 3.1 WTP Overview

The WTP is located at the eastern end of the 200 East Area of the Hanford Site, near the former Grout Treatment Facility, 241-AP Tank Farm Complex, and Plutonium Uranium Extraction Plant. Figure 3-1 shows the location of the WTP on the Hanford Site.

Waste from the Hanford double-shell tank system will be transferred to the WTP Pretreatment (PT) Facility. The tank waste consists of solids (sludge), liquids (supernatant), and saltcake (dried salts that will dissolve in water to form supernatant).

The term low-activity waste (LAW) feed generally refers to the supernatant portion of Hanford's double-shell tank waste, although it can include high-level waste (HLW) solids. Hanford tank waste is from a variety of nuclear process facility sources. It historically has been managed as HLW. Since the Hanford project began, different chemical processes have been used to separate plutonium from spent fuel and targets received from many different nuclear reactors. Many waste components were added to the tanks or blended together as part of an in-tank process. In addition, some of the wastes were later reprocessed, resulting in significant blending of the wastes. Evaporators were used to reduce the waste volume. Waste was also left in a waste tank at high temperature which impacts the minerals that formed from these wastes (Agnew and others 1997).

As a result of these previous waste processing activities, the tanks are filled with millions of gallons of waste as sludge, saltcake, and aqueous supernatant. Most of the saltcakes' contents are expected to be dissolved when the waste delivery processing occurs in the tank farms to prepare the waste for sluicing to WTP (Kirkbride and others 2007). The sludges consist primarily of oxides, hydroxides, or silicates of Al, Fe, Cr, Bi, Ni, U, Cd, Zr, and many more trace species and radionuclides (24590-101-TSA-W000-0004-114-00021). The supernatants are primarily sodium salts of $NO_3^-$, $NO_2^-$, $OH^-$, $CO_3^{-2}$, $Al(OH)_4^-$, $PO_4^{-3}$, $SO_4^{-2}$, $F^-$, $Cl^-$, $CrO_4^{-2}$, along with potassium and other trace species (Kirkbride and others 2007). Both sludge and saltcake contain large amounts of oxalate ($C_2O_4^{-2}$) and many other organics, including solvents and chelates (24590-WTP-RPT-RT-07-002).

A recent study investigated the feed composition estimate (called a feed vector) and grouped the feeds into 13 groups based on which element or elements in the feed, after WTP pretreatment, will limit the

waste concentration in immobilized HLW (IHLW) glass (24590-WTP-RPT-PE-07-001). The 13 groups are named as follows:

| | | |
|---|---|---|
| aluminum low-leach | iron-aluminum | calcium |
| aluminum high-leach | phosphate | uranium |
| chromium | iron | sulfate |
| bismuth | zirconium-aluminum | |
| fluoride | sodium | |

In addition to these 13 groups, there is the LAW group, with low undissolved solids concentration, and the lead (Pb) group, that were impacting a single feed batch (24590-WTP-RPT-PE-07-001).

Figure 3-2 shows a layout of the WTP. Three main process buildings (PT Facility, HLW Facility, and LAW Facility) will contain most of the dangerous waste management operations and include major areas for pretreating and vitrifying (immobilizing) tank waste. The PT Facility will receive and pretreat the waste before vitrification. Two separate vitrification facilities will be used to immobilize the pretreated waste. The LAW Facility will immobilize the majority of the supernatant and dissolved saltcake from the Hanford tank waste. The HLW Facility will immobilize the HLW fraction of the Hanford tank waste. Other smaller support buildings will provide for storage or transfer of materials used in the treatment process and for storage of wastes.

Figure 3-3 provides a simplified diagram of the WTP processes. Mixed wastes from the double-shell tank system (shown in the lower left corner of the diagram) will be received and processed through the various WTP pretreatment operations (including feed evaporation, ultrafiltration, and ion exchange). The resultant pretreated wastes will, in turn, be fed to the LAW or HLW vitrification systems. The treatment of offgas from the pretreatment and vitrification processes will result in point-source emissions to the environment from each of the three processing facility stacks. In actuality, each facility stack is a bundle of individual flues (or pipes) from that facility that are supported together inside a structural steel lattice (see Figure 3-4). The individual flues separately exhaust process and building ventilation from the facility. Figure 3-5 depicts the process flues that correspond to each of the three main process buildings. For example, the PT Facility has flues that exhaust the process and vessel ventilation system, the reverse flow diverter/pulse jet mixer system, and the process cell emissions. Additional nonprocess-related flues are not shown on Figure 3-5 because they do not contribute COPCs or radionuclides of potential concern (ROPC) emissions to the environment. The nonprocess flues do, however, influence contaminant dispersion profiles by virtue of their effect upon parameters important to air dispersion modeling (e.g., gas flow rate, velocity, and temperature). Flue parameters relevant to air dispersion modeling are presented in *Hanford Tank Waste Treatment and Immobilization Plant Risk Assessment Air Quality Modeling Protocol* (24590-WTP-RPT-ENV-08-001), Supplement 5 of this document.

## 3.2 Pretreatment Overview

The LAW supernatant will be transferred to the WTP PT Facility as solutions that contain some undissolved solids (HLW waste or LAW-precipitated salts). The HLW feed will be transferred as slurry to the WTP PT Facility.

Wastes having sodium molarity less than 5 will be received into the PT Facility and concentrated in the waste feed evaporator. Wastes having a sodium molarity greater than or equal to 5 will bypass the waste feed evaporator. Once the sodium molarity is acceptable for further processing (either as-received or after evaporation), the waste will go through the following processes:

- LAW feeds will be blended with HLW feeds in an ultrafilter preparation tank. The ratio of LAW to HLW undissolved solids will be established to support the respective glass production rates. The blended HLW and LAW feed streams will undergo a filtration process that separates LAW liquid stream (permeate) from the slurry. The LAW permeate will then be processed through the ion exchange (IX) process discussed below. The concentrated solids slurry will be caustic leached (if warranted), washed, and blended with cesium concentrate from the IX and strontium/transuranic (TRU) solids from $^{90}$Sr/TRU precipitation (see below), before being transferred to the HLW Facility.
- Some feeds will contain organic complexants that cause the Sr and some TRU waste to remain in solution. This waste will undergo a $^{90}$Sr/TRU precipitation process before filtration. The filtration step will then separate the $^{90}$Sr/TRU solids, manganese oxide solids (a by-product from the precipitation process), and entrained solids from permeate (LAW stream). The $^{90}$Sr/TRU precipitate will be washed and stored for blending with HLW feed before HLW vitrification. The $^{90}$Sr/TRU precipitate (solids) will not be caustic leached. Filtration permeates are processed through the IX processes.
- After filtration, the permeate will undergo IX to remove $^{137}$Cs. The $^{137}$Cs eluate will be concentrated by evaporation; the concentrated eluate will then be blended with pretreated HLW solids before transfer to the HLW Facility. The last step in the pretreatment process is to concentrate the treated LAW liquid by evaporation before transferring the waste to the LAW Facility.

The PT Facility will also contain an offgas treatment system designed to abate emissions from the pretreatment processes. The offgas treatment system consists of several control devices:

- Caustic scrubber
- High-efficiency mist eliminators (HEME)
- Primary high-efficiency particulate air (HEPA) Filters
- Secondary HEPA filters
- Thermal catalytic oxidizer unit
- After-cooler
- Carbon bed adsorbers
- Adsorber outlet filter
- Demisters
- Electric heaters
- Primary HEPA filters
- Secondary HEPA filters
- Exhaust fans
- Stacks/exhaust flues

See the DWP, Chapter 4 for details.

Liquid effluents will be either recycled back into the facility or sent to the Hanford Site Liquid Effluent Retention Facility or 200 Area Effluent Treatment Facility.

## 3.3 LAW Vitrification

Treated supernatants from the PT Facility will be transferred to the LAW Facility for processing. The LAW vitrification process will consist of two melter systems operated in parallel. Each melter system has a set of feed preparation vessels, a large-capacity joule-heated ceramic melter, and an offgas treatment system. The facility will also have a secondary offgas system shared by the two melter systems. The following description applies to each of the two LAW melter systems.

Pretreated LAW waste feeds will be received into one of two LAW concentrate receipt vessels inside the LAW Facility. Batches of concentrated LAW feed will be transferred from these vessels to feed preparation vessels, where glass formers and sucrose will be added and blended to form a uniform batch of feed to the LAW melters. The slurry feed will be transferred to the melter feed vessels, where it is fed continuously to the LAW melters.

Each LAW melter is designed to nominally produce 15 metric tons per day of immobilized LAW (ILAW) glass and operate at an approximate temperature between 950 °C and 1150 °C. The feed will enter the melter from the top and form a cold cap above the melt pool. Volatile components in the feed will be evaporated or decomposed, then drawn off through the melter offgas system. Nonvolatile components will react to form oxides or other compounds dissolved in the glass matrix. Bubblers will agitate the mixture to increase the glass production rate. An airlift system will pour the glass from the melter into stainless steel containers.

The LAW melter offgas system consists of the following control devices:

- Primary and secondary film coolers, one set for each melter
- Submerged bed scrubbers (SBS)
- Wet electrostatic precipitators (WESP)
- Electric heaters
- HEPA filters
- Carbon adsorber
- Selective catalytic oxidizer
- Selective catalytic reduction (SCR) unit
- Heat exchanger
- Caustic scrubber
- Melter offgas exhausters
- Stacks/exhaust flues

Each LAW melter system will have its own primary offgas equipment, including a film cooler, SBS, and WESP. Particulates and condensables, including entrained or volatilized radionuclides in the melter offgas stream, will be captured in the SBS and WESP. Condensables from the SBS and the WESP will be collected in the liquid effluent system and recycled to the treated LAW evaporator in the pretreatment facility. The primary offgas systems will join after the WESP and will be routed to the secondary offgas system. At this point, the LAW vessel vent header will join the offgas. The secondary offgas system will provide final filtration, remove mercury, destroy organics, reduce oxides of nitrogen ($NO_x$), and remove

halides. This will be done by using HEPA filters, a treated activated carbon bed, a thermal catalytic oxidizer, a selective catalytic reducer, and a caustic scrubber.

See the DWP, Chapter 4 for details.

## 3.4 HLW Vitrification

The HLW Facility will receive the pretreated HLW feed from the PT Facility. Treated HLW slurries and the LAW intermediate waste products (separated $^{90}$Sr/TRU and $^{137}$Cs) will make up the feed to the HLW Facility. The HLW vitrification process will consist of two joule-heated ceramic melters fed by independent feed and blending vessel trains, a dedicated offgas treatment system for each melter, and a common secondary effluent collection system. The HLW feed concentrate will be transferred from the PT Facility to one of two melter feed preparation vessels in the HLW Facility. The feed concentrate will be blended with glass-forming chemicals and sucrose, and then mixed to ensure a uniform mixture. The melter feed slurry will be transferred to the melter feed vessel, where it will be fed to a dedicated HLW melter.

Each of the two HLW melters is designed to operate at an approximate temperature between 950 °C and 1150 °C at nominal rates of 3.0 to 3.75 metric tons per day of IHLW glass. Melter feed slurry will be introduced at the top of the melter and form a cold cap on the surface of the melt pool. Water and volatile components will evaporate or decompose and then be drawn off through the offgas system. Nonvolatile components will react to form oxides, which will become part of the molten glass. Bubblers will agitate the mixture to increase the glass production rate.

Each HLW melter will have dedicated primary and secondary offgas systems where the offgas from the melter will pass through a film cooler, SBS, WESP, HEMEs, and HEPA filters to remove particulates and radionuclides. The offgas will then pass through a secondary offgas system consisting of treated activated carbon, silver mordenite, thermal catalytic oxidation, and SCR. This secondary system will remove mercury and halides, destroy organics, and reduce $NO_x$.

An airlift system inside the melter will pour molten HLW glass into stainless steel canisters. The filled canister will then be inspected, the glass sampled if necessary, and the canister sealed. The canisters from the two melters will be decontaminated by a nitric acid/cerium ($HNO_3/Ce^{+4}$) chemical milling process that dissolves a thin layer of the canister outer wall material. Canister decontamination waste effluents will be recycled to the PT Facility.

The purpose of the HLW primary offgas treatment system is to cool the melter offgas and to remove offgas aerosols and particulates generated by the melter and from the vessel ventilation air. This treatment system consists of the following:

- Film coolers
- SBSs
- SBS condensate receiver vessels
- WESPs
- HEMEs
- Electric heaters
- HEPA filters

- Activated-carbon columns
- Silver-mordenite columns
- Thermal catalytic organic oxidizers
- $NO_x$ SCR units
- Heat exchangers
- Booster fans
- Stack fans
- Stacks/exhaust flues

See the DWP, Chapter 4 for details.

## 3.5 Stacks and Flues

The PT, LAW, and HLW Facilities will each have separate stacks from which the treated emissions derived from process operations and other sources will be released to the environment. The stacks will house a bundle of individual emission units (flues) that are associated with their respective sources. Thus, each of the three facilities will have one stack only. Additional information regarding the individual flues and their corresponding offgas parameters (e.g., flow rate, velocity, and temperature) is included in Supplement 5.

In addition to the process offgas system, building ventilation systems will be incorporated into each of the processing plants. Treated building ventilation systems will also be vented to the atmosphere through dedicated flues. Figure 3-5 presents a simplified graphic of the expected emission sources and the associated flues. Those flues whose emissions are estimated using the steady-state flowsheet model also include the unique stream identifiers (e.g., HOP-31) that are shown in Figure 3-3 and discussed further in Supplement 2.

The offgases associated with pretreatment processes will be exhausted through the PT Facility stack via flues PT-S3 and PT-S4. Flue PT-S3 is used to discharge the PT offgas and vessel ventilation emissions, and PT-S4 is used to discharge PT pulse jet ventilation emissions. The emissions associated with potential leaks to processing cells will be discharged through a third flue within the PT stack identified as PT-S2. The treated offgases associated with LAW vitrification processes will be discharged through the LAW vitrification stack via flue LV-S3. The emissions associated with leaks to the LAW vitrification process cells will be discharged through flue LV-S2. The treated offgases associated with the two HLW melter offgas streams will be discharged through the HLW Facility stack through flues HV-S3a and HV-S3b, and the HLW pulse jet ventilation offgas stream will be discharged through flue HV-S4. The emissions associated with potential leaks to process cells will be discharged through the HLW Facility stack via the HV-S2 flue.

## 3.6 Facility Control Philosophy

This section presents an overall control philosophy for the WTP. The goal of the facility control philosophy is to satisfy the following criteria:

- Preservation of worker and public safety
- Protection of the environment

- Preservation of equipment integrity
- Assurance of product quality
- Minimization of plant lifetime costs

The design, construction, and commissioning of the WTP are being conducted in a manner that is protective of employees, the public, and the environment. The process systems, piping, vessels, and equipment have been specifically designed to provide primary confinement of hazardous, radioactive, and chemical materials. The facility structures, along with their respective ventilation systems, will provide secondary confinement of airborne and liquid releases. The ventilation system will support confinement of airborne contamination within the building by directing the flow of air from areas of less contamination potential to areas of greater contamination potential. The ventilation system will also filter the building exhaust air.

Diagnostics will be used to optimize throughput and reduce downtime. A plant information computer with data entry and reporting capabilities will be provided to process information needed for facilitating plant optimization. Provisions will be made for overview and scheduling information.

The confinement and shielding requirements, combined with the need to provide hazard isolation and accessible areas for plant operation, have led to the building configuration of multiple cells and caves connected by transfer tunnels and shielded doors. This configuration provides a series of barriers enclosing the various zones, which are classified according to the contamination potentials.

Throughout the design phase, design reviews are conducted by multidiscipline teams to ensure safety and provide for feedback and improvement. The process systems, facility structure, and facility design ensure that WTP operations will be safe and protective of human health and the environment.

## 3.7 References

Agnew SF, Boyer J, Corbin RA, Duran TB, Fitzpatrick JR, Jurgensen KA, Ortiz TP, and Young BL. 1997. *Hanford Tank Chemical and Radionuclide Inventories HDW Model*, LA-UR-96-3860, Rev 4. Los Alamos National Laboratory, Los Alamos, NM.

Kirkbride and others. 2007. *Tank Farm Contractor Operation and Utilization Plan*, HNF-SD-WM-SP-12, Rev 6. CH2M Hill Hanford Group Inc., Richland, WA.

WA7890008967. *Hanford Facility Resource Conservation and Recovery Act (RCRA) Permit, Dangerous Waste Portion for the Treatment, Storage, and Disposal of Dangerous Waste, Part III, Operating Unit 10, (Waste Treatment and Immobilization Plant).*

24590-WTP-RPT-ENV-08-001, Rev 1, *Hanford Tank Waste Treatment and Immobilization Plant Risk Assessment Air Quality Modeling Protocol*

24590-WTP-RPT-PE-07-001, Rev 1, *WTP Waste Feed Analysis and Definition-EFRT M4 Final Report.*

24590-WTP-RPT-PT-02-005, Rev 6, *Flowsheet Bases, Assumptions, and Requirements.*

24590-WTP-RPT-RT-07-002 Rev 0, *Impact of Non-Aqueous Phase Organic Compounds in the Hanford WTP*.

24590-101-TSA-W000-0004-114-00021, Rev 00B, *Report - Estimate of Hanford Waste Insoluble Solid Particle Size and Density Distribution*.

**Figure 3-1 Location of the WTP on the Hanford Site**

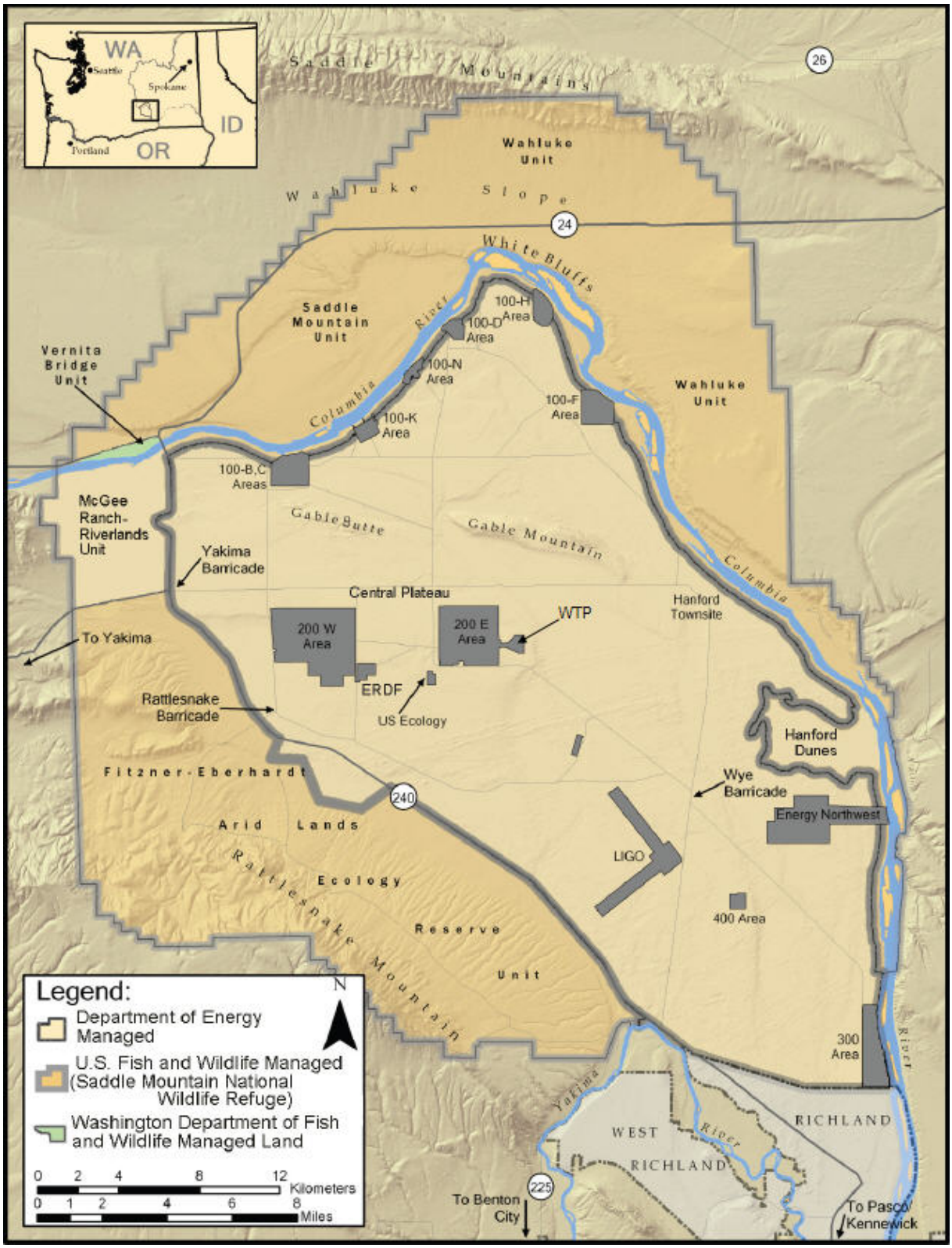


**Figure 3-2 WTP Layout**

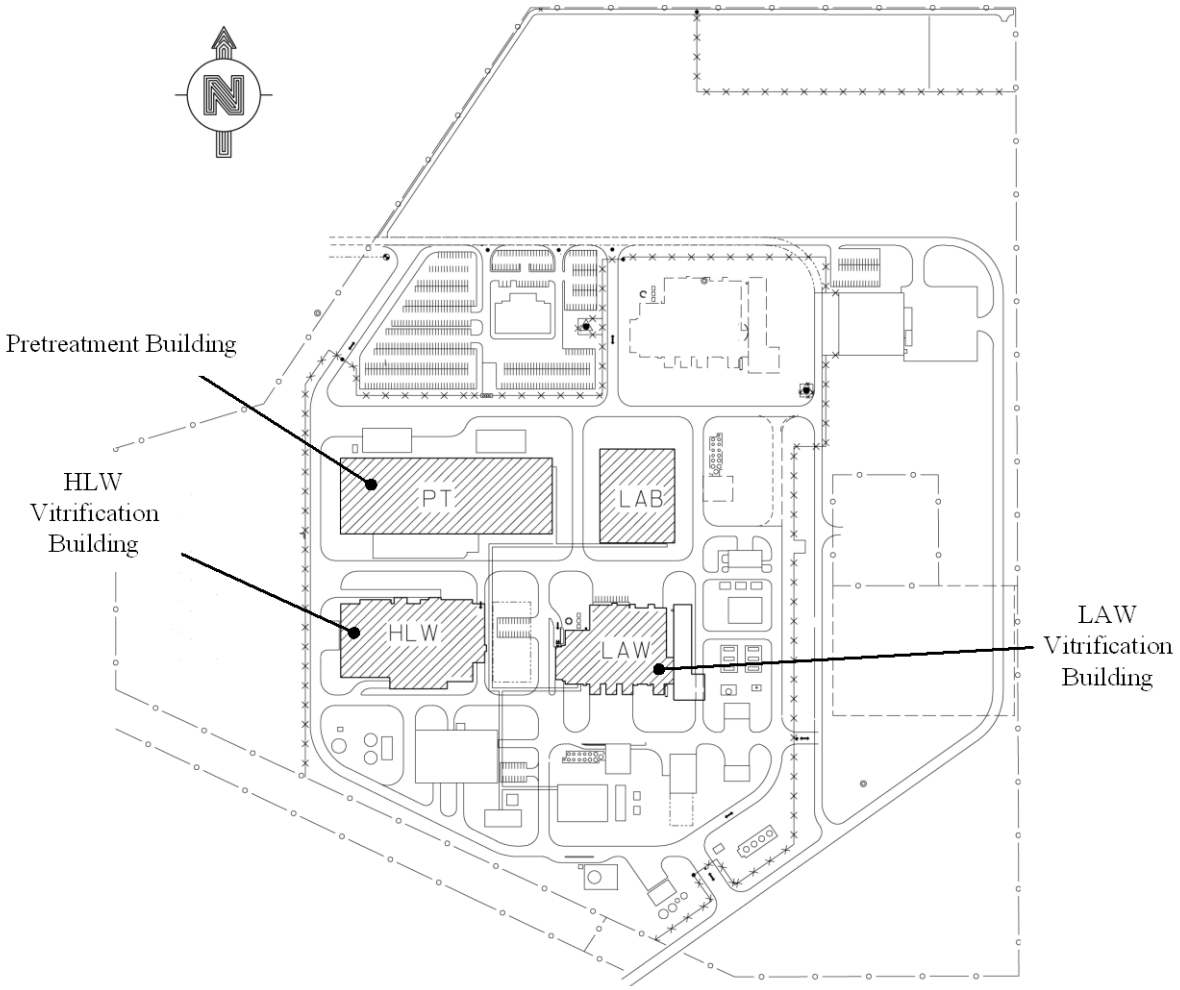


**Figure 3-3 Simplified Process Diagram**

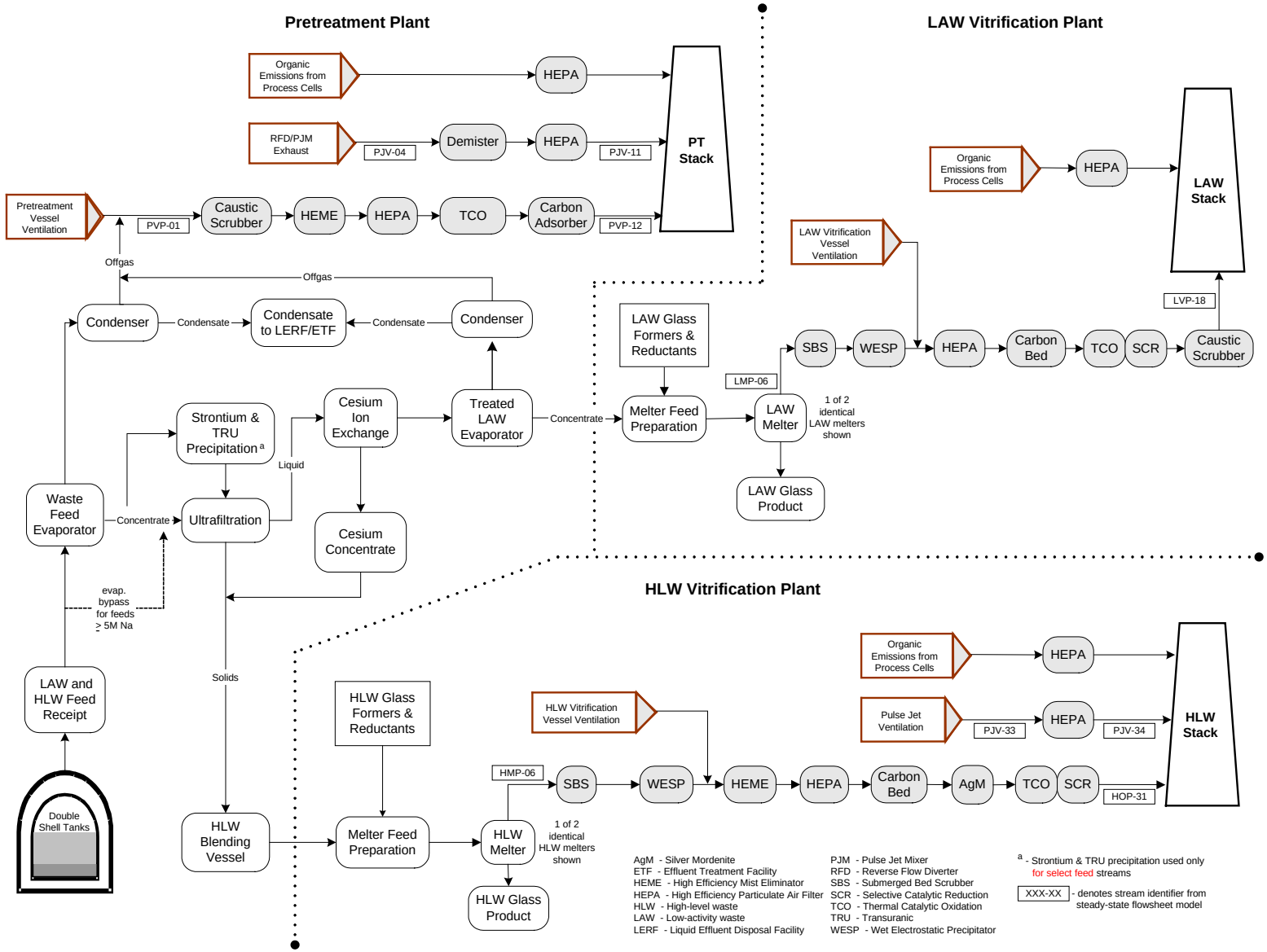
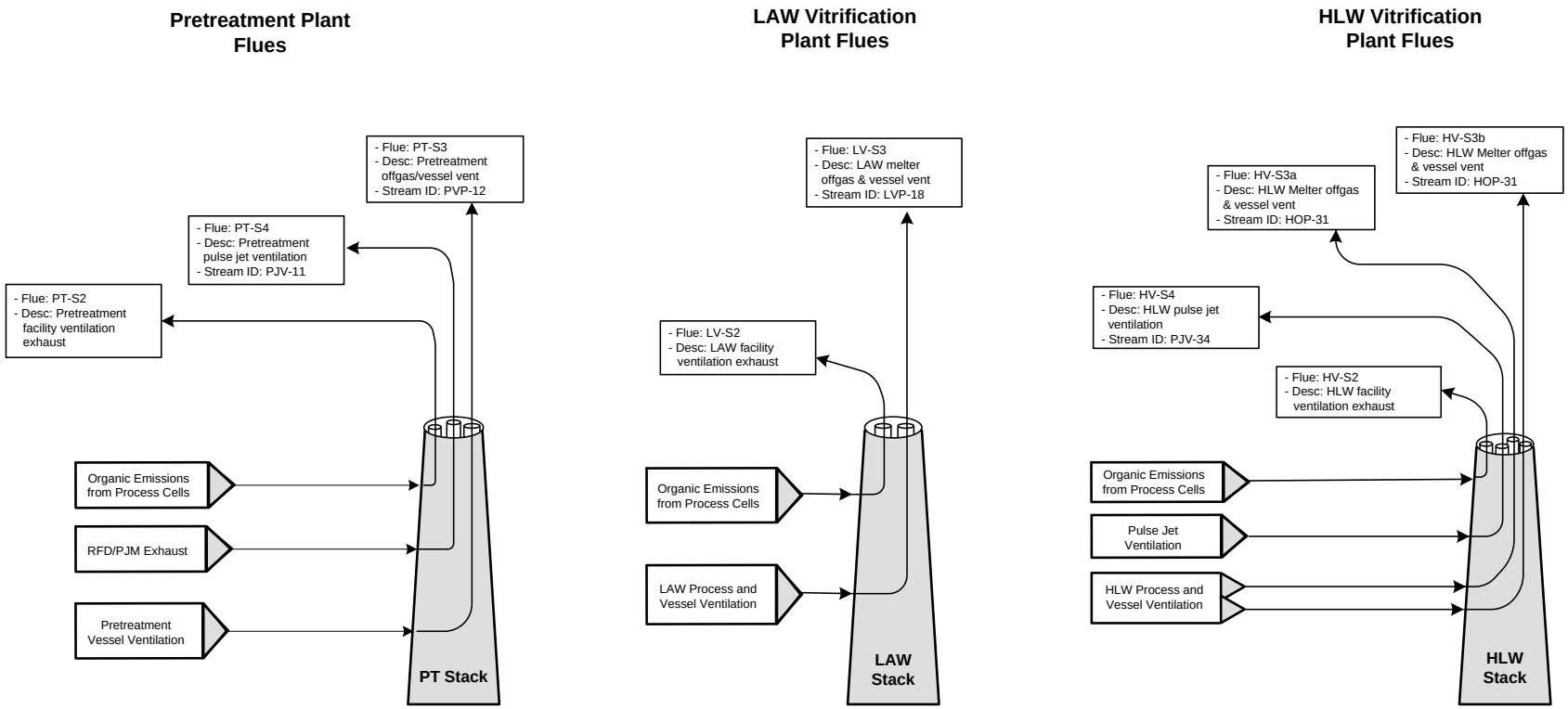


**Figure 3-4 Photos of the LAW Facility Stack**

The LAW Facility stack

LAW Facility flues enclosed within the stack lattice

**Figure 3-5 WTP Stacks and Flues**



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# 4 Constituents of Potential Concern

The *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA 2005, herein referred to as the HHRAP) recommends that the selection of COPCs focus on compounds that (1) are likely to be emitted because of the presence of the compound or its precursors in the waste feed, (2) are potential products of incomplete combustion (PICs), (3) are potentially toxic to humans, and/or (4) have a definite propensity for bioaccumulating or bioconcentrating in human and ecological food chains. The process for identifying COPCs is described in Supplement 1.

## 4.1 Identification of COPCs and ROPCs for the Quantitative Pre-demonstration Test Risk Assessment

The COPCs and ROPCs identified in Supplement 1 include an extensive list of chemicals and radionuclides that are (1) potentially present in the waste to be processed or (2) potentially produced as PICs during the WTP processing of waste. The process of identifying of COPCs and ROPCs for the PRA-selected chemicals is in accordance with the recommendations in the HHRAP.

Final COPCs and ROPCs carried through the quantitative risk assessment will be all those COPCs and ROPCs for which:

- Appropriate physical/chemical parameters are available to quantitatively estimate potential emissions and fate and transport behavior of the constituent through the environment
- Appropriate human health or ecological toxicity data are available to quantitatively evaluate potential effects of the constituent

Constituents not included in the quantitative risk assessment will be discussed qualitatively as part of the uncertainty assessment.

### 4.1.1 Identification of Organic and Inorganic COPCs for Inclusion in the Quantitative PRA

The list of organic COPCs consists of many categories of volatile and semivolatile organic compounds. EPA (1994) has identified several categories of WTP COPCs (e.g., dioxins/furans, polycyclic aromatic hydrocarbons [PAHs], polychlorinated biphenyls [PCBs], phthalates, other chlorinated organics, and nitroaromatics) as having the highest potential to cause increased risk to human health from chronic exposure. The organic and inorganic COPCs that can be carried through the quantitative risk evaluation is identified in Supplement 4. Note the data available are continually changing. Therefore, the PRA and FRA will update this information.

This list of inorganic COPCs includes the most stable form of the 11 chemicals listed below, also evaluated as ROPCs:

| | | |
|---|---|---|
| antimony | nickel | uranium |
| barium | selenium | yttrium |
| cadmium | strontium | zirconium |
| cobalt | tin | |

The PRA and FRA will evaluate the chemical toxicity (i.e., not associated with radioactivity) as well as the effect of the radioactivity for these constituents.

### 4.1.2 Identification of ROPCs for Inclusion in the Quantitative PRA

The preliminary ROPCs were identified per the method described in Supplement 1. Supplement 4 describes the available toxicity and physical/chemical data appropriate for evaluation of chronic human health, acute human health, and chronic ecological exposures to ROPCs.

## 4.2 Uncertainty in the COPC and ROPC Lists

The identification of COPCs and ROPCs for the PRA is uncertain because these constituents are identified before operation of the WTP and must, therefore, rely on assumptions regarding what may be in the waste feed and what may be produced as PICs. Because test data collected for the FRA environmental performance demonstration will be restrained by detection limits and variations in actual waste feed, the uncertainty will not be eliminated by these tests.

In both the PRA and FRA, uncertainty is introduced into the risk assessment by COPCs that cannot be carried through the quantitative assessment due to lack of toxicity data (all ROPCs have adequate toxicity data to be carried through the quantitative assessment).

The following section briefly describes the sources of uncertainty in the identification of COPCs and ROPCs for the PRA. Section 10 provides an overview of how these uncertainties will be evaluated, along with uncertainties in all other steps of the risk assessment.

### 4.2.1 Uncertainty in Identification of COPCs and ROPCs for PRA

Sources of uncertainty in the identification of COPCs and ROPCs include the following:

- Uncertainty in the contents of waste feed from the double-shell tanks (DST)
- Uncertainty in PICs that may be produced by the WTP, once operational

While a considerable amount of analytical data are available for the contents of the DSTs, the contents of all tanks have not been fully characterized. To compensate for deficits in the analytical data, the regulatory data quality objective (DQO) (Wiemers et. al. 1998) that was used as the basis for the COPC list incorporated constituents that could be present in the tanks, based on Hanford activities, even if those constituents have not been detected in analytical samples.

Limited PIC data are available from bench and pilot-scale tests performed on surrogate waste. This surrogate waste was designed to represent the most difficult-to-destroy chemicals potentially present in the tank waste and, thus, to provide a conservative estimate of potential PICs. To maintain a conservative bias in the PRA, PICs identified by EPA (1998) as present in stack emissions from existing hazardous waste incinerators were included in the COPC list, along with WTP-specific PICs identified in the bench-scale testing. The ROPCs are not produced as PICs.

### 4.2.2 Uncertainty in COPCs Not Included in the Quantitative Assessment

Some COPCs identified as potentially present in the waste or as PICs cannot be carried through the quantitative risk assessment because appropriate toxicity data are not available to characterize their potential effects on human or ecological receptors. Constituents without toxicity information will not be included in the quantitative human health or ecological risk assessments. If these constituents are similar in toxicity and persistence to the constituents with toxicity data, the total risk or hazard would be underestimated approximately proportionately. Hazards would be underestimated by a factor proportional to the number of constituents without data divided by the total number of constituents. Similarly, for ecological receptors, if the toxicity and persistence of the constituents without toxicity data are similar to the toxicity and persistence of the constituents with toxicity data, the total hazard would be underestimated proportionally, according to the percentage of constituents without ecological toxicity data. Supplement 4 contains a tally of constituents with data and identifies the kind of data available. The PRA and FRA will address the uncertainty associated with COPCs without toxicity data.

## 4.3 Summary of Identification of COPCs and ROPCs

The list of COPCs and ROPCs selected for the PRA includes many more compounds than are expected in actual facility emissions. The list is long because assumptions were used to compensate for the uncertainty regarding the exact makeup of the waste and the lack of environmental performance demonstration data (it was assumed that all chemicals potentially present in the waste would be emitted along with all chemicals identified as PICs from any type of combustion unit). The list of preliminary COPCs and ROPCs includes numerous chemicals (especially organic chemicals) that have never been detected in the tank waste.

Supplement 1 documents the process used to identify preliminary COPCs and ROPCs for the PRA. Supplement 4 summarizes the current availability of data to quantitatively evaluate the preliminary COPCs and ROPCs. These tables also provide a list of the COPCs and ROPCs that will be quantitatively evaluated in the PRA. The uncertainty assessment will qualitatively address preliminary COPCs and ROPCs not quantitatively addressed in the PRA.

Supplement 4 identifies the human receptor groups and exposure pathways for which risks/hazards can be quantified for each COPC and ROPC (based on the availability of physical/chemical parameters for fate and transport modeling and toxicity data for evaluating effects on human health receptors). The human receptors identified in these tables are as follows:

- Hanford site industrial worker
- Residential receptors
- Subsistence receptors
- Nursing infant
- Acute receptor

Also, Supplement 4 identifies the ecological receptors for which hazards can be quantified for each COPC and ROPC. The ecological receptors identified in these tables are as follows:

- Terrestrial plants and invertebrates
- Terrestrial mammals and birds
- Aquatic biota, salmonids, and benthic invertebrates

The COPC and ROPC lists will be reevaluated for the FRA following completion of the environmental performance demonstration tests. This reevaluation will take into account any new information gathered during the PRA and performance demonstration tests and will include input and approval by Ecology and EPA.

## 4.4 References

EPA. 1994. *Draft Exposure Assessment Guidance for Resource Conservation and Recovery Act Hazardous Waste Combustion Facilities: Attachment, April 15, 1994*, Table 1 – Chemicals Recommended for Identification and Table 2 – Chemicals for Potential Identification. US Environmental Protection Agency, Washington, DC.

EPA. 1998. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, Peer Review Draft, EPA/530/D-98/001B. US Environmental Protection Agency, Washington, DC.

EPA. 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, Final, EPA/530/R-05/006. US Environmental Protection Agency, Washington, DC.

Wiemers KD, Lerchen ME, Miller M, and Meier K. 1998. *Regulatory Data Quality Objectives Supporting Tank Waste Remediation System Privatization Project*, PNNL-12040, Rev. 0. Pacific Northwest National Laboratory, Richland, Washington.

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# 5 Estimation of Emissions

The WTP is being designed to pretreat and vitrify radioactive mixed waste. A bounding estimate of stack emissions from the WTP has been developed to allow for numerical quantification of the human and ecological risks associated with airborne emissions. The emissions estimate individually considers 409 organic, inorganic, and radionuclide constituents of potential concern that could result from processing Hanford tank waste through the WTP. This section provides an overview of the assumptions and methodology used to arrive at the WTP stack emission estimates.

## 5.1 Emissions Sources

The SLRA considers potential emissions from the following sources:

**Process Emissions.** Process emissions are defined as chemicals and radionuclides released from the WTP plant stacks as a result of normal (i.e., routine) operations. Emissions associated with waste processing are discussed in Section 5.2.

**Process Upset Emissions.** Process upset emissions are defined as chemicals and radionuclides released from the WTP stacks as a result of nonroutine operations (such as a process malfunction). Process-upset emission rates are assumed to be higher than normal process emission rates because the upset condition is assumed to result in decreased offgas treatment efficiency or increased formation of PICs. However, process-upset emissions are for a shorter duration. For the PRA, the conservative assumption that all upset conditions result in increased emission rates for short durations will be used. Process upset conditions are further described in Section 5.3.

**Non-Steady State Operations Emissions.** The WTP may have idle time and will have maintenance time. Changeout of HEPA filter media and replacement of catalysts are examples of maintenance activities. These non-steady-state operations are assumed to be bounded by the upset factor multipliers (see Section 5.3).

**Fugitive Emissions.** Fugitive emissions are defined as emissions of chemicals and radionuclides from non-stack sources. The WTP processing buildings that will manage the Hanford tank waste will be operated under negative pressure, and the air from the process buildings will be released to the atmosphere through one of the stacks or flues described in Section 3.5. Since the WTP will not have emissions that do not pass through a stack or flue, by definition, the fugitive emissions from the facility will be zero. However, the WTP emissions will consist of vapor phase organics that are assumed to be derived from valves and other ancillary equipment leaking in WTP process cells. These vapor-phase organic emissions are analogous to fugitive emissions in that the vapor-phase will be unabated by the HEPA filtration systems that control particulate emissions from process cells. Fugitive emissions and unabated organic emissions from process cells are further described in Section 5.4.

The SLRA will not consider emissions associated with accidental releases or with the retrieval and transfer of wastes from the Hanford DST system. Accident scenarios, such as the rupture of a tank or vessel line, are addressed in the hazards analysis and other nuclear and process safety documents. Emissions associated with the transfers from the Hanford DST system are expected to be sufficiently bounded by the WTP process emissions estimates, as described in Section 5.5.

## 5.2 Process Emissions

The methods, assumptions, and resulting process emission rates are documented in 24590-WTP-RPT-PO-03-008, *Integrated Emissions Baseline Report for the Hanford Tank Waste Treatment and Immobilization Plant*. This report is included as Supplement 2 to this work plan and is summarized below.

The process emissions estimate was developed using the WTP Project's baseline steady-state flowsheet model. The steady-state flowsheet tracks modeled constituents across the PT, LAW, and HLW Facilities, and provides a steady-state representation of process stream compositions at unit operation locations. The steady-state conditions provide an overall material and energy balance with time-averaged flow rates. The steady-state flowsheet allows for the use of simple equipment decontamination factors or more complex thermodynamic calculations to evaluate the modeled constituents of concern. Evaporator partitioning and organic vessel vent emissions were predicted from known liquid-phase concentrations using vapor-liquid equilibrium expressions. Henry's Law constants were compiled for the organic vapor-liquid equilibrium calculations.

Decontamination factors are defined as the ratio of the constituent concentration going into a unit operation to the concentration of the constituent coming out of the unit operation. Decontamination factors for offgas treatment equipment vary from one constituent to another and are established based on the physical properties of a constituent (e.g., offgas phase), published literature, or available research and testing results.

Vapor phase and particle phase emissions are tracked in the emissions model. The offgas phase of a constituent is described by the variable $F_v$, which denotes the fraction of the ROPC or COPC air concentration in the vapor phase (EPA 2005). Particle phase constituents are defined as having an $F_v < 0.05$ (i.e., having a vapor concentration of less than 5 %). Vapor phase constituents have an $F_v = 1$. A subset of organic COPCs falls into the category of particle-bound ($0.05 \leq F_v < 1$). Particle-bound constituents have a portion of vapor condensed onto the particle surface. The emissions model separately tracks the vapor and particle fractions of particle-bound constituents. Section 6 provides additional discussion of constituent phase and how phase is handled in the air dispersion and risk modeling steps. A complete list of $F_v$ values for COPCs and ROPCs is included in Supplement 4 of this work plan.

The steady-state flowsheet tracks the main constituents expected to have the greatest impact on the material and heat balance of the plant. The emission rates for COPCs not analyzed directly in the steady-state flowsheet (with the exception of PICs) were estimated using the modeling output from a constituent that was in the steady-state flowsheet. The correlations of modeled and unmodeled constituents were based on constituents having similar physical properties, with an adjustment made for differing feed concentrations, if applicable.

The emission rates of PICs were estimated based on research and technology testing data from small-scale melter runs spiked with hazardous organic constituents at the Vitreous State Laboratory of the Catholic University of America.

Additional details on process emissions estimation, including the basis for feed composition, treatment efficiencies, the correlation of modeled and unmodeled constituents, and PIC emission rates are described in 24590-WTP-RPT-PO-03-008.

### 5.2.1 Total Organic Emission Rate Correction

The WTP emission COPCs and ROPCs include the following categories:

1) COPCs and ROPCs identified as having potential to be present in Hanford tank waste
2) COPCs identified by the EPA as potentially being emitted as PICs during thermal processing of organic materials
3) COPCs detected as PICs during Research and Technology testing of proposed WTP vitrification melters

The organic emissions are determined by stack sampling during demonstration testing of the WTP HLW and LAW melters. Sampling and analytical methods that will be used during the demonstration testing will include those to detect volatile, semi-volatile, and non-volatile feed and PIC species.

Each of these sampling and analytical methods will be calibrated as appropriate to detect organic COPCs. In addition, COPCs that are tentatively identified will be included as part of the known fraction and be quantified in the FRA. The COPCs for which one of the methods is appropriate will be included at the achieved detection limit to provide quantification of the stack concentration of the COPC ($C_i$). The COPCs will be summed according to their classification as a volatile ($C_i^{VOC}$), semi-volatile ($C_i^{SVOC}$), or gravimetric ($C_i^{GRAV}$) fractions. If a constituent is quantified and not included on the WTP COPC list, it will also be added to the appropriate classification ($C_i^{VOC}$, $C_i^{SVOC}$, $C_i^{GRAV}$) to obtain a sum for each emission fraction. Constituents appropriate to the sampling/analytical methods that were not detected and are not included on the WTP COPC list will not be summed.

The total organic emission (TOE) factors $F_{TOE}^{VOC}$, $F_{TOE}^{SVOC}$, $F_{TOE}^{GRAV}$ will be determined as described below. Each classification of COPC, non-detect (at the appropriate detection limit), tentatively identified compounds (TIC), and the detected constituent will be multiplied by the appropriate TOE factor. These calculated values will be used to assess risks in the FRA.

Only a limited number of organic compounds can be accurately identified and quantified using standard stack gas sampling and analysis methods. A portion of the emissions profile remains unaccounted for. The EPA developed the TOE test to account for unidentified organic compounds because existing methods did not fully determine the total mass of organics present in stack gas emissions. The TOE determination measures organic fractions for three boiling point ranges: (1) a volatile, field gas chromatograph fraction (boiling points less than 100 ºC); (2) the semivolatile, total chromatographable organics fraction (boiling points from 100 ºC to 300 ºC); and (3) the non-volatile, gravimetric (*GRAV*) fraction (boiling points greater than 300 ºC). The TOE will be measured during the performance demonstration tests and used in the FRA in conjunction with the identified organic compounds to calculate TOE factors that can then be used to quantitatively evaluate potential risks from the unidentified fractions of organic compounds in the stack gas. A separate TOE factor will be calculated using Equations 5-1 through 5-3 for each fraction.

$$F_{TOE}^{VOC} = TO_{TOTAL}^{VOC}/\Sigma_i C_i^{VOC} \qquad \text{(Eq. 5-1)}$$

where:

$F_{TOE}^{VOC}$ = TOE factor for the volatile fraction (unitless)

$TO_{TOTAL}^{VOC}$ = Total volatile organic emission (milligrams per cubic meter [mg/m$^3$])

$C_i^{VOC}$ = Stack concentration of $i^{th}$ identified volatile COPC (mg/m$^3$)

$$F_{TOE}^{SVOC} = TO_{TOTAL}^{SVOC}/\Sigma_i C_i^{SVOC} \quad \text{(Eq 5-2)}$$

where:

$F_{TOE}^{SVOC}$ = TOE factor for the semivolatile fraction (unitless)
$TO_{TOTAL}^{SVOC}$ = Total semivolatile organic emission (mg/m$^3$)
$C_i^{SVOC}$ = Stack concentration of $i^{th}$ identified semivolatile COPC (mg/m$^3$)

$$F_{TOE}^{GRAV} = TO_{TOTAL}^{GRAV}/\Sigma_i C_i^{GRAV} \quad \text{(Eq 5-3)}$$

where:

$F_{TOE}^{GRAV}$ = TOE factor for the nonvolatile fraction (unitless)
$TO_{TOTAL}^{GRAV}$ = Total nonvolatile organic emission (mg/m$^3$)
$C_i^{GRAV}$ = Stack concentration of $i^{th}$ identified nonvolatile COPC (mg/m$^3$)

Using the assumption that the unknown portion of the emission is similar in toxicity and chemical properties to the known compounds, a risk will then be attributed to the unknown portion of the stack emission by multiplying the emissions rate of each identified compound by its fraction's TOE factor. The application of the TOE factors will be discussed in the FRA as an uncertainty that has potential to overestimate and underestimate risks, depending on whether fate and transport and/or toxicological data are available for the identified compounds. The TICs described in this section will be included in identified compound fractions for the purposes of determining the TOE factors.

### 5.2.2 Estimating Emissions of Nondetect COPCs

Nondetect COPCs and ROPCs will be managed as follows:

- If the compound is in the surrogate feed used during demonstration testing, it is assumed present at the detection limit
- If the compound is on the COPC list and not detected, it is assumed present at one-half the detection limit
- If the chemical is not on the COPC list, not a risk driver, and not detected in any of the test runs, assume the chemical is not present
- If the chemical is a risk driver (PAHs, PCBs) and not found, assume it is present at the detection limit
- If the chemical is detected in one run but not in others, assume the detected value and one-half the detection limit for the other runs

Whether a compound is detected will be based on the detection limits as described below for the non-isotope and isotope dilution methods.

The FRA will use the methodology recommended in HHRAP to quantify nondetects for COPCs analyzed with non-isotope dilution methods. A nondetect in this case would be a value below the reliable detection limit (RDL). The RDL is defined as 2.623 times the method detection limit (MDL). The MDL is defined

as the minimum concentration of a substance that can be measured (via non-isotope dilution methods) and reported with 99 % confidence that the analyte concentration is greater than 0. The MDL is determined from analysis of a sample in a specific matrix type containing the analyte, and is considered the lowest level at which a compound can be reliably detected. Procedures for determining an MDL are specified in 40 CFR 136, Appendix B.

The FRA will also follow the HHRAP recommendation that the "method-defined estimated detection limit (EDL)," as defined by SW-846 (EPA 1986), be used to quantify nondetects for COPCs analyzed with isotope dilution methods. The EDL is defined as a laboratory estimate of the concentration of a given analyte required to produce a gas chromatogram signal with a peak height of at least 2.5 times the background signal level.

### 5.2.3 Blank Corrections

Blank samples are used as a quality control check and are intended to indicate whether contamination was introduced into a sample in the field while the samples were being either collected, transported to the laboratory, or prepared and analyzed. This helps ensure that the measured levels of target analytes are indeed from the vitrification system and not from prior contamination of the sampling train. Corrections to account for concentrations of compounds detected in blank samples will be applied only to metals, following procedures outlined in EPA Method 29, *Determination of Metals Emissions from Stationary Sources* (40 CFR 60). No blank correction factors will be applied to the results from any other analytes. The rationale for this distinction is discussed below.

The overall basis for not allowing blank correction of emissions data used in this risk assessment is the assumption that blank correction will reduce the accuracy of the determination and would represent a non-conservative uncertainty. Consequently, disallowing these blank corrections is a conservative assumption that is consistent with a screening-level risk assessment. This limits the use of blank data to providing the basis, if necessary, for retesting. The EPA stack sampling methods are not explicitly designed for generating data to be used in risk assessment. The fact that these methods may provide guidelines for blank correction of data does not automatically provide a facility assurance that blank correction will be allowed or if allowed that the extent of correction delineated in the method will be allowed. Also, caution must be practiced in applying blank results to correct or qualify sample results for any purpose, as blanks are provided in minimal quantities (e.g., one per test condition, or one per test) and therefore, are at best only qualitative indicators of the validity of a data set.

The approach for blank correction described in 40 CFR 60, Method 29, is very detailed. (It is one of the few EPA methods that provides for subtraction of reagent blank values within the limitations of the method specifications). This is identical to the approach described in EPA SW-846, Method 0060, which was originally designed to meet the data needs for hazardous waste incinerators and boiler and industrial furnaces risk evaluations. As with all the emission testing, every reasonable effort to identify potential for contamination and to minimize that potential will be made. These efforts will be taken into consideration, as well as the extent of any contamination, when determining the data to input into the risk assessment.

### 5.2.4 Tentatively Identified Compounds

The HHRAP guidance recommends identifying and quantifying as many non-target organic compounds as possible from the emission test results, regardless of the compounds' toxicity. Identifying a large portion of the "unknown" peaks in a gas chromatogram leads to a more complete organics mass balance, and less uncertainty in the overall risk evaluation. The 30 largest TICs, for which the peaks are greater

than 10 % of the nearest internal standard, will be identified, quantified, and added to the list of detected compounds from the vitrification system. These compounds will then be evaluated in the same manner as any other detected compound. The inclusion of TICs quantitatively in the risk assessment will be discussed in the uncertainty section as having potential to overestimate and underestimate risks.

### 5.2.5 Maximum Emissions Rate Correction

To the greatest extent possible, emissions rates during the vitrification systems' performance demonstration testing will be measured under what would be considered "worst-case" operating conditions. Reasonable maximum emissions rates will be determined in accordance with current HHRAP guidance by using the maximum of the three emission rates identified for each COPC or ROPC during a particular test condition, adjusted for process upsets.

## 5.3 Process Upset Emissions

Process upset conditions include periods of startup, shutdown, process malfunction (i.e., the unit is operating outside the permitted operating conditions), or equipment failure. Periods when process equipment is being maintained or in an idle condition are also included. Process upset conditions are generally assumed to result in greater than normal stack emissions during the short period of the upset. However, EPA has indicated that upsets are not generally expected to significantly increase stack emissions over the lifetime of a facility (HHRAP).

The potential for increased emissions during upset events will be addressed through the application of upset factors. These upset factors, as described below, will be applied (i.e., adjustments will be made) to the estimated emissions that are environmentally modeled. These upset factors are based on the amount of time the facility is expected to operate in an upset condition and the estimated magnitude of stack emissions during upset relative to routine operating conditions. The preferred method for estimating this upset factor is through the use of data from existing facilities that have operating conditions similar to the proposed WTP. The frequency and duration of upset events may be estimated based on the HHRAP:

- Data from continuous emissions monitoring systems that measure operating parameters such as stack carbon monoxide or oxygen
- Data on combustion chamber, air pollution control system (APCS), or stack gas temperature
- Ratio of automatic waste feed cut-off frequency and duration to operating time
- Variations in the APCS operating conditions

The potential magnitude of emissions during upset events may be estimated based on stack test data collected during upset conditions.

The EPA default upset factors represent worst-case conditions and will be used for the PRA. The EPA default upset factors are based on the data described above from operating hazardous waste combustion facilities. The default upset factors are expected to over-predict upset emissions from the WTP for several reasons, including:

- Carbon monoxide is frequently used as an indicator of upset conditions, and automatic waste feed cut-offs are often triggered by increased stack gas concentrations of carbon monoxide. However, routine operations, such as adjusting waste feed or air intake rates, will cause brief spikes in carbon monoxide concentration.

- Test data used for these defaults is based on hazardous waste combustion facilities designed for the destruction of liquid or solid organic waste, or both. The technology and waste feed of the WTP melters are different and less subject to upset than these facilities.

The HHRAP default upset factors are 2.8 for organic chemicals and 1.45 for metals, calculated as shown below.

**Organics**. A default facility is assumed to operate under upset conditions 20 % of the time and stack emissions are assumed to be 10 times greater than normal during this time:

$$\text{Upset Factor} = (0.80)\ (1) + (0.20)\ (10) = 2.8$$

**Metals**. A default facility is assumed to operate under upset conditions 5 % of the time with stack emissions 10 times greater than normal during this time:

$$\text{Upset Factor} = (0.95)\ (1) + (0.05)\ (10) = 1.45$$

The EPA has not determined a default upset factor for radionuclides. For the PRA, radionuclides are assumed to behave similarly to metals, with an upset factor of 1.45. The same upset factors will be used for both the plausible and worst-case scenarios in the PRA and the FRA.

These default upset factors (2.8 for organics and 1.45 for inorganics and radionuclides) will be used for all vapor-phase emissions. An upset factor of one (1) will be used for all particle and particle-bound emissions, as described below.

The entire pretreatment and vitrification processes will be contained within buildings designed such that the only exits for air and emissions will be through one or more HEPA filters. When the process is operating normally, all air and emissions will pass through numerous air pollution control devices. However, even if the process experiences an upset condition or shuts down and all of the active pollution control devices operate poorly or fail completely, the only way for air and emissions to pass out of the facility will be through the HEPA filters.

The removal efficiency for a single stage of HEPA filtration is, by definition, 99.97 % (decontamination factor = 3333) for 0.3-$\mu m$ particles. This decontamination factor applies to the efficiency of the HEPA filter material. According to the *Nuclear Air Cleaning Handbook* (DOE 2003), the theoretical removal efficiency for multi-stage HEPA filtration systems is $3333^n$, where $n$ is the number of HEPA filter stages.

For the WTP, all multi-stage HEPA filter designs consist of two stages of filters in series. The theoretical maximum decontamination factor for two-stage designs would be 11,108,889. The Handbook (DOE 2003) states that for systems that adhere to the design, construction, testability, and maintainability of ASME N509, an appropriate multi-stage decontamination factor under normal operating conditions is $3000^n$. The theoretical maximum decontamination factor under this method would equate to a decontamination factor of 9,000,000.

For conservatism, WTP has assumed that the decontamination factor of the first stage filter is 2000 and the decontamination factor of the second stage filter is 100. The two-stage HEPA filter decontamination factor for the WTP is, therefore, 2000 (1st stage) times 100 (2nd stage) = 200,000.

Therefore, an upset factor of one (1) will be applied to the particulate and particulate-bound emissions estimates for organics, inorganics, and radionuclides because the HEPA filter removal efficiency used in the emissions estimate already includes an assumption of decreased removal efficiency due to upset conditions such as moisture in the filters.

## 5.4 Fugitive Emissions

Fugitive emissions are defined as "emissions, which could not reasonably pass through a stack, chimney, vent, or other functionally equivalent opening" (WAC 173-400-030, *General Regulations for Air Pollution Sources*). The WTP process buildings that manage the Hanford tank waste will be operated under negative pressure, and the air from the process buildings will be released to the atmosphere through a stack or flue. Transfer lines between buildings that will contain Hanford tank waste will be double-wall pipe. Therefore, the WTP will not emit fugitive emissions.

Building ventilation and process offgases will be treated by abatement systems that employ best-available control technology for criteria pollutants, radionuclides, and toxic air pollutants prior to release to the environment through a stack or flue. Organic compounds could be released into the process cells from ancillary equipment. These emissions will be treated by HEPA filters that will abate particulate or particle-bound organic compounds. Organic compounds existing in the vapor phase will not be captured by the HEPA filters. These organic emissions from process cells have been quantified for purposes of risk assessment.

Organic emissions from process cells will be quantified by establishing the total organic emissions associated with ancillary equipment in process cells. This total includes particle, particle-bound, and vapor-phase contributions that are associated with ancillary equipment, such as valves, pump seals, compressor seals, and connectors. The methodology and emissions factors used to estimate releases from ancillary equipment are consistent with the EPA guidance document *Protocol for Equipment Leak Emission Estimates* (EPA 1995). After establishing the total organic emissions, the fraction of emissions considered to be particle or particle-bound in the offgas will be removed. The particle and particle-bound organic constituents will be captured by HEPA filtration systems in the process cell ventilation system where the concentration is reduced by a factor of 200,000 (Supplement 2, 24590-WTP-RPT-PO-03-008). The remaining vapor-phase organic fraction will be carried forward to the corresponding facility flue where the emission rates are considered in conjunction with other process emissions for risk assessment.

A detailed discussion of the methods, calculations, and results associated with organic emissions from process cells is described in 24590-WTP-HAC-50-00001, *Estimated Organic Emissions from Process Cells*. This calculation is included as Supplement 3 to this work plan.

## 5.5 Uncertainty in WTP Emissions Estimate

Although there are uncertainties associated with the parameters used to arrive at estimated process emissions, these uncertainties have been recognized and managed through conservative assumptions applied throughout the emissions estimation process. For example, analytical uncertainty is associated with the organic, inorganic, and radionuclide characterization data that describes the waste feed streams to the WTP. To accommodate characterization uncertainties, the inorganic and radionuclide source terms are based upon the known concentrations for constituents in tanks that the WTP expects to process.

Because less data are available for organics in the tank waste, conservatism has been applied with respect to the organic feed vector. For organic compounds, the emission estimate assumes that incoming organic

concentrations are elevated by applying a scalar such that uncertainty in the feed vector is compensated for and sufficiently bounded (24590-WTP-RPT-PO-03-008). The methodologies applied to assigning feed concentrations should ensure that the actual concentrations of organic, inorganic, and radionuclide constituents encountered during operations will be conservatively bounded by the emissions estimate assumptions.

The conservatism applied to the feed vector is also applied to the assignment of equipment decontamination factors. In cases where a particular treatment process has a range of achievable treatment efficiencies, the lower end of the range (which translates to the higher offgas emission rate) has been applied in the emissions estimate. The ranges of treatment efficiencies for individual treatment processes are derived from a variety of sources, including research and technology data, engineering studies, vendor literature, and regulatory guidance. For example, in establishing filtration removal efficiencies, the dual-HEPA filtration systems used in the WTP offgas treatment systems have an assumed decontamination factor of 200,000 for particle and particle-bound constituents in the offgas (24590-WTP-RPT-PO-03-008). This decontamination factor is consistent with the assumptions used across other Hanford permitting applications and is considered conservative, even for particle sizes of 0.3 μm, which are most likely to pass through HEPA filtration.

The WTP emissions estimate does not estimate the emissions that could result from retrieval of waste feed from the Hanford DSTs. Although these emissions are not included, the risks associated with retrieval of DST feeds will be sufficiently bounded for the following reasons:

- The WTP feed vector assumes receipt of the entire DST inventory, and has been developed to conservatively overestimate the constituent concentrations present in the tank contents. As described above, the organic feed vector scaled up expected feed concentrations to account for uncertainties in characterization information.
- DST retrieval operations would be infrequent and, therefore, the assumed continuous 24 hours per day, 7 days per week, operation of WTP at 100 % efficiency would dominate any long-term risk calculations. Any acute risks associated with the DST retrieval are not expected to coincide with either the timing or location of acute risks estimated for the WTP due to temporal and spatial differences.
- Entrainment losses of particle-bound constituents from the DST tank system would be comparable to the control in the WTP facility (i.e., both offgas discharge streams are controlled by HEPA filtration systems that provide a high removal efficiency for particulates).
- Losses of all constituents are being assessed and controlled under regional air-permitting control authorities.

Based on the above description, the uncertainties associated with not including the DST emissions are likely bounded by the WTP estimates. The DST transfers to the WTP will likely neither over- or under-estimate the risks.

## 5.6 References

### 5.6.1 Project Documents

24590-WTP-HAC-50-00001, *Estimated Organic Emissions from Process Cells*.

24590-WTP-RPT-PO-03-008, Rev 0, *Integrated Emissions Baseline Report for the Hanford Tank Waste Treatment and Immobilization Plant*.

### 5.6.2 Codes and Standards

40 CFR 60. App. A-8. "Method 29 - Determination of Metals Emissions from Stationary Sources," US Environmental Protection Agency, *Code of Federal Regulations*, as amended.

40 CFR 136. App. B. "Definition and Procedure for the Determination of the Method Detection Limit -Revision 1.11," US Environmental Protection Agency, *Code of Federal Regulations*, as amended.

WAC 173-400-030. *General Regulations for Air Pollution Sources*, Washington Administrative Code, last updated 7 July 2002.

### 5.6.3 Other Documents

DOE. 2003. *Nuclear Air Cleaning Handbook*, DOE-HDBK-1169-2003, November 2003, US Department of Energy, Washington, DC.

EPA. 1986. *Test Methods for Evaluating Solid Waste, Physical and Chemical Methods*, SW-846, Third Edition (as amended by Updates I, II, IIA, IIB, and III). US Environmental Protection Agency, Washington, DC.

EPA. 1995. *Protocol for Equipment Leak Emission Estimates*, EPA-453/R-95-017. US Environmental Protection Agency, Research Triangle Park, North Carolina.

EPA. 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, Final, EPA/530/R-05/006. US Environmental Protection Agency, Washington, DC.

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# 6 Environmental Modeling

Environmental modeling refers to several types of models used to simulate the route of chemicals and radionuclides from the stack toward human and ecological receptors. This section describes the environmental modeling approach for the WTP. Air dispersion modeling is discussed first (Section 6.1), followed by soil accumulation modeling (Section 6.2), surface water accumulation modeling (Section 6.3), sediment accumulation modeling (Section 6.4), and terrestrial plant accumulation modeling (Section 6.6). Modeling for other media (such as specific animals and fish) is briefly discussed in Section 6.7 (more detailed information is provided in Sections 7 and 8, because these media are modeled slightly differently for human health and ecological risk). Uncertainties related to environmental modeling are discussed in Section 6.8. A summary of environmental modeling is presented in Section 6.9.

## 6.1 Air Dispersion Modeling

Air dispersion modeling will be used to estimate the ambient air quality and deposition rates resulting from emissions of vapor, particle-bound, and particle-phase chemicals and radionuclides during operations of the WTP. This section provides details of the approach that will be used in this task. Specific air model settings are described in the *Hanford Tank Waste Treatment and Immobilization Plant Risk Assessment Air Quality Modeling Protocol* (24590-WTP-RPT-ENV-08-001, Supplement 5).

### 6.1.1 Model Selection

The Industrial Source Complex Short Term Model, Version 3 (ISCST3) (EPA 1995, EPA 2002) was initially proposed to evaluate the air quality in the vicinity of the WTP. This model, preferred by the EPA (*Guideline on Air Quality Models*, 40 CFR 51, Appendix W)[1], is generally considered a conservative model for applications such as the SLRA. The model uses emissions source data and hourly meteorological data to estimate ambient air concentrations and deposition rates of gases and particles at locations (receptors) of interest in the vicinity of the facility (EPA 2002). The ISCST3 is an Eulerian "plume" model that sends emissions out in a straight line from the emission source, in the direction of the wind at the time of release. The plume continues spreading out and traveling away from the emission source, becoming more and more dilute with distance. The use of this model was evaluated for application to the WTP.

After this initial evaluation, it was determined that the CALPUFF model, a Lagrangian "puff" model, would be more appropriate in this application. The EPA adopted CALPUFF as a guideline model and added it to 40 CFR Part 51, Appendix W (*Federal Register*, 15 April 2003), giving it equivalent status to the AERMOD model for long range transport. The air modeling regulation, 40 CFR 51, Appendix W, provides for case-by-case approvals for other uses of CALPUFF, provided that it is demonstrated to be suitable. In addition, there are several advantages to using the CALPUFF modeling system (Version 5.6) for this application, which would result in a more realistic and representative characterization of the air quality:

- **Gaussian puff dispersion formulation**: Plumes are treated as a series of Gaussian puffs that move and disperse according to local conditions that vary in time and space.

[1] Note, ISCST3 has subsequently been replaced by AERMOD as the preferred model in 40 CFR 51, Appendix W.

- **Meteorology:** Wind and other meteorological variables are allowed to vary in a three-dimensional space.
- **Wet- and dry-deposition mechanisms**: Deposition processes are included for particle and vapor phase pollutants that depend on the characteristics of the pollutant, the surface geophysical parameters, and meteorological conditions; the model accounts for the mass of pollutant removed from the plume when deposition occurs.
- **Other improvements and refinements:** The algorithms in CALPUFF have been designed to take advantage of recent improvements in scientific understanding of boundary layer meteorology, dispersion modeling, and chemistry.

The most significant advantage the CALPUFF modeling system provides, in comparison to other dispersion models (such as ISCST3) that use meteorological data from a single station, is a more realistic treatment of the wind field, including upper air data. The CALPUFF model gets the upper air data input from the Mesoscale Model, version 5 (commonly known as MM5). The MM5 model was run for Washington, Oregon, part of Idaho, and British Columbia by the University of Washington. MM5 is a prognostic model that produces gridded upper-air wind fields and is used as input into the CALPUFF model. "Gridded wind fields" indicates that the model provides wind speeds and direction at specific intervals (4 km) over the modeling region. The CALPUFF upper air input is much more comprehensive than simply using a single set of upper air data from one station. Also, note that rather than performing external calculations of the mixing height and providing these results as input into the model (when using ISCST3), CALPUFF handles those calculations internally, since it has a very comprehensive set of meteorological data as input. Surface wind regimes typically have complex, three-dimensional qualities that are significantly influenced by geophysical parameters, such as topography, so that a single-surface observation site is often not sufficient to accurately characterize the wind flow regime in a region. CALPUFF's three-dimensional wind field provides a more accurate representation of the wind flow influencing regional air quality impacts. The CALPUFF model releases the pollutant puffs into that three-dimensional wind field, which has varying wind flow patterns and accounts for complex terrain features, thereby producing a more realistic depiction of dispersion.

One of the unique characteristics of Hanford is that Battelle's Pacific Northwest National Laboratory (PNNL) operates the meteorological monitoring network in and around the Hanford site. There are 30 surface monitoring stations included in the network, which provides a comprehensive set of meteorological conditions throughout the Hanford site and in surrounding areas (8 stations are located outside the Hanford site boundary). Data from 26 of these stations will be included in the CALPUFF run to provide a very representative picture of surface meteorological conditions in the region around the WTP site.

All of the monitoring stations measure wind speed and direction at 10 m above ground level and temperature at 1 m above ground level. Other variables to be used in the modeling, including relative humidity, dew-point temperature, barometric pressure, cloud cover, and ceiling height are only measured at the main Hanford Meteorological Station, which is located near the center of the Hanford site and approximately 5 miles west of the 200 East Area location where the WTP will be located. These supplemental data are expected to be representative of atmospheric conditions at the WTP.

The CALPUFF system consists of three main components: CALMET, CALPUFF, and CALPOST. The approved versions of the CALMET, CALPUFF, and CALPOST programs was used in this analysis and is supplemented by EPA's *Risk Assessment Guidance for Superfund* (RAGS), Part A (EPA 1989), and RAGS Part B (EPA 1991) models for radionuclides. This model can handle a large number of sources

that could occur from a typical industrial source, including point sources (such as stacks) and area sources (such as fugitive emissions from an open area). In the case of the WTP, there are no fugitive emissions, and CALPUFF is used exclusively for point source emissions.

The CALPUFF model is used to calculate ambient concentrations and wet and dry deposition rates for COPCs and ROPCs at pre-determined exposure locations. The terrain elevation of each receptor is included in the model input. Terrain elevations are obtained from digitized maps of the Hanford Site for receptors located within the site or from US Geological Survey (USGS) digitized maps for receptors located outside of the site.

### 6.1.2 Detailed Discussion of CALPUFF Modeling

The following sections present an overview of the components in the CALPUFF modeling system, the application of the CALPUFF model, and post-processing of CALPUFF results to determine air quality impacts.

Sufficient data is available from a variety of sources to run the CALMET, CALPUFF, and CALPOST components. The CALMET module is used to combine various types of meteorological and geophysical data with the necessary control information into the particular format required for use in the dispersion-modeling component of the CALPUFF model. CALPOST is then used as a post-processing program to read the formatted output file generated by CALPUFF and summarize modeled results. The objective of this section is to describe the collection, preparation, and application of all data necessary to run the CALPUFF modeling system. *Hanford Tank Waste Treatment and Immobilization Plant Risk Assessment Air Quality Modeling Protocol* (Supplement 5) discusses the model settings in detail; the discussion below provides background information and highlight model settings that were modified from default values, and subsequently validated through an independent assessment and comparison of model results to actual observed weather data (24590-CM-HC4-HKYM-00001-01-00002).

#### 6.1.2.1 CALMET Modeling

The CALMET model uses a grid system consisting of square horizontal cells (*NX* by *NY*) and vertical layers (*NZ*) to create a three-dimensional wind field over a specified domain area. To develop the wind field in the domain area, the model must start with an initial “guess” field. Several options are available for initializing the wind field, including a spatially uniform guess field or objective analysis of all available weather observations; however, use of output data from a gridded prognostic model (such as Pennsylvania State’s Mesoscale Model 5 [MM5]) is preferred due to its ability to provide a spatially varying wind field and take into account geographic features influencing mesoscale wind patterns. Once defined, this initial wind field is adjusted objectively using local geophysical data and surface meteorological observations.

In addition to MM5 data, the CALMET model incorporates a variety of other meteorological and geophysical datasets in developing the three-dimensional wind fields, including upper air, surface, precipitation, terrain, and land use data. Surface and upper air observations are used to refine the MM5 predictions to account for local scale effects not resolved by the MM5 prognostic model. Inclusion of geophysical data further influences the development of the wind fields, especially in complex flow applications and light wind situations where terrain-induced flows dominate surface wind patterns. The CALMET model is used to combine MM5 simulation data with surface meteorological observations, upper air observations, and geophysical data into the format required by the dispersion-modeling component CALPUFF.

The following sections briefly discuss the preparation of the meteorological and geophysical datasets, as well as the application of the CALMET module.

#### 6.1.2.2 Preparation of Data

**MM5 Data.** A five-year subset of the University of Washington's archived MM5 data, spanning 1 January 2002 to 31 December 2006, was obtained and used in this modeling application. The data were processed using the CALMM5 module, which processes the MM5 data for direct input into the CALMET model.

**Surface Data.** Surface meteorological measurements are used in the construction of CALMET input files to supplement the MM5 wind data in defining the three-dimensional wind field. Hourly surface meteorological data was obtained for the 1 January 2002 to 31 December 2006 period from 26 of the 30 stations comprising the Hanford Meteorological Monitoring Network. These stations cover all quadrants of the Hanford Site and provide a comprehensive set of representative surface wind data for the area. All of this data is used in developing the three-dimensional wind field for each hour of the five-year modeling period. In addition, the main Hanford Meteorological Station, located near the center of the Hanford site, collects precipitation and cloud cover data that is used in the model. The stations are operated by Mission Support Alliance (MSA) on a continuous basis; MSA maintains a comprehensive quality assurance program to ensure the quality of the data collected in the Hanford Meteorological Monitoring Network.

**Integration of MM5 and Surface Data**. The three-dimensional wind field model uses a combination of upper-level MM5 data and surface data to adequately describe wind conditions at plume height. Most surface data is collected from towers at heights of 10 m; the highest surface collection height is 124 m.

**Geophysical Data.** Land use and terrain data are both incorporated into the CALMET module to modify wind field projections and, subsequently, affect dispersion calculations in the CALPUFF model. Terrain height and land use data are obtained electronically from the USGS's website (http://edcftp.cr.usgs.gov/pub/data/landcover/states/washington.nlcd.bin.gz, accessed 2010) and pre-processed using the software provided in the CALPUFF modeling system. Terrain data is available for digital elevation model data with each file covering a 1º (latitude) by 1º (longitude) area corresponding to the east or west half of a 1:250,000 (1º-latitude by 2º-longitude) topographic map. The terrain dataset's resolution varies from 70 m to 90 m in North America, with an absolute accuracy of 130 m in the horizontal and 30 m in the vertical.

Land use data is also available from the USGS's website at the 1:250,000-scale. Each land use file covers the full 1º (latitude) by 2º (longitude) area corresponding to a 1:250,000-scale topographic map with approximately 200 m resolution.

#### 6.1.2.3 CALMET Input Assumptions

The CALMET program requires inputs regarding wind characteristics and the potential influence of land terrain on wind patterns. Assumptions regarding the validity or relative importance (i.e., weight) of surface wind observations and upper air data must be programmed into CALMET to enable the model to predict conditions within a three-dimensional wind field over a specified domain area. These parameters deal with the CALMET model's treatment of surface and upper air wind data in developing the wind field (Table 6-1). Supplement 5 provides additional detail about specific model settings.

**BIAS.** BIAS is the relative weight that is given to the vertically extrapolated surface wind observations versus the upper air data. The "initial guess wind field" developed by the CALMET model is computed as an inverse distance weighting of the surface and upper air observations, modified by the height-dependent bias factors (BIAS). BIAS is a layer-dependent factor that modifies the weights of surface and upper air data. Negative BIAS reduces the weight of upper air observations (e.g., BIAS = -0.1 reduces weight of upper air data by 10 %; BIAS = -1.0 reduces weight by 100 %). Positive BIAS reduces the weight of surface stations (e.g., BIAS = 0.2 reduces the weight of surface stations by 20 %; BIAS = 1 reduces their weight by 100 %). Zero BIAS leaves weights unchanged from the inverse distance weighting function. A value of BIAS must be entered for each vertical level being modeled.

For example, upper air observations may be given little weight within a local valley, where surface observations may better reflect wind flow patterns. Similarly, upper air observations may be given heavy weight above the valley, while the surface observations can be eliminated above the valley. BIAS may be important, because the model may have difficulty in overcoming a poorly defined initial guess wind field. BIAS is used to enable CALMET to compute more accurate and smooth transitions between wind vectors such that unrealistic wind conditions are not produced. CALMET BIAS will be set at 0 (default) to represent 9 vertical levels, ranging from the surface to 4000 m.

**IEXTRP.** The vertical extrapolation of surface wind observations is provided through the variable IEXTRP. This option is used to calculate the winds at levels above the surface (typically wind speeds increase with height above the surface, and this must be taken into account). IEXTRP has four options:

- IEXTRP = 1 do not extrapolate the surface data
- IEXTRP = 2 extrapolate vertically using a power law equation
- IEXTRP = 3 extrapolate vertically using user-defined scaling factors
- IEXTRP = 4 extrapolate vertically using similarity theory

IEXTRP will be set equal to 4.

**R1.** R1 is a weighting parameter for the diagnostic wind field in the surface layer. This parameter controls the relative weighting of the first-guess wind field produced by the diagnostic wind field model and the surface layer observations. R1 is the distance from an observation station at which the observation and the first-guess wind field are equally weighted.

There is no default value provided for this parameter in the model guidance. R1 will be set equal to 10.

**R2.** R2 is a weighting parameter for the diagnostic wind field in the layers aloft. This parameter controls the relative weighting of the first-guess wind field produced by the diagnostic wind field model and the upper air observations. R2 is the distance from an observation station at which the observation and the first-guess wind field are equally weighted.

There is no default value provided for this parameter in the model guidance. R2 will be set equal to 12.

**RMAX1.** An observation is excluded from interpolation if the distance from the observation station to a particular grid point exceeds a maximum radius of influence. RMAX1 is the radius of influence over land in the surface layer (km).

This parameter should reflect the limiting influence of terrain features on the interpolation at the surface. Larger terrain features will tend to reduce RMAX1, although no default value is given in the model guidance. RMAX1 will be set equal to 12.

**RMAX2.** An observation is excluded from interpolation if the distance from the observation station to a particular grid point exceeds a maximum radius of influence. RMAX2 is the radius of influence over land in the layers aloft (km).

RMAX2 is generally larger than RMAX1 because the effects of terrain decrease with height. RMAX2 will be set equal to 12.

#### 6.1.2.4 CALMET Application

The first phase of this modeling analysis will involve the production of the three-dimensional meteorological fields to be used by the CALPUFF modeling system to characterize pollutant dispersion. The CALMET model is used to generate these wind fields, which are then input into the second module of the system, the dispersion model CALPUFF. A CALMET input file is developed to define all control information and coordinate all datasets necessary for a model run. CALMET is applied using the previously described datasets and the methods explained below.

The CALMET model will be run for a 100 km by 100 km grid with a 1 km grid mesh size and 9 vertical levels, ranging from the surface to 4000 m. The CALMET grid is centered in the middle of the Hanford site, near where the WTP facilities are to be built, so that the CALMET model grid extends approximately 50 km in all directions from the WTP facility (see Figure 6-1).

#### 6.1.2.5 CALPUFF Modeling

This section describes the preparation of the input data necessary for the second module of the CALPUFF system, the dispersion model CALPUFF. This data includes source characteristics, modeling options, and receptor locations. Air quality impacts of emissions from the proposed WTP at the Hanford site are estimated from CALPUFF model simulations using the year of CALMET-generated meteorological fields previously discussed.

Building wake effects can have a significant impact on the dispersion of emissions near a stack. The turbulence induced by buildings produces a phenomenon, known as *building downwash*, in which a stack plume can be brought downward toward the ground much sooner than if the buildings were not there, resulting in localized areas of elevated emission concentrations. The CALPUFF model has built-in algorithms to evaluate the potential for downwash.

#### 6.1.2.6 CALPUFF Model Options

The EPA has provided guidance for the operation of both the CALMET and CALPUFF models (Earth Tech Inc. 2000a, 2000b). This guidance is used to determine the most appropriate model options and settings used for these models. Some of the key options proposed for this application of the CALPUFF model are as follows:

- Wind speed profile: Industrial Source Complex model – rural
- Plume element modeled: puff

- Pasquill-Gifford dispersion curves used with other default dispersion options
- CALPUFF partial path treatment of terrain
- Transitional plume rise, stack downwash, and partial plume penetration modeled
- Default wet and dry deposition parameters for the particle and vapor deposition

The model will be run for six scenarios to determine the location of the maximum impacts, ensure that the grid is sufficiently extended to capture the worst-case depositions, and focus on areas of particular interest to the risk assessment:

- Point of maximum onsite impact (100 m receptor grid spacing), Figure 6-2
- Downwind offsite impact area (500 m receptor grid spacing), Figure 6-3
- In the vicinity of Gable Mountain (500 m receptor grid spacing), Figure 6-4
- Along the Columbia River (500 m receptor grid spacing), Figure 6-5
- Hunter/Gatherer area along the site perimeter (current exposures, 1 km receptor grid spacing), Figure 6-6
- Hunter/Gatherer area within the site interior (future exposures, 1 km receptor grid spacing), Figure 6-7

### 6.1.3 Other Modeling Parameters

This section discusses the modeling input parameters for the air dispersion and deposition modeling including emissions data, meteorological data, exposure locations, calculations of deposition rates, and model variable settings.

#### 6.1.3.1 Emissions Source Information

Identification of emission sources and quantification of emission rates for each specific COPC and ROPC are described in Section 5, Estimation of Emissions. Stack heights for the WTP have been established at about 200 feet (about 61 m). Data required for model execution, such as stack diameters, stack gas flow velocities, and stack gas temperatures, is provided in Supplement 5. The data will be updated in the PRA.

**Unit Emission Rates.** The CALPUFF model is run with unitized (normalized) 1.0 g/s emission rates for both particles and vapors from each facility stack. The vapor and particle fractions of particle-bound constituents are modeled separately in CALPUFF. There is a linear relationship between the emissions rate from a single stack and the modeled impacts (air concentrations and deposition rates) at an individual location. Therefore, the modeled impact at that location, based on a unit emissions rate from a single stack, can simply be multiplied by the actual emissions rate of an individual COPC and ROPC to determine the actual depositions. By using spreadsheets, the impacts from a specific stack can be determined for each COPC and ROPC at each location in the receptor grid.

**Analysis of Multiple Flues.** The current WTP design is based on nine flues contributing primarily to COPC and ROPC emissions, with an additional five exhaust flues that contribute primarily to stack flow rate. These flues are bundled together in their respective stacks in the PT, HLW, and LAW Facilities. In actuality, the facility stacks are structural steel lattices that support the individual flues for that facility. The PT stack contains the process vessel vent flue, the pulse jet vessel exhaust flue, and the C2, C3, and C5 exhaust flues (facility ventilation). The HLW stack contains the HLW pulse jet vessel exhaust flue,

two HLW melter flues, the IHLW storage area exhaust flue, and the C3 and C5 exhaust flues (facility ventilation). The LAW stack contains the LAW melter offgas flue, and the C3 and C5 exhaust flues (facility ventilation). The contribution of each flue (gas temperature, humidity, and flow rate) will be combined such that an "effective" stack for each facility can be modeled separately in the air dispersion modeling process (see *WTP Stack Parameters and Flow* [24590-WTP-HPC-M30T-00002]).

All air dispersion modeling information (including but not limited to input files, meteorological data, and output files) will be provided in electronic format with the risk assessment reports.

#### 6.1.3.2 Calculation of Deposition Rates

The determination of deposition rates is an important input into the human health and ecological risk assessments being conducted for the WTP. The CALPUFF model will be used to calculate both wet and dry deposition rates, in addition to ambient concentrations, at each exposure location.

Dry deposition occurs in the absence of precipitation; wet deposition is influenced by precipitation type and rate. The two types of deposition result from different physical processes and, therefore, must be considered separately. CALPUFF has algorithms built into the model to calculate these processes. CALPUFF requires the use of many parameters. The CALPUFF model was run using the Interagency Workgroup on Air Quality Modeling (IWAQM) Phase 2 recommendations, with minor exceptions. The WTP Project met with Ecology and EPA on August 15, 2007, to review model settings and assumptions. In a subsequent meeting, the WTP Project and Ecology concurred with the proposal to remove the Rattlesnake Mountain meteorological station (Station 20) from the CALMET inputs (CCN 194345). Modeling was conducted using the settings agreed to during these meetings. This CALPUFF run will be updated with the latest flue and stack design parameters before running the risk assessment model runs (PRA and FRA).

**Dry Deposition.** The CALPUFF model calculates the deposition velocity from particle diameter, mass fraction, and particle density, which are the data input into the model for each particle size-fraction. The calculation of deposition velocities within the model includes the effects of Brownian motion, inertial impaction, and gravitational settling. Particularly for the larger particles, the key parameter governing the rate of dry deposition is the terminal settling velocity. The terminal settling velocity, in turn, is affected primarily by the particle size and density; large particles have the highest terminal velocities (and, therefore, the highest deposition rates), and small particles have lower terminal velocities. It is important to note that particles have a positive terminal settling velocity and, therefore, are subject to dry deposition.

**Wet Deposition.** The wet-deposition flux is calculated by using a scavenging ratio to model the wet removal of particles and gases in the atmosphere. The scavenging coefficient appears to depend on a complex combination of the characteristics of the COPC and ROPC (such as solubility and reactivity for gases; size distribution for particles), as well as the nature of the precipitation (such as liquid or frozen). The input screens of the CALPUFF model have suggested scavenging coefficients for use in the model.

**Scavenging Coefficient for Wet Deposition.** An empirical scavenging coefficient approach is used in CALPUFF to compute the plume depletion and wet deposition fluxes due to precipitation scavenging. Generally, soluble species have higher values for scavenging coefficients than insoluble species.

This parameter may be of only limited importance at Hanford because of the small number of wet scavenging events that occur in the area on an annual basis. Scavenging coefficients for wet deposition are presented in Table 6-1.

**Deposition Rate Calculations.** The COPC and ROPC emissions can occur in either the vapor or particle phase, and COPCs and ROPCs in both phases are subject to wet and dry deposition. Particle size is a primary influence on the calculation of both dry and wet deposition of COPCs and ROPCs in the particle phase. Therefore, distribution of particle sizes in the stack emissions at the WTP is an important input parameter in the model for determining deposition rates. Particles released from the HEPA filters based on the HEPA design are projected to be no greater than 0.3 microns. Particles around 1 micron (and less) are expected to have a very low terminal velocity and are effectively suspended in air, indicating particles passing through the HEPA filters (0.3 microns) will behave similarly. A single particle size of 1 micron will be assumed to be representative for all pure particles released from the stacks because of the use of HEPA filtration. In addition, a particle size of 2.5 microns will be modeled to represent the transport of particle-bound constituents. With regard to partitioning of particle-bound constituents, the particle size in the desert environment would dominate any particle size from the stack. Particle-bound constituents would disassociate from particulates in the stack gas, and reassociate with the natural airborne particulates in the local environment (see CCN 194345). This approach to air dispersion modeling is premised by the following assumptions:

- There is sufficient 2.5 micron ambient particles to accommodate the estimated release of particle-bound constituents (for adsorption and transport purposes)
- The pure particulates released from the stack do not have a predisposition to agglomerate into larger particles as suggested for particle-bound constituents (constituents with $F_v \leq 0.05$ are treated as particulates - modeled as 1 micron)
- The vapor phase dispersion and deposition is not affected by the stack particle size

### 6.1.4 Model Output

Modeled output is provided on a stack-by-stack basis for each air modeling receptor (node) evaluated. This information will provide the means to understand the relative risks of each source and helps facilitate the management of risks. The output will also identify applicable CALPUFF grids used to evaluate depositions across a particular area (such as a water body, Gable Mountain, or riparian area). The locations of the air modeling receptors used in the risk assessment are shown graphically with designations indicating whether the node represents an air concentration or deposition value. Complete files of all the air modeling projects (including input files and output files) will also be provided in run-ready electronic format.

#### 6.1.4.1 Chronic Output

Chronic output from the WTP, to be evaluated in the risk assessment, will be based on the annual average ambient air concentrations and deposition rates for each COPC and ROPC at each exposure location, as calculated by the CALPUFF model. The annual average concentrations and deposition rates will be modeled for the period of 1 January 2002 to 31 December 2006 using the available MM5 and Hanford surface meteorological data.

#### 6.1.4.2 Acute Output

The acute output from the WTP, to be evaluated in the risk assessment, will be based on the highest one-hour average air concentrations, as required by EPA guidance (EPA 2005), for each COPC and ROPC at each exposure location, as calculated by the CALPUFF model. The use of one-hour average air

concentrations is to support the analysis of worst-case acute effects in the risk assessment. An acute inhalation scenario is recommended by EPA (2005) because it is possible for air concentrations of COPCs to significantly exceed the annual average concentration for a brief period of time and, thus, result in acute effects to receptor populations via inhalation. Because the acute effects are only due to direct inhalation, deposition rates are not important in determining the acute risk. Concentrations in soil and other media reflect long-term deposition of COPCs and ROPCs. The long-term cumulative concentration in these media will be greater than the concentration resulting from any single acute event. Therefore, the acute exposure scenario is only applicable to the inhalation pathway.

The highest one-hour average concentration will be calculated for the worst-case hour (that is, the hour with the meteorological conditions that result in the highest concentration). Acute emissions estimates include process upset and fugitive emissions in addition to normal stack emissions as described in Section 5. Acute emissions-modeling does not include accidental (i.e., catastrophic) releases. Because the concentrations required to cause acute radiation effects due to external exposures would only result from an accident scenario, this event is not considered in the acute scenario.

#### 6.1.4.3 Exposure Point Concentrations

Exposure point concentrations (EPC) used for estimating doses of COPCs and ROPCs depend on the location of the receptor exposure. The location of the various receptor populations identified for the quantitative risk assessment will correspond to the receptor exposure grids defined during air dispersion modeling. These receptor grids represent key locations on and off the Hanford site (see Section 7.1.1) where a receptor is exposed to contaminates via a pathway specific to the receptor's exposure scenario. Emissions will be modeled separately for each WTP facility (PT, LAW, and HLW). The individual flues associated with each of the facility stacks (which are in actuality structural steel lattices that support the individual flues for that facility) will be combined and modeled as three separate stacks. There are nine air model species possible from each facility stack:

- Unitized yearly air concentration from vapor phase, *Cyv* (in $\mu g \cdot s/g \cdot m^3$)
- Unitized yearly air concentration from particle phase (modeled as 1 micron diameter particles), $Cyp_1$ (in $\mu g \cdot s/g \cdot m^3$)
- Unitized yearly air concentration from particle phase (modeled as 2.5 micron diameter particles), $Cyp_{2.5}$ (in $\mu g \cdot s/g \cdot m^3$)
- Unitized yearly average dry deposition from vapor phase, *Dydv* (in $s/m^2 \cdot yr$)
- Unitized yearly average dry deposition from particle phase (modeled as 1 micron diameter particles), $Dydp_1$ (in $s/m^2 \cdot yr$)
- Unitized yearly average dry deposition from particle phase (modeled as 2.5 micron diameter particles), $Dydp_{2.5}$ (in $s/m^2 \cdot yr$)
- Unitized yearly average wet deposition from vapor phase, *Dywv* (in $s/m^2 \cdot yr$)
- Unitized yearly average wet deposition from particle phase (modeled as 1 micron diameter particles), $Dywp_1$ (in $s/m^2 \cdot yr$)
- Unitized yearly average wet deposition from particle phase (modeled as 2.5 micron diameter particles), $Dywp_{2.5}$ (in $s/m^2 \cdot yr$)

Vapor and particulate transport, concentration, and deposition are modeled independently. As described in Section 6.1.3.2, vapors will be represented by their respective model species, particulates will be

represented by the 1 micron particles, and particle-bound constituents will be represented by 2.5 micron particles. Air concentrations and wet- and dry-deposition rates for particle-bound constituents are the sum of their respective vapor and particle fractions at a given location. As discussed in Section 5.2, the vapor and particle fractions for particle-bound constituents are determined from a constituent's unique $F_v$ value. By multiplying the modeled vapor concentration or deposition value by $F_v$, the respective vapor portion of a particle-bound constituent concentration or deposition is estimated. Likewise, by multiplying the unitized 2.5 micron particle concentration or deposition value by 1-$F_v$, the respective particle-bound portion of the constituent concentration or deposition is estimated. The vapor and particle-bound portions of a particle-bound constituent are then summed to estimate the total constituent concentration or deposition. A complete list of $F_v$ values for each COPC and ROPC is included in Supplement 4 of this work plan.

There are a total of 27 possible maximum concentration and deposition values for each receptor exposure grid (3 stacks, each with 3 vapor phases [*Cyv*, *Dydv*, *Dywv*] and 3 particle phases, in either 1- or 2.5-micron sizes [$Cyp_1$, $Dydp_1$, $Dywp_1$, $Cyp_{2.5}$, $Dydp_{2.5}$, $Dywp_{2.5}$]) for each year of the air model run. It is not practical to assume a receptor can be simultaneously living at up to 27 different points within a receptor exposure grid. Some degree of simplification is used to combine concentration and deposition values while still preserving conservatism in the derived EPC.

Data produced from the CALPUFF was evaluated for basic statistical quantities, distribution, outliers, and spatial distribution in order to determine an appropriate value for use as input to the WTP environmental risk assessment (24590-WTP-RPT-ENV-13-001). The results of this evaluation indicated a high degree of non-normality in the data distribution with multiple outliers. This is of particular concern in large receptor exposure grids where the probability of exposure at a single discrete location is less probable than at a smaller, localized receptor grid. As a result, it was concluded that the exposure point concentration values from the CALPUFF modeling for input to the environmental risk assessment should be the 90th percentile values for large grids, namely the offsite receptor grid. For smaller receptor grids (onsite maximum, Columbia River, and Gable Mountain) the maximum discrete values for air concentration and deposition (as applicable to the exposure scenario) are appropriate as a means of bounding exposures at those locations while simplifying data selection. For the very large hunter/gatherer areas (where average exposures are of concern) the distribution-free 95 % upper confidence limit of the median provides a sufficiently conservative estimate of air concentration and deposition. This approach ensures sufficient conservatism without misrepresenting potential exposures due to a highly improbable exposure to extreme deposition and air concentrations.

In summary, the following CALPOST results will serve as inputs to the risk assessment:

- Onsite maximum, Columbia River, and Gable Mountain receptor grids: maximum discrete air concentration and deposition values (as applicable to the exposure scenario, without regard to multiple locations) of all modeled species (all years/grid points)
- Offsite receptor grid: 90th percentile of the air concentration and deposition values of all modeled species (all years/grid points)
- Hunter/gatherer receptor grid: 95 % upper confidence limit of the median (distribution-free) of air concentration and deposition values of all modeled species (all years/grid points)

In order to determine the appropriate incremental cancer risk or toxic effect, each contaminant must be classified as either a carcinogen or noncarcinogen. For this risk assessment, a contaminant is classified as a carcinogen if a cancer slope factor (*CSF*) is available or if the EPA classification is A, B1, B2, or C (see

Section 7.2.1.1 for more details on *CSF*s and the EPA classifications for contaminants; also, note that all ROPCs are classified as carcinogens). A COPC is classified as a noncarcinogen if an oral or inhalation reference dose (*RfD*) is available (see Section 7.2.1.1 for more details on *RfD*s) or if no *CSF* or *RfD* is available. Note that only COPCs have *RfD*s; ROPCs do not have *RfD*s (however, the stable isotope of ROPCs can have *RfD*s and they are evaluated for both carcinogenic and noncarcinogenic effects). Some contaminants may be classified as both a carcinogen and a noncarcinogen (if they have both a *CSF* and a *RfD*); in this case, both the carcinogenic and the noncarcinogenic will be used in the risk assessment. Once the EPC of a constituent has been computed, the corresponding receptor dose and health impacts are assessed as described in subsequent portions of this work plan.

## 6.2 Soil Accumulation Modeling

Concentrations of COPCs and ROPCs in soil will be estimated from deposition rates of vapor, and particle phases predicted by the air dispersion modeling. The particle and vapor fractions of particle-bound constituents will be recombined at each deposition location and handled as a single, particle-bound constituent in the risk modeling steps. For the SLRA, deposition is assumed to occur for the potential operating lifespan of the facility (40 years). The COPC and ROPC concentrations in soil will be calculated for vapor, particle, and particle-bound phases. The emissions report, included in Supplement 2 of this work plan, specifies the COPC and ROPC phases along with the constituent-specific $F_v$ parameter values. Both wet and dry deposition of particles, particle-bound, and vapor constituents will be considered in the soil modeling.

Various equations are used in the soil accumulation modeling. Some parameter values used in this modeling are functions of other parameters, which are functions of yet other parameters. To avoid confusion, the primary equations for soil accumulation modeling appear in Section 6.2; supporting/intermediary equations appear in Appendix A. A cross-reference to these supporting/intermediary equations is presented in this section.

The EPA guidance (EPA 2005) for calculating emissions concentrations in soil includes terms that account for loss of COPCs by several mechanisms, including:

- Degradation (biotic and abiotic)
- Leaching
- Surface runoff
- Volatilization
- Soil erosion

Although not mentioned in EPA guidance, radiological decay for ROPCs is comparable to degradation for COPCs and is also considered as a soil loss mechanism in the soil modeling. Therefore, all five soil-loss mechanisms will be considered as possible soil-loss mechanisms in the calculation of soil concentrations. Equations to calculate the soil loss mechanisms are located in Appendix A.

A number of soil loss parameters are dependent on the available water, calculated as ($P+I-RO-E_v$), which is related to precipitation ($P$), irrigation ($I$), surface runoff ($RO$), and evapotranspiration ($E_v$) in the Hanford site area. Climate in the region results in greater evapotranspiration than precipitation (DOE 1997). Some areas are irrigated; however, the high evapotranspiration and scarce water resources minimize the potential for runoff due to excessive irrigation. Therefore, neither natural precipitation nor

irrigation provides adequate water to generate surface runoff, and these processes should have a negligible effect on the concentration of COPCs and ROPCs in soil.

All six soil-loss mechanisms are possible, with varying degrees of influence on the soil modeling. However, based on the discussion above on available water, the calculation of soil concentrations is likely to include soil loss due to degradation (biotic and abiotic), radiological decay, leaching, and volatilization. The calculation of soil concentrations is not likely to include soil loss due to surface runoff and soil erosion. For completeness, the equations presented below and in Appendix A will include all six soil-loss mechanisms.

Because some of the soil loss mechanisms are calculated with depth-specific parameters, the total soil loss across all soil loss mechanisms shown above is depth-specific. For this risk assessment, soil concentrations are determined for three specific soil depths: tilled soil, untilled soil, and root zone soil.

The tilled soil condition assumes that deposited emissions are mixed to a tilled depth of 20 cm for plants grown in domestic scenarios (for example, produce grown by a farmer and grain and silage grown for consumption by domestic animals).

The untilled soil condition assumes emissions are deposited on the top 2 cm of soil and stay there (i.e., no mixing occurs). Untilled soil concentrations are used to calculate direct exposure to soil (such as ingestion) by human and ecological receptors, but the untilled soil depth of 2 cm is considered too shallow to estimate plant concentrations for consumption by human and ecological receptors (i.e., no plant concentrations are modeled from the untilled soil concentrations).

The root-zone soil depth is where deposited emissions are assumed to be mixed to a root-zone depth of 15 cm for exposure of invertebrates and wild plants collected by American Indian receptors and forage ingested by domestic and wild animals. Use of root zone soil concentrations for these pathways is conservative because:

- Mixing will occur naturally as a result of plant roots and digging by worms, insects, and larger animals.
- Plant roots and soil invertebrates will exist below 2 cm and, therefore, be exposed to clean soil below this depth.

For this risk assessment, the time period over which deposition may occur (denoted as $tD$) is 40 years. This represents the time period of WTP operation, during which emissions and consequential deposition occur. For soil modeling, the time period at the beginning of the WTP operation is from 0 to 40 years. Receptor exposures are assumed to occur from year $T_1$ (when the receptor arrives at the exposure location) to $T_2$ (when the receptor departs from the exposure location). Receptors that arrive at the exposure location before WTP shutdown ($T_1 < tD$) are considered part of the current exposure scenario. Receptors that arrive at the exposure location at the time of or subsequent to WTP shutdown ($T_1 \geq tD$) are considered part of the future exposure scenario.

As with EPC estimation, in order to apply the appropriate equation for soil modeling, each contaminant must be classified as either a carcinogen or noncarcinogen. This is because the exposure or dose averaging time differs depending upon whether the constituent toxicity values are based on average doses (i.e., incidence of cancer) or threshold doses (i.e., health degradation such as nervous system damage). The effective soil concentration used for computing receptor dose is a function of the receptor averaging

time, so constituent carcinogenicity is a necessary consideration in soil concentration modeling. Some contaminants may be classified as both a carcinogen and a noncarcinogen (if they have both a *CSF* and a *RfD*); in this case, both the carcinogenic soil model and the noncarcinogenic soil model will be used to estimate soil concentrations. Because carcinogenic risk is averaged over the lifetime of an individual ($AT_C$), the soil concentration averaged over the exposure duration (represented by *Cs*) is used for dose assessment for carcinogenic compounds. Because the hazard quotient associated with noncarcinogenic constituents is based on a threshold dose rather than a lifetime exposure, the highest annual average soil concentration ($Cs_{tD}$) occurring during the exposure duration period is used for dose assessment for noncarcinogenic constituents. $Cs_{tD}$ typically occurs at the end of the operating life of the emission source (EPA 2005). Note that because risks for noncarcinogens are based on a threshold dose, receptor exposure averaging time ($AT_N$) is limited to the exposure duration (*ED*).

Eight soil equations are provided below for the various scenarios encountered (i.e., the combinations of whether the contaminant is carcinogenic or noncarcinogenic, whether the soil loss constant [represented by the variable *ks*] is a positive value [meaning there is soil loss] or zero [meaning there is no soil loss], and whether exposure occurs during or after the period of emission/deposition).

The following equations are used for calculating soil concentrations, depending on whether the COPC is carcinogenic or noncarcinogenic. Parameters for soil concentration equations are defined in text that follows each equation. Supporting equations are shown in Appendix A. The equations and parameters are from EPA's HHRAP.

To compute the soil concentration, the soil deposition term (*Ds*) must first be computed. The equation to calculate *Ds* is:

$$Ds = \frac{Q \cdot CF}{Z_s \cdot BD} \cdot \left[F_v \cdot (Dydv + Dywv) + (Dydp + Dywp) \cdot (1 - F_v)\right] \qquad \text{(Table B-1-1 in HHRAP)}$$

Where:

*Ds* = deposition term to soil (mg/kg·yr). *Ds* is constituent-specific, site-specific, and depth-specific.

*Q* = constituent-specific emission rate (g/s for COPCs, Ci/s for ROPCs). *Q*, obtained from calculations after the air dispersion modeling (that is, the unitized emission rate of 1 g/s or 1 Ci/s multiplied by the estimated COPC/ROPC specific emission rate), is constituent-specific, site-specific, and flue stack-specific.

$CF_1$ = units conversion factor of 100 ($mg \cdot m^2/kg \cdot cm^2$) for COPCs. For ROPCs, the conversion factor is $1 \times 10^8$ ($pCi \cdot m^2/Ci \cdot cm^2$)

$F_v$ = fraction of COPC constituent air concentration in vapor phase (unitless). $F_v$ is constituent-specific and ranges from 0 to 1, and is shown in Supplement 4. Constituents with a vapor fraction less than 0.05 are modeled as entirely particulate with an $F_v$ value of 0 (CCN 097844). When $F_v$ is not available, it is empirically derived for most constituents (except metals and some mercury compounds) using Eqs. A-2-1 and A-2-2 (when appropriate) in the HHRAP.

*Dydv* = unitized yearly average dry deposition from vapor phase ($s/m^2 \cdot yr$). *Dydv*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dydv* value exists for a constituent, the model uses *Dydv* = 0 $s/m^2 \cdot yr$.

*Dywv* = unitized yearly average wet deposition from vapor phase ($s/m^2 \cdot yr$). *Dywv*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dywv* value exists for a constituent, the model uses *Dywv* = 0 $s/m^2 \cdot yr$.

*Dydp* = unitized yearly average dry deposition from particle phase ($s/m^2 \cdot yr$). *Dydp*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dydp* value exists for a constituent, the model uses *Dydp* = 0 $s/m^2 \cdot yr$.

*Dywp* = unitized yearly average wet deposition from particle phase ($s/m^2 \cdot yr$). *Dywp*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dywp* value exists for a constituent, the model uses *Dywp* = 0 $s/m^2 \cdot yr$.

$Z_s$ = soil mixing zone depth (cm). $Z_s$ is site-specific. Three different values (depths) are used for $Z_s$: untilled soil (2 cm), root-zone soil (15 cm), and tilled soil (20 cm).

*BD* = soil bulk density ($g/cm^3$). A site-specific value of 1.3 $g/cm^3$ (Halvorson et al. 1998) is used.

The soil term equation combines the unitized stack deposition rate with the mass flow rate of constituents from the stack and the quantity of soil to arrive at a time-dependent soil concentration. For constituents that undergo soil loss ($ks > 0$), this concentration is increasing due to continued stack deposition during WTP operations, while simultaneously decreasing due to soil loss. After WTP shutdown, constituent accumulation in the soil stops and the loss continues. The soil loss is an exponential function of the soil deposition term. In instances where there is no soil loss ($ks = 0$), soil concentration is directly proportional to the rate of deposition and time, and reaches a maximum when deposition ceases (at time *tD*).

As previously discussed, the hazard quotient associated with noncarcinogenic constituents is based on a threshold dose rather than a lifetime exposure. Per guidance in the HHRAP, the highest annual soil concentration ($Cs_{tD}$) occurring during the exposure duration period is used for dose assessment for noncarcinogenic constituents. For this to be the case, it is assumed that all receptors exposed in the current timeframe are present at time *tD* when soil accumulation is at a maximum. Likewise, it is assumed that all receptors exposed in the future timeframe are assumed to arrive at their respective exposure locations at time *tD*.

For noncarcinogenic constituents, when the soil loss is zero or unknown, current and future exposure scenarios are:

$RfD > 0$

$ks = 0$

$T_1 < tD$ or $tD < T_1$

$$Cs_{tD} = Ds \cdot tD \qquad \text{(modified Eq. 5-1B in HHRAP)}$$

For noncarcinogenic constituents, when the soil loss is greater than zero, current and future exposure scenarios are as follows:

$RfD > 0$

$ks > 0$

$T_1 < tD$ or $tD < T_1$

$$Cs_{tD} = \frac{Ds \cdot \left[1 - e^{(-ks \cdot tD)}\right]}{ks} \qquad \text{(Eq. 5-1E in HHRAP)}$$

where:

| | | |
|---|---|---|
| $Cs_{tD}$ | = | maximum soil concentration; occurs at time *tD* (mg/kg soil). |
| *Ds* | = | deposition term to soil (mg/kg·yr). *Ds* is constituent-specific, site-specific, and depth-specific. |
| *ks* | = | overall soil loss constant due to all processes ($yr^{-1}$). |
| $T_1$ | = | the time at the start of exposure (yr). |
| $T_2$ | = | the time at the end of exposure (yr). |
| *tD* | = | time period over which deposition occurs (time period of WTP operation) (yr). A value of *tD* = 40 yr is used as the operating lifetime of the WTP. |
| *e* | = | base of the natural logarithm (unitless). $e = \sum_{i=0}^{\infty} \frac{1}{i!} \approx 2.718282$. |

Because carcinogenic risk is averaged over the lifetime of an individual, the average soil concentration (represented by *Cs*) over the exposure duration (from $T_1$ to $T_2$) is used for dose assessment for carcinogenic compounds. Because soil concentrations may require many years to reach steady state, the equations used to calculate the average soil concentration over the period of receptor exposure are derived by integrating the instantaneous soil concentration equation over the period of receptor exposure and dividing the result by the exposure period (refer to Appendix A). Furthermore, during the time period following the cessation of WTP emissions (e.g., future scenarios), soil concentrations decline gradually due to various soil loss mechanisms and may require many more years to reach steady state. Again, integrating the instantaneous soil concentration equation over the period of exposure and dividing by the exposure period will yield an average exposure concentration for the receptor. Because the function for soil concentration changes from accumulation to degradation when emissions cease, exposures before and after WTP shutdown must be distinguished for carcinogens.

For carcinogenic constituents, when the soil loss is zero or unknown, current exposures scenarios:

$CSF > 0$

$ks = 0$

$T_1 < T_2 = tD$

$$Cs = \frac{Ds \cdot (tD + T_1)}{2} \qquad \text{(modified Eq. 5-1B in HHRAP)}$$

For carcinogenic constituents, when the soil loss is greater than zero, current exposures scenarios:

$CSF > 0$

$ks > 0$

$T_1 < T_2 = tD$

$$Cs = \frac{Ds}{ks \cdot (tD - T_1)} \cdot \left( \left( tD + \frac{e^{-ks \cdot tD}}{ks} \right) - \left( T_1 + \frac{e^{-ks \cdot T_1}}{ks} \right) \right) \qquad \text{(Eq. 5-1C in HHRAP)}$$

For carcinogenic constituents, when the soil loss is zero or unknown, future exposures scenarios:

$CSF > 0$

$ks = 0$

$T_1 = tD < T_2$

$$Cs = Ds \cdot tD \qquad \text{(modified Eq. 5-1B in HHRAP)}$$

For carcinogenic constituents, when the soil loss is greater than zero, future exposures scenarios:

$CSF > 0$

$ks > 0$

$T_1 = tD < T_2$

$$Cs = \frac{Cs_{tD}}{ks \cdot (T_2 - tD)} \cdot \left( 1 - \frac{e^{-ks \cdot (T_2 - tD)}}{ks} \right) \qquad \text{(modified Eq. 5-1C in HHRAP)}$$

where:

| | | |
|---|---|---|
| $Cs$ | = | average soil concentration; maximum occurs at time $tD$ (mg/kg soil or pCi/g). |
| $Cs_{tD}$ | = | soil concentration at time $tD$ ($Cs_{tD} = Ds \cdot (1-e^{-ks \cdot tD})/ks$), assuming no soil loss (mg COPC/kg soil). |
| $Ds$ | = | deposition term to soil (mg/kg-yr). $Ds$ is constituent-specific, site-specific, and depth-specific. |
| $ks$ | = | overall soil loss constant due to all processes ($yr^{-1}$). |
| $T_1$ | = | the time at the start of exposure (yr). |
| $T_2$ | = | the time at the end of exposure (yr). |
| $tD$ | = | time period over which deposition occurs (time period of WTP operation) (yr). A value of $tD$ = 40 yr is used as the operating lifetime of the WTP. |
| $e$ | = | base of the natural logarithm (unitless). $e = \sum_{i=0}^{\infty} \frac{1}{i!} \approx 2.718282$. |

If the exposure period spans the period of operation and a period of time subsequent to operations, the average soil concentration will include exposure to soil during WTP operations (the period of contaminant accumulation), and exposure to soil after WTP operation (during contaminant degradation).

For carcinogenic constituents, when the soil loss is zero or unknown, exposures spanning current and future scenarios:

$CSF > 0$

$ks = 0$

$T_1 < tD < T_2$

$$Cs = \frac{Ds}{2 \cdot (T_2 - T_1)} \cdot \left(2 \cdot T_2 \cdot tD - tD^2 - T_1^2\right) \qquad \text{(modified Eq. 5-1B in HHRAP)}$$

For carcinogenic constituents, when the soil loss is greater than zero, exposures spanning current and future scenarios:

$CSF > 0$

$ks > 0$

$T_1 < tD < T_2$

$$Cs = \frac{\left(\frac{Ds \cdot tD - Cs_{tD}}{ks}\right) + \left(\frac{Cs_{tD}}{ks}\right) \cdot \left(1 - e^{-ks \cdot (T_2 - tD)}\right)}{(T_2 - T_1)} \qquad \text{(Eq. 5-1D in HHRAP)}$$

where:

| | | |
|---|---|---|
| $Cs$ | = | average soil concentration; maximum occurs at time $tD$ (mg/kg soil or pCi/g). |
| $Cs_{tD}$ | = | soil concentration at time $tD$ ($Cs_{tD} = Ds \cdot (1 - e^{-ks \cdot tD})/ks$), assuming no soil loss (mg/kg soil or pCi/g). |
| $Ds$ | = | deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific. |
| $ks$ | = | overall soil loss constant due to all processes ($yr^{-1}$). |
| $T_1$ | = | the time at the start of exposure (yr). |
| $T_2$ | = | the time at the end of exposure (yr). |
| $tD$ | = | time period over which deposition occurs (time period of WTP operation) (yr). A value of $tD$ = 40 yr is used as the operating lifetime of the WTP. |
| $e$ | = | base of the natural logarithm (unitless). $e = \sum_{i=0}^{\infty} \frac{1}{i!} \approx 2.718282$. |

Figure 6-8 shows the exposure timing of the receptors discussed in Section 7 with respect to instantaneous and running average soil concentration levels of a hypothetical COPC. The figure illustrates the conservative assumption regarding the timing of receptor exposures. The figure shows two examples of soil concentration (with and without soil loss) to illustrate the time dependence of soil concentrations relative to WTP operations. The figure shows the time of increase and subsequent leveling off of the instantaneous soil concentration used for noncarcinogen assessment, as represented by the blue line. No known soil loss occurs in this case so the concentration is represented by a straight line that reaches a maximum, *CstD*, at the cessation of operations with no post-operations losses. The green line

illustrates the instantaneous soil concentration in the soil as deposition occurs simultaneous with soil loss, again with a maximum soil concentration at the end of operations. The figure also shows the average soil concentration (for example, carcinogen assessment) without and with soil loss occurring. The magenta line mimics the blue line, only it has half the magnitude because it represents an average concentration over time. The red line shows gradual accumulation of a contaminant in the soil as deposition occurs, with simultaneous soil loss, and post-operation soil loss. Because it represents an average exposure concentration from time = 0, its decrease after the cessation of operations is much more gradual than the instantaneous soil concentration (with loss). As applied, the equations above are used to determine the maximum potential exposure concentration of each receptor. Appendix A of this document provides additional detail including derivation of the equations above.

## 6.3 Surface Water Accumulation Modeling

Concentrations of COPCs and ROPCs in surface water will be estimated from EPCs described in Section 6.1.4.3 as derived from the Columbia River air modeling exposure grid (see Section 6.1.2.6). For this risk assessment, deposition is assumed to occur for the potential operating lifespan of the facility (40 years). The COPC and ROPC concentrations in surface water (water in a pond, stream, river, or other water body, that is, the Columbia River) are calculated for vapor, particle, and particle-bound phases. The emissions report, included in Supplement 2 of this work plan, specifies the COPC and ROPC phases along with the constituent-specific $F_v$ parameter values. Both wet and dry deposition of particles, particle-bound, and vapor constituents will be considered in the surface water modeling. Note that for evaluation of future exposure scenarios (after cessation of emissions), air concentration and deposition rates are zero; thus, no surface water accumulation occurs.

Various equations are used in the surface water accumulation modeling. Some parameter values used in this modeling are functions of other parameters, which are functions of yet other parameters. To avoid confusion, the primary equations for surface water accumulation modeling appear in Section 6.3; supporting/intermediary equations appear in Appendix A. A cross-reference to these supporting/intermediary equations is presented in this section.

The COPC and ROPC concentrations in surface water will be calculated for the drinking water, dermal contact, and fish ingestion pathways in the human health risk assessment, and the direct contact (aquatic life and fish) and indirect ingestion pathways for ecological receptors. The COPC and ROPC surface water concentrations are determined after considering the following mechanisms loaded into the water column (i.e., a volume of water of uniform horizontal cross-section that extends from the surface to the bottom of the water body):

- Direct deposition
- Direct diffusion of vapor phase COPCs and ROPCs into the surface water
- Runoff from impervious surfaces within the watershed (that is, the area potentially contributing water to the Columbia River)
- Runoff from pervious surfaces within the watershed
- Soil erosion over the total watershed
- Chemical, biological, or radiological transformation of compounds within the surface water body

As noted previously in Section 6.2, evapotranspiration exceeds precipitation in the Hanford site area, resulting in insufficient water available to cause significant erosion or runoff of COPCs and ROPCs

(since evapotranspiration exceeds precipitation, there is no water to run off; the water goes back up into the air). Thus, surface runoff and soil erosion are expected to be insignificant soil loss mechanisms and insignificant surface water loading mechanisms. Since surface runoff and erosion to the Columbia River are assumed to be negligible, a watershed area is provided for information, but not used. Also, since the maximum constituent concentrations in surface water and sediment will be used as inputs for these pathways, a surface water area (or "effective" area) is provided for information, but will not be used. Therefore, surface runoff and soil erosion will not be included as surface water loading mechanisms unless they are included as soil loss mechanisms (note that EPA 2005 recommends that the soil loss due to soil erosion should not be included in the soil accumulation modeling). Also, as noted in EPA 2005, the chemical, biological, or radiological transformation of compounds within the surface water body should not be included as a load to the surface water body because of limited data and uncertainty associated with this mechanism.

Therefore, contaminant loading to surface water for the PRA will be from direct deposition and vapor phase dry deposition diffusion. For completeness, the equations presented below include all potential surface water loading mechanisms.

The COPCs and ROPCs in surface water will be estimated using equations presented below. These equations are from EPA 2005; however, because this guidance does not address ROPCs, minor changes (e.g., the use of unit conversion factors) have been made to these equations to address ROPCs. Supporting and intermediate equations are presented in Appendix A of this work plan. Values for the Hanford-specific and site-specific parameters used in surface water modeling are presented in Table 6-3. Values for the COPC- and ROPC-specific parameters are presented in Supplement 4.

The site-specific equation used to quantify the total COPC and ROPC load to the surface water body is:

$$L_T = L_{DEP} + L_{DIF} + L_{RI} + L_R + L_E \qquad \text{(Eq.5-28 in HHRAP)}$$

where:

$L_T$ = total COPC or ROPC load to the water body, including deposition, runoff, and erosion (g/yr for COPCs and Ci/yr for ROPCs). Note that because there are three facility stacks, $L_T$ is calculated for each individual stack before summing across all three stacks to obtain a total direct deposition load to the water body.

$L_{DEP}$ = total (wet and dry) particle-phase and total (wet and dry) vapor-phase direct deposition load to water body (g/yr for COPCs and Ci/yr for ROPCs). $L_{DEP}$ is calculated in Eq. 5-29 of the HHRAP.

$L_{DIF}$ = vapor-phase dry deposition diffusion load to water body (g/yr for COPCs and Ci/yr for ROPCs). $L_{DIF}$ is calculated in Eq. 5-30 of the HHRAP.

$L_{RI}$ = runoff load from impervious surfaces (g/yr for COPCs and Ci/yr for ROPCs). $L_{RI}$ is calculated in Eq. 5-31 of the HHRAP, but is assumed to equal zero for this risk assessment.

$L_R$ = runoff load from pervious surfaces (g/yr for COPCs and Ci/yr for ROPCs). $L_{RP}$ is calculated in Eq. 5-32 of the HHRAP, but is assumed to equal zero for this risk assessment.

$L_E$ = soil erosion load to the surface water body (g/yr for COPCs and Ci/yr for ROPCs). $L_E$ is calculated in Eq. 5-33 of the HHRAP, but is assumed to equal zero for this risk assessment.

Once the total load to the water body ($L_T$) is estimated, the total water body COPC or ROPC concentration ($C_{wtot}$) will be calculated. This total water body concentration is subsequently used to estimate the total concentration in the water column (see below), as well as the concentration adsorbed to the bed sediment (see Section 6.4). The equation used to estimate the total water body concentration for COPCs is:

$$C_{wtot} = \frac{L_T}{Vf_x \cdot f_{wc} + k_{wt} \cdot A_W \cdot (d_{wc} + d_{bs})} \qquad \text{(Eq. 5-35 in HHRAP)}$$

Where:

$C_{wtot}$ = total water body COPC or ROPC concentration, including the water column and bed sediment (mg/L for COPCs and pCi/L for ROPCs). Note that for ROPCs, a unit conversion factor of $1 \times 10^9$ pCi·m$^3$/Ci·L must be applied.

$L_T$ = total COPC or ROPC load to the water body (g/yr for COPCs and Ci/yr for ROPCs). $L_T$ is calculated in Eq. 5-28 in the HHRAP.

$Vf_x$ = average annual volumetric flow rate through the water body (m$^3$/yr). $Vf_x$ is site-specific. A value of $Vf_x$ = 1.05961E+11 m$^3$/yr (PNNL 2006, based on 3360 m$^3$/s for Priest Rapids Dam) (Table 6-3).

$f_{wc}$ = fraction of the total water body COPC or ROPC concentration in the water column (unitless). $f_{wc}$ ranges from 0 to 1 and is calculated in Eq. 5-36A in the HHRAP.

$k_{wt}$ = overall total water body COPC or ROPC dissipation rate constant (1/yr). $k_{wt}$ is calculated in Eq. 5-38 in the HHRAP.

$A_w$ = average annual water body surface area (m$^2$). A value of $A_w$ = 3.642E+07 m$^2$ is used based on all of the Columbia River within the Hanford Site boundary (PNNL 2006) (Table 6-3 and Figure 3-1).

$d_{wc}$ = average annual depth of the water column (m). An estimated value of $d_{wc}$ = 28.4 ft (8.66 m) (modeling data from Columbia Basin Research 2000, Hanford Reach full pool depth at the downstream end of the segment) is used (Table 6-3).

$d_{bs}$ = depth of the upper benthic sediment layer (m). The recommended default value of 0.03 m (HHRAP Section 5.4.7) is used (Table 6-3).

Once the total water body COPC and ROPC concentration ($C_{wtot}$) is estimated, the total COPC and ROPC concentration in the water column ($C_{wctot}$) will be calculated. This total concentration in the water column will subsequently be used to estimate the dissolved-phase water concentration (see below) and to model direct contact (aquatic life and trout) and water ingestion exposure in the ecological risk assessment. The total concentration in the water column includes both dissolved COPCs and ROPCs and COPCs and ROPCs sorbed to suspended solids. The equation used to estimate the total concentration in the water column is:

$$C_{wctot} = f_{wc} \cdot C_{wtot} \cdot \left( \frac{d_{wc} + d_{bs}}{d_{wc}} \right) \qquad \text{(Eq. 5-45 in HHRAP)}$$

where:

$C_{wctot}$ = total COPC or ROPC concentration in the water column (mg/L for COPCs and pCi/L for ROPCs)

$f_{wc}$ = fraction of the total water body COPC or ROPC concentration in the water column (unitless). $f_{wc}$ ranges from 0 to 1 and is calculated in Eq. 5-36A in the HHRAP.

$C_{wtot}$ = total water body COPC or ROPC concentration, including the water column and bed sediment (mg/L for COPCs and pCi/L for ROPCs). $C_{wtot}$ is calculated in Eq. 5-35 in the HHRAP

$d_{wc}$ = average annual depth of the water column (m). An estimated value of $d_{wc}$ = 28.4 ft (8.66 m) (modeling data from Columbia Basin Research 2000, Hanford Reach full pool depth at the downstream end of the segment) is used (Table 6-3).

$d_{bs}$ = depth of the upper benthic sediment layer (m). The recommended default value of 0.03 m (HHRAP Section 5.4.7 is used (Table 6-3).

Once the total COPC and ROPC concentration in the water column ($C_{wctot}$) is estimated, the dissolved phase COPC and ROPC water concentration ($C_{dw}$) will be calculated. The equation for this concentration is:

$$C_{dw} = \frac{C_{wctot}}{1 + Kd_{sw} \cdot TSS \cdot CF} \qquad \text{(Eq. 5-46 in HHRAP)}$$

where:

$C_{dw}$ = dissolved-phase water concentration (mg/L for COPCs and pCi/L for ROPCs)

$C_{wctot}$ = total COPC or ROPC concentration in the water column (mg/L for COPCs and pCi/L for ROPCs). $C_{wctot}$ is calculated in Eq. 5-45 in the HHRAP.

$Kd_{sw}$ = suspended sediments/surface water partition coefficient (L/kg). $Kd_{sw}$ is shown in Supplement 4. If no $Kd_{sw}$ value exists for an organic constituent, then $Kd_{sw}$ is estimated using Eq. A-2-11 in the HHRAP and a default $f_{oc,sw}$ = 0.075 (fraction of organic carbon in suspended sediments, HHRAP Section A2-2.10) provided the constituent $K_{oc}$ value (soil organic carbon-water partition coefficient) is known. If $Kd_{sw}$ is not available and cannot be estimated, a value of 0 L/kg is used for $Kd_{sw}$ to estimate $C_{dw}$.

$TSS$ = total suspended solids concentration (mg/L). Since a site-specific value is not available, a default value of 10 mg/L (HHRAP Section 5.7.4.1) is used (Table 6-3).

$CF$ = units conversion factor of $1 \times 10^{-6}$ (kg/mg)

The dissolved-phase COPC and ROPC water concentration ($C_{dw}$) will be used in the human health risk assessment as the source of drinking water, the source of water for the sweat lodge exposure pathway, and, depending on the constituent, for the modeling of fish concentrations (see Section 7.1.7.5). $C_{dw}$ is used for the modeling of fish concentrations for all COPCs and ROPCs in the ecological risk assessment.

## 6.4 Sediment Accumulation Modeling

River sediment concentrations are modeled using the previously modeled total water body concentrations (see Section 6.3). Sediment concentrations are used in the ecological risk assessment (ERA) and to model fish concentrations for specific COPCs for the human health risk assessment (see Section 7.1.7.5).

Various equations are used in the sediment accumulation modeling. Some parameter values used in this modeling are functions of other parameters, which are functions of yet other parameters. To avoid confusion, the primary equations for sediment accumulation modeling appear in Section 6.4; supporting/intermediary equations appear in Appendix A. A cross-reference to these supporting/intermediary equations is presented in this section.

The equation for calculating COPC and ROPC concentrations sorbed to bed sediment is:

$$C_{sed} = f_{bs} \cdot C_{wtot} \cdot \left( \frac{Kd_{bs}}{\theta_{bs} + Kd_{bs} \cdot C_{BS}} \right) \cdot \left( \frac{d_{wc} + d_{bs}}{d_{bs}} \right) \qquad \text{(Eq. 5-47 in HHRAP)}$$

where:

- $C_{sed}$ = COPC or ROPC concentration in bed sediment (mg/kg for COPCs and pCi/g for ROPCs). Note that a unit conversion factor of $1 \times 10^{-3}$ kg/g is used for ROPCs.
- $f_{bs}$ = fraction of total water body COPC or ROPC concentration in the benthic sediment. (unitless); $f_{bs}$ ranges from 0 to 1 and is calculated in Eq. 5-36B in the HHRAP.
- $C_{wtot}$ = total water body COPC or ROPC concentration, including the water column and bed sediment (mg/L for COPCs and pCi/L for ROPCs). $C_{wtot}$ is calculated in Eq. 5-35 in the HHRAP.
- $Kd_{bs}$ = bed sediment/sediment pore water partition coefficient (L/kg). $Kd_{bs}$ is shown in Supplement 4. If no $Kd_{bs}$ value exists for an organic constituent, then $Kd_{bs}$ is estimated using Eq. A-2-12 in the HHRAP and a default $f_{oc,bs}$ = 0.04 (fraction of organic carbon in bottom sediments, HHRAP Section A2-2.10), provided the constituent $K_{oc}$ value (soil organic carbon-water partition coefficient) is known. If no $Kd_{bs}$ value exists for a constituent, and if $Kd_{bs}$ cannot be estimated, a value of 0 L/kg is used.
- $\theta_{bs}$ = bed sediment porosity ($L_{pore\ water}/L_{sediment}$). The recommended default value of 0.6 L/L (EPA 2005) is used (Table 6-3).
- $C_{BS}$ = bed sediment concentration (g/cm$^3$). The recommended default value of 1 g/cm$^3$ (HHRAP Section 5.7.4.1) is used (Table 6-3).
- $d_{wc}$ = average annual depth of water column (m). An estimated value of $d_{wc}$ = 28.4 ft (8.66 m) (modeling data from Columbia Basin Research 2000, Hanford Reach full pool depth at the downstream end of the segment) is used (Table 6-3).
- $d_{bs}$ = depth of upper benthic sediment layer (m). The recommended default value of 0.03 m (HHRAP Section 5.4.7) is used (Table 6-3).

## 6.5 Special Considerations for Mercury Modeling

Note that special equations for mercury modeling of each of these load parameters are stipulated in the HHRAP and provided in Appendix A. The HHRAP (EPA 2005) and the *Screening Level Ecological*

*Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA 1999, herein referred to as the SLERAP) assume that stack emissions contain a variable mix of elemental and divalent mercury, but no methyl mercury. These guidance sources state that mercury emissions are partitioned in the atmosphere in both the elemental ($Hg^0$) and divalent ($Hg^{2+}$) form, with part of the mercury going into the global cycle, and another part subject to inhalation or uptake in environmental media (soil, water, and plants), and it should be assumed that 48 % of the divalent mercury and 0.2 % of the elemental mercury is deposited. However, mercury has been detected in some tank waste sample analyses and dimethylmercury (DMHg) has been detected in tank headspace samples in very small quantities. Some DMHg may be present in the waste feed, and some could be generated in plant processes, resulting in non-zero concentrations in plant emissions.

In the RAWP, it is assumed that stack emissions of mercury will be in the elemental, divalent and dimethyl form. The emissions estimate provides emissions of non-methyl mercury and dimethyl mercury. The partitioning of non-methyl form of mercury into elemental and divalent forms will be performed according to the HHRAP. However, the assumption is made that DMHg emissions do not enter the global cycle, transform (into other forms) or decay, and that 100 % of the DMHg is available for inhalation and uptake by environmental media. This is a conservative approach because in all likelihood, a substantial portion of any DMHg emitted will become part of the global cycle. Likewise, by this approach, no credit is taken for decay, oxidation, or other transformation of DMHg in the atmosphere.

Per HHRAP guidance, it is assumed that 48 % of the non-methyl mercury emitted will be deposited (Appendix A, or equations in Table B-1-1 [land], and Tables B-4-8 through B-4-12 [surface water] of the HHRAP). A portion of the non-methyl mercury emissions is assumed to convert into a mono-methyl form through interaction with organic media upon deposition. It is assumed that methyl mercury (MHg) is formed only after deposition to soil or surface water. Per EPA guidance (EPA 2005 and 1999), it is assumed that the fraction of methyl mercury in dry soil is 2 % (Appendix A, or equations in Table B-1-1 in the HHRAP) and the fraction of methyl mercury in surface water is 15 % (HHRAP Table B-4-24). Note also that because there are three facility stacks, each load type will be calculated for each individual stack before summing across all three stacks to obtain a total load.

Figure 6-9 is an illustration summarizing the assumptions with regards to mercury partitioning.

## 6.6 Terrestrial Plant Accumulation Modeling

The models used to calculate concentrations of contaminants in plants consumed by both human and nonhuman receptors will be the same. The use of the same models for human and nonhuman receptors is based on previous stakeholder and tribal nations' requests. Plants, such as homegrown vegetables or wild produce, are consumed by humans and animals (e.g., as forage for browsing animals and as silage).

Various equations are used in the terrestrial plant accumulation modeling. Some parameter values used in this modeling are functions of other parameters, which are functions of yet other parameters. To avoid confusion, the primary equations for terrestrial plant accumulation modeling appear in Section 6.6; supporting/intermediary equations appear in Appendix A. A cross-reference to these supporting/intermediary equations is presented in this section.

The COPC and ROPC concentrations in plants will be estimated for aboveground produce and belowground produce. Aboveground produce will be exposed to particulate deposition (i.e., direct deposition onto the plant surfaces) and vapor phase contamination (i.e., air-to-plant transfer), as well as root uptake from soil and subsequent transfer to aboveground foliage. Aboveground plant parts are categorized

as protected (i.e., the plant structure prevents accumulation of contaminants through the deposition and air-to-plant pathways) and unprotected. For example, corn kernels are protected by husks. Protected plant parts will be limited in this evaluation to grain used as animal feed. All other plant parts for human and animal consumption will be considered unprotected (i.e., not physically shielded from deposition). Belowground produce will only be exposed to contaminants from the soil through root uptake.

Concentrations of COPCs and ROPCs in plants will be estimated using the equations presented below as recommended in the HHRAP. Plant modeling for carbon-14 and hydrogen-3 (tritium) are special cases, based on guidance from Regulatory Guide 1.109 (NRC 1977) and are discussed in Section 6.6.2. Note that for all COPCs and ROPCs except carbon-14 and tritium, concentrations for various types of plants (e.g., aboveground plant due to direct deposition, belowground plant due to root uptake) are modeled. For carbon-14 and tritium, a single "concentration in vegetation" is modeled and used in the subsequent risk assessment. Values for site-specific parameters used in plant modeling are located in Table 6-4, while values for the chemical-specific parameters are presented in Supplement 4.

### 6.6.1 Aboveground Plants/Direct Deposition

The equations used to estimate the aboveground plant concentration due to direct deposition are presented below. Special consideration is given to modeling for total mercury, divalent mercury, and methyl mercury. No estimates of aboveground plant concentration due to direct deposition will be made for carbon-14 and tritium (see Section 6.6.2). The aboveground plant concentrations due to direct deposition will be estimated for the following plant types: produce, forage, and silage.

The following equation calculates the aboveground plant concentration due to direct deposition for all COPCs except total mercury, divalent mercury, and methyl mercury, and for all ROPCs except carbon-14 and tritium:

$$Pd = \frac{CF \cdot Q \cdot (1 - F_v) \cdot [Dydp + (Fw \cdot Dywp)] \cdot Rp \cdot [1 - e^{(-kp \cdot Tp)}]}{Yp \cdot kp} \qquad \text{(Eq. 5-14 in HHRAP)}$$

The equation to calculate the aboveground plant concentration due to direct deposition for total mercury is:

$$Pd_{(Hg)} = \frac{0.48 \cdot CF \cdot Q \cdot (1 - F_v) \cdot [Dydp + (Fw \cdot Dywp)] \cdot Rp \cdot [1 - e^{(-kp \cdot Tp)}]}{Yp \cdot kp} \qquad \text{(Table B-2-7 in HHRAP)}$$

where:

$Pd$ = concentration of COPC or ROPC in aboveground plant due to direct (wet and dry) deposition (mg COPC/kg plant tissue DW and pCi ROPC/g plant tissue DW).

$Pd_{(Hg)}$ = concentration of total mercury in aboveground plant due to direct (wet and dry) deposition (mg COPC/kg plant tissue DW).

$CF$ = units conversion factor of 1000 (mg/g) for COPCs and $1 \times 10^9$ (pCi-kg/Ci-g) for ROPCs.

*Q* = COPC or ROPC-specific emission rate (g/s for COPCs and Ci/s for ROPCs), derived as described in Section 5.

$F_v$ = fraction of COPC or ROPC air concentration in vapor phase (unitless). $F_v$ is constituent-COPC-specific, ranges from 0 to 1, and is shown in Supplement 4. Constituents with a vapor fraction less than 0.05 are modeled as entirely particulate with an $F_v$ value of 0 (CCN 097844). When $F_v$ is not available, it is empirically derived for most constituents (except metals and some mercury compounds) using Eqs. A-2-1 and A-2-2 (when appropriate) in the HHRAP.

*Dydp* = unitized yearly average dry deposition from particle phase ($s/m^2 \cdot yr$). *Dydp*, from the air dispersion modeling, is stack-specific.

*Fw* = fraction of COPC or ROPC wet deposition that adheres to plant surfaces (unitless). A value of 0.2 is used for anions and two specific organic COPCs (p-chloroaniline and n-nitrosodi-n-propylamine) that ionize to anionic forms. A value of 0.6 is used for cations and all other organics (HHRAP Section 5.3.1). See Table 6-4.

*Dywp* = unitized yearly average wet deposition from particle phase ($s/m^2 \cdot yr$). *Dywp*, from the air dispersion modeling, is stack-specific.

*Rp* = interception fraction of the edible portion of plant for aboveground produce (unitless). *Rp* is plant-type-specific, with a value of 0.39 (representing a weighted average of fruits and vegetables [HHRAP Section 5.3.1.1]) used for produce, a value of 0.05 for forage, and a value of 0.46 (HHRAP Section 5.4.1.1) for silage. See Table 6-4.

*e* = base of the natural logarithm (unitless). $e = \sum_{i=0}^{\infty} \frac{1}{i!} \approx 2.718282$.

*kp* = plant surface loss coefficient ($yr^{-1}$). The recommended default value of 18 $yr^{-1}$ (HHRAP Section 5.3.1.2) is used for all COPCs. For ROPCs, the effective *kp* includes a radioactive decay component (see Table 6-4).

*Tp* = length of plant exposure to deposition per harvest of edible portion of plant for aboveground produce (yr). The recommended default values of 0.164 yr for produce (HHRAP Section 5.3.1.3), 0.12 yr for forage, and 0.16 yr for silage (HHRAP Section 5.4.1.3) are used (Table 6-4).

*Yp* = yield or standing crop biomass of the edible portion of the plant for aboveground produce (productivity) ($kg/m^2$). *Yp* is site-specific and plant-type-specific. The recommended default value of 2.24 $kg/m^2$ (representing a weighted average of fruits and vegetables [HHRAP Section 5.3.1.4]) is used for produce, while a value of 0.15 $kg/m^2$ for forage (site-specific value, [Wisiol 1984]), and a value of 0.8 $kg/m^2$ (HHRAP Section 5.4.1.4) is used for silage. See Table 6-4.

0.48 = multiplier for modeling of total mercury (unitless), as shown in Table B-2-7 in the HHRAP.

The effective plant surface loss coefficient for ROPCs includes a component for wind removal, water removal, and growth dilution (14.06 day half-life), and a component for loss due to radioactive decay (isotope nuclear half-life). Equation 5-15 of the HHRAP is used to determine the effective plant surface loss coefficient; however, the radionuclide half-life plus 14.06 days is substituted for the term $t_{½}$ in Equation 5-15 of the HHRAP.

The following equation calculates the aboveground plant concentration due to direct deposition for divalent mercury:

$$Pd_{(Hg^{2+})} = 0.78 \cdot Pd_{(Hg)} \qquad \text{(Table B-2-7 in HHRAP)}$$

The equation to calculate the aboveground plant concentration due to direct deposition for methyl mercury is:

$$Pd_{(MHg)} = 0.22 \cdot Pd_{(Hg)} \qquad \text{(Table B-2-7 in HHRAP)}$$

where:

| | | |
|---|---|---|
| $Pd_{(Hg^{2+})}$ | = | concentration of divalent mercury in aboveground produce due to direct (wet and dry) deposition (mg COPC/kg plant tissue DW) |
| $Pd_{(MHg)}$ | = | concentration of methyl mercury in aboveground produce due to direct (wet and dry) deposition (mg COPC/kg plant tissue DW) |
| $Pd_{(Hg)}$ | = | concentration of total mercury in aboveground produce due to direct (wet and dry) deposition (mg COPC/kg plant tissue DW). $Pd_{(Hg)}$ is calculated in Table B-2-7 in the HHRAP and shown above for produce, forage, and silage. |
| 0.78 | = | multiplier for modeling of divalent mercury (unitless), as shown in Table B-2-7 in the HHRAP |
| 0.22 | = | multiplier for modeling of methyl mercury (unitless), as shown in Table B-2-7 in the HHRAP |

Note that in the equations to calculate the concentration in aboveground plants due to direct deposition, several parameters are stack-specific. This necessitates estimating the concentration in aboveground plants due to direct deposition for each stack individually. The individual concentrations from the three facility stacks will then be summed to obtain the overall concentration in aboveground plants due to direct deposition.

Also, note that in the equations to calculate the concentration in aboveground plants due to direct deposition, several parameters are plant-type-specific (produce, forage, and silage, for example). That is, when estimating the concentration in aboveground plants due to direct deposition for produce, the produce-specific parameters will be used. Likewise, when estimating the concentration in aboveground plants due to direct deposition for forage and silage, the forage-specific parameters and the silage-specific parameters will be used, respectively.

### 6.6.2 Aboveground Plants/Air-to-Plant Transfer

The equations used to estimate the aboveground plant concentration due to air-to-plant transfer are presented below. Per the HHRAP (EPA 2005), special consideration is given to modeling for total mercury, divalent mercury, and methyl mercury. Special consideration is also given to modeling for carbon-14 and tritium [see detailed discussion below, based on NRC guidance (NRC 1977)]. The aboveground plant concentrations due to air-to-plant transfer are estimated for the following plant types: produce, forage, and silage.

The following equation calculates the aboveground plant concentration due to air-to-plant transfer for all vapor-phase COPCs and ROPCs, except total mercury, divalent mercury, methyl mercury, carbon-14 and tritium:

$$Pv = \frac{Q \cdot F_v \cdot Cyv \cdot Bv_{ag} \cdot VG_{ag}}{\rho_a} \qquad \text{(Eq. 5-18 in HHRAP)}$$

The equation to calculate the aboveground plant concentration due to air-to-plant transfer for total mercury is:

$$Pv_{(Hg)} = \frac{0.48 \cdot Q \cdot F_v \cdot Cyv \cdot Bv_{ag} \cdot VG_{ag}}{\rho_a} \qquad \text{(Table B-2-8 in HHRAP)}$$

Where:

$Pv$ = concentration of COPC or ROPC in aboveground plant due to air-to-plant transfer (mg COPC/kg plant tissue DW and pCi ROPC/g plant tissue DW). Note that a unit conversion factor of $1 \times 10^9$ (pCi/mCi) is used for ROPCs.

$Pv_{(Hg)}$ = concentration of total mercury in aboveground plant due to air-to-plant transfer (mg COPC/kg plant tissue DW).

$Q$ = COPC- or ROPC-specific emission rate (g/s for COPCs and Ci/s for ROPCs), derived as described in Section 5.

$F_v$ = fraction of COPC or ROPC air concentration in vapor phase (unitless). $F_v$ is shown in Supplement 4. Constituents with a vapor fraction less than 0.05 are modeled as entirely particulate with an $F_v$ value of 0 (CCN 097844). When $F_v$ is not available, it is empirically derived for most constituents (except metals and some mercury compounds) using Eqs. A-2-1 and A-2-2 (when appropriate) in the HHRAP.

$Cyv$ = unitized yearly average air concentration from vapor phase (μg-s/g-m$^3$ for COPCs and mCi-s/Ci-m$^3$ for ROPCs). $Cyv$, from the air dispersion modeling, is stack-specific.

$Bv_{ag}$ = COPC or ROPC air-to-plant biotransfer factor for aboveground plant (unitless). $Bv_{ag}$ is shown in Supplement 4. The $Bv_{ag}$ value for produce is used to estimate aboveground plant concentration due to air-to-plant transfer for produce, while the $Bv_{ag}$ value for forage (denoted as $Bv_{forage}$ in Supplement 4) is used to estimate aboveground plant concentration due to air-to-plant transfer for both forage and silage (HHRAP Section A2-2.12.4). When $Bv_{ag}$ values are not available, but sufficient information exists, guidance in HHRAP Section A2-2.12.4 was used where applicable for estimating $Bv_{ag}$. Note that because no values for $Bv_{ag}$ could be found for radionuclides that are in vapor phase, $Pv$ for air-to-plant transfer cannot be quantified for a few ROPCs.

$VG_{ag}$ = empirical correction factor for the aboveground plant due to air-to-plant transfer (unitless). For produce, the recommended default values (HHRAP Section 5.3.2.1) for $VG_{ag}$ are used: a value of 0.01 for COPCs and ROPCs with a $\log_{10}$ of the octanol/water partitioning coefficient ($K_{ow}$) ≥ 4 and a $VG_{ag}$ value of 1 for COPCs and ROPCs with a log $K_{ow}$ < 4. $K_{ow}$ is COPC-specific. If no $K_{ow}$ value exists for a constituent, the model conservatively uses $VG_{ag}$ = 1. For forage and silage, the recommended default values of 1 and 0.5, respectively (HHRAP Section 5.4.2.1), are used for $VG_{ag}$. See Table 6-4.

$\rho_a$ = density of air (g/m$^3$). The recommended default value of 1200 g/m$^3$ (EPA 2005) is used.

0.48 = multiplier for modeling of total mercury (unitless), as shown in EPA 2005.

The following equation calculates the aboveground plant concentration due to air-to-plant transfer for divalent mercury:

$$Pv_{(Hg^{2+})} = 0.78 \cdot Pv_{(Hg)} \qquad \text{(Table B-2-8 in HHRAP)}$$

The equation to calculate the aboveground plant concentration due to air-to-plant transfer for methyl mercury is:

$$Pv_{(MHg)} = 0.22 \cdot Pv_{(Hg)} \qquad \text{(Table B-2-8 in HHRAP)}$$

where:

$Pv_{(Hg^{2+})}$ = concentration of divalent mercury in aboveground plant due to air-to-plant transfer (mg COPC/kg plant tissue DW).

$Pv_{(MHg)}$ = concentration of methyl mercury in aboveground plant due to air-to-plant transfer (mg COPC/kg plant tissue DW).

$Pv_{(Hg)}$ = concentration of total mercury in aboveground plant due to air-to-plant transfer (mg COPC/kg plant tissue DW). $Pv_{(Hg)}$ is calculated in Table B-2-8 in the HHRAP.

0.78 = multiplier for modeling of divalent mercury (unitless), in Table B-2-8 of the HHRAP.

0.22 = multiplier for modeling of methyl mercury (unitless), as shown in Table B-2-8 of the HHRAP.

Note that in the equations to calculate the concentration in aboveground plants due to air-to-plant transfer, several parameters are stack-specific. This necessitates estimating the concentration in aboveground plants due to air-to-plant transfer for each facility stack individually. The individual concentrations from the three facility stacks then will be summed to obtain the overall concentration in aboveground plants due to air-to-plant transfer.

Also note that in the equations to calculate the concentration in aboveground plants due to air-to-plant transfer, several parameters are plant-type-specific (i.e., when estimating the concentration in aboveground plants due to air-to-plant transfer for produce, the produce-specific parameters are used). Likewise, when estimating the concentration in aboveground plants due to air-to-plant transfer for forage and silage, the forage-specific parameters and the silage-specific parameters are used, respectively.

As mentioned above, special consideration is given to modeling for carbon-14 and tritium. Risk calculations for most ROPCs are based on the assumption that radionuclides are present as particulates or vapors. However, special consideration must be given to carbon-14 and tritium, as these ROPCs are processed by vegetation with natural carbon and hydrogen, respectively. Thus, the vegetation ingestion pathway for carbon-14 and tritium is dependent on the exchange of carbon and hydrogen between plants and the environment. For this assessment, guidance from Regulatory Guide 1.109 (NRC 1977) is used to account for the bioaccumulation of carbon-14 and tritium in plants that could lead to human exposure

through vegetation ingestion. This is achieved through the use of correction factors and by using the assumptions that all carbon-14 is released by the WTP in oxide form ($^{14}CO$ or $^{14}CO_2$) and tritium is released in water vapor. These correction factors will be applied to the concentration (e.g., pCi/L) estimated at the point of exposure by the air model.

The concentration of carbon-14 in vegetation is calculated assuming that its ratio to the natural carbon in vegetation is equal to the ratio of carbon-14 to natural carbon in the atmosphere surrounding the vegetation (NRC 1977):

$$C_{V(C-14)} = \frac{C_{A(C-14)} \cdot \mathrm{p} \cdot 0.11}{0.16} \qquad \text{(NRC 1977)}$$

where:

| | | |
|---|---|---|
| $C_{V(\text{C-14})}$ | = | concentration of carbon-14 in vegetation (pCi ROPC/g plant tissue DW). |
| $C_{A(\text{C-14})}$ | = | concentration of carbon-14 in the surrounding air (pCi/m$^3$). $C_{A(\text{C-14})}$ is obtained from the air dispersion modeling. |
| p | = | ratio of the total annual release time to the total annual time during which photosynthesis occurs; a conservative ratio of 1.0 is used. |
| 0.11 | = | fraction of the total plant mass that is natural carbon (dimensionless). |
| 0.16 | = | concentration of natural carbon in the atmosphere (g/m$^3$). |

The concentration of tritium in vegetation will be calculated based on the equilibrium between moisture in the air and water in plants (NRC 1977):

$$C_{V(H-3)} = C_{A(H-3)} \cdot 0.75 \cdot (0.5 \div Humidity) \qquad \text{(NRC 1977)}$$

where:

| | | |
|---|---|---|
| $C_{V(\text{H-3})}$ | = | concentration of tritium in vegetation (pCi ROPC/g plant tissue DW). |
| $C_{A(\text{H-3})}$ | = | concentration of tritium in the surrounding air (pCi/m$^3$). $C_{A(H-3)}$ is obtained from the air dispersion modeling. |
| 0.75 | = | fraction of the total plant mass that is water (dimensionless). |
| 0.5 | = | ratio of tritium concentration in plant water to tritium concentration in atmospheric water (dimensionless). |
| *Humidity* | = | absolute humidity of the atmosphere (g/m$^3$). A site-specific relative humidity value of 55.1 %, (the equivalent of 6.0 g/m$^3$ absolute humidity) is used. The value was determined from the average of relative humidity measurements taken from the Hanford Meteorological Station for years 2002 through 2006 (conversion from relative to absolute humidity was done using the average temperature and atmospheric pressure taken from the Hanford Meteorological Station for years 2002 through 2006 [12.5 ºC and 0.98 atm]). |

The concentration of carbon-14 and tritium in vegetation will be used as the total plant concentration for these ROPCs throughout the risk assessment, instead of estimating concentrations for specific types of

plants (e.g., produce, forage, silage, and grain) and specific parts of the plants (i.e., aboveground and belowground).

### 6.6.3 Root Uptake

The concentration of COPCs and ROPCs in plants due to root uptake from the soil will be calculated for aboveground and belowground plants. These concentrations are calculated for all COPCs and all ROPCs except carbon-14 and tritium (see Section 6.6.2). The concentration in plants due to root uptake from the soil is a function of the soil concentration and a soil-to-plant bioaccumulation uptake factor. Section 6.6.3.1 discusses the modeling of aboveground plants due to root uptake. Section 6.6.3.2 discusses the modeling of belowground plants due to root uptake. A discussion of uptake factors is presented in Section 6.6.3.3.

#### 6.6.3.1 Root Uptake/Aboveground Plants

The concentration in aboveground plants due to root uptake is a function of the soil concentration and the soil-to-plant bioaccumulation uptake factor for aboveground plants. The aboveground plant concentrations due to root uptake will be estimated for the following plant types: produce, forage, silage, and grain. No estimates of aboveground plant concentration due to root uptake will be made for carbon-14 and tritium, because a "vegetation concentration" will be estimated as the total plant concentration for these two isotopes (see Section 6.6.2). Also, the untilled soil depth of 2 cm is considered too shallow to estimate plant concentrations for consumption by human and ecological receptors; thus, only root zone soil concentrations (depth of 15 cm for wild produce, forage, wild grain/seed) and tilled soil concentrations (depth of 20 cm for domestic produce, silage, domestic grain) are used to model aboveground plants due to root uptake.

The following equation calculates the aboveground plant concentration due to root uptake for all COPCs and for all ROPCs, except carbon-14 and tritium:

$$Pr_{ag} = Cs \cdot Br_{ag} \qquad \text{(Eq. 5-20A in HHRAP)}$$

where:

$Pr_{ag}$ = concentration of COPC or ROPC in aboveground plant due to root uptake (mg COPC/kg plant tissue DW and pCi ROPC/g plant tissue DW). $Pr_{ag}$ is calculated separately for domestic vegetation (tilled soil – 20 cm depth) and wild vegetation (root-zone soil – 15 cm depth). See the discussion below for appropriate combinations of plant types (i.e., produce, forage, silage, and grain) and soil depths.

$Cs$ = soil concentration (mg/kg for COPCs and pCi/g for ROPCs). $Cs$ is depth-specific and calculated in accordance with Section 6.2 (HHRAP equations 5-1B though 5-1E, as modified for exposure timing and duration).

$Br_{ag}$ = plant-soil bioconcentration factor for aboveground produce (unitless). $Br_{ag}$ is shown in Supplement 4. Separate $Br_{ag}$ values are used for produce (denoted as $Br_{ag}$), forage and silage (per EPA 2005, $Br_{forage}$ is used to denote and estimate both forage and silage), and grain (denoted as $Br_{grain}$). The values for $Br_{ag}$ in Supplement 4 (organic COPCs), will be compared against the calculated mass-limited uptake factors that are described in Section 6.6.3.3 (values shown in Appendix A), and the smaller of the two values will be used in the calculation of the aboveground plant concentration due to root uptake

($Pr_{ag}$). The use of the smaller value in this comparison prevents the overestimation of $Pr_{ag}$, because, in some cases, the derived uptake factors ($Br_{ag}$) are not physically possible, leading to the prediction of more chemical being accumulated by an organism from the soil than is released from the facility and deposited onto the soil. In this situation, use of the mass-limited uptake factor prevents the overestimation of $Pr_{ag}$.

Note that in the equations above, four different plant types (produce, forage, silage, and grain) are modeled. When estimating the concentration in aboveground plants due to root uptake for produce, the produce-specific parameters are used. Likewise, when estimating the concentration in aboveground plants due to root uptake for forage, silage, and grain, the forage-specific parameters, silage-specific parameters, and grain-specific parameters will be used, respectively.

Note also that in the equations above, two different soil depths (tilled soil and root-zone soil) are used because untilled soil (2 cm depth) is considered too shallow for plants with root uptake. However, not every combination of the two soil types with the four plant types is appropriate. The following combinations of soil types and plant types will be used in estimating the aboveground plant concentration due to root uptake:

- When estimating concentrations for aboveground plants due to root uptake for domestic produce, the tilled soil concentrations will be used.
- When estimating concentrations for aboveground plants due to root uptake for wild produce, the root-zone soil concentrations will be used.
- When estimating concentrations for aboveground plants due to root uptake for forage, the root-zone soil concentrations will be used.
- When estimating concentrations for aboveground plants due to root uptake for silage, the tilled soil concentrations will be used.
- When estimating concentrations for aboveground plants due to root uptake for grain, both the tilled soil concentrations and root-zone soil concentrations will be used. Grain modeled from tilled soil will be used in subsequent modeling of domesticated animals (e.g., animals on a farm, such as chickens), while grain modeled from root-zone soil will be used in subsequent modeling of wild animals (e.g., game animals such as wild fowl).

#### 6.6.3.2 Root Uptake/Belowground Plants

The concentration in belowground plants due to root uptake is a function of the soil concentration, the soil-to-plant bioaccumulation uptake factor for belowground plants, and a correction factor for belowground produce. The belowground plant concentrations due to root uptake will be estimated for only one plant type: produce. No estimates of belowground plant concentration due to root uptake will be made for carbon-14 and tritium, because a “vegetation concentration” will be estimated as the total plant concentration for these two isotopes (see Section 6.6.2). Also, the untilled soil depth of 2 cm is considered too shallow to estimate plant concentrations for consumption by human and ecological receptors; thus, only root zone soil concentrations (depth of 15 cm for wild produce) and tilled soil concentrations (depth of 20 cm for domestic produce) will be used to model belowground plants due to root uptake.

The following equation calculates the belowground plant concentration due to root uptake for all COPCs and for all ROPCs, except carbon-14 and tritium:

$$Pr_{bg} = Cs \cdot Br_{rootveg} \cdot VG_{rootveg} \qquad \text{(Eq. 5-20B in HHRAP)}$$

where:

$Pr_{bg}$ = concentration of COPC or ROPC in belowground plant due to root uptake (mg COPC/kg plant tissue DW and pCi ROPC/g plant tissue DW). $Pr_{bg}$ is calculated separately for domestic vegetation (tilled soil - 20 cm depth) and wild vegetation (root-zone soil - 15 cm depth).

$Cs$ = soil concentration (mg/kg for COPCs and pCi/g for ROPCs). $Cs$ is depth-specific and calculated in accordance with Section 6.2 (HHRAP equations 5-1B though 5-1E, as modified for exposure timing and duration).

$Br_{rootveg}$ = plant-soil bioconcentration factor for belowground plants (unitless). Note that per the HHRAP equation 5-20B, for organic COPCs, $Br_{rootveg}$ can be calculated as $RCF \div (Kd_s \times CF)$, where $RCF$ is the root concentration factor (mL/g), $Kd_s$ is the soil-water partition coefficient (L/kg), and $CF$ is a units-conversion factor of 1 (kg·mL)/(g·L). Values for $RCF$, $Kd_s$, and $Br_{rootveg}$ are shown in Supplement 4. The values for $Br_{rootveg}$ in Supplement 4 (organic COPCs) will be compared against the calculated mass-limited uptake factors for produce that are described in Section 6.6.3.3 (values in Appendix A), and the smaller of the two values will be used in the calculation of the belowground plant concentration due to root uptake ($Pr_{bg}$). The use of the smaller value in this comparison prevents the overestimation of $Pr_{bg}$, because in some cases, the derived uptake factors ($Br_{rootveg}$) are not physically possible, leading to the prediction of more chemical being accumulated by an organism from the soil than is released from the facility and deposited onto the soil. In this situation, use of the mass-limited uptake factor prevents the overestimation of $Pr_{bg}$.

$VG_{rootveg}$ = empirical correction factor for belowground plants (unitless). For belowground plants, the recommended default values (HHRAP Section 5.3.3) for $VG_{rootveg}$ are used: a value of 0.01 for COPCs and ROPCs with a $\log_{10} K_{ow} \geq 4$ and a $VG_{rootveg}$ value of 1 for COPCs and ROPCs with a $\log_{10} K_{ow} < 4$ (see Table 6-4). If no $K_{ow}$ value exists for a constituent, the model conservatively uses $VG_{rootveg} = 1$. $K_{ow}$ is constituent-specific and shown in Supplement 4.

Note that in the equation above, two different soil depths (tilled soil and root-zone soil) will be used because untilled soil (2 cm depth) is considered too shallow for plants with root uptake. Domestic root vegetables grown in tilled soil (20 cm depth) will be used in subsequent human health risk equations for the resident consuming produce, while wild root vegetables grown in root-zone soil (15 cm depth) will be used in subsequent human health risk equations for American Indian scenarios where wild produce is gathered and consumed (see Section 7.1.3 for a description of the receptors and exposure pathways).

### 6.6.3.3 Mass-Limited Soil-to-Plant Uptake Factors

The concentrations of contaminants in plants due to root uptake, for both aboveground and belowground plants, are a function of the soil concentration and soil-to-plant bioaccumulation uptake factor. Soil

concentrations used in the modeling of aboveground and belowground plants due to root uptake will be from the root-zone depth (15 cm, wild vegetation) and from the tilled soil depth (20 cm, domestic vegetation); the untilled soil depth (2 cm) is considered too shallow for the modeling of aboveground and belowground plants due to root uptake. The uptake factors for organic chemicals recommended in the HHRAP and SLERAP are calculated from regression equations developed for a few chemicals and exposure situations. In some cases these derived uptake factors are not physically possible because they predict that an organism will accumulate more chemical from the soil than is released from the facility and deposited onto the soil. This problem affects a subset of the organic chemicals being evaluated for the WTP.

For example, if 1 mg of methyl alcohol is deposited per square meter of soil at the point of maximum deposition (calculated as [total deposition rate]×[total years of deposition]×[units conversion factor]), then applying the root-to-aboveground produce transfer factor ($Br_{ag}$) recommended by EPA companion database (EPA 2005) would give a resulting accumulation of 8.38 mg of methyl alcohol in the aboveground edible tissues of plants in one growing season in a 1 square meter area (calculated as [concentration in soil]×[EPA uptake factor]×[EPA default value for yield for produce]). This is more than 8 times the amount of chemical available from 40 years of WTP emissions. This overestimate would then be carried through the risk assessment. For example, if the aboveground plant concentration were overestimated by a factor of more than 8, then risk to human and ecological receptors from ingestion of aboveground plant tissue would also be overestimated by a factor of more than 8. This uptake factor problem does not apply to all COPCs but is limited to some organic chemicals. Uptake factors for organic chemicals are calculated using regression equations; uptake factors for inorganic chemicals and radionuclides are taken from more empirical sources, are sufficiently known, and are not included in this discussion.

There are a variety of ways that this problem may be corrected, depending on the source of the original uptake factor and the amount of uptake information available. Possible solutions include:

- Identify published, empirically-derived uptake factors for the organic chemicals, including development of more representative equations for estimating uptake factors for organic chemicals.
- Conduct laboratory experiments to measure realistic, site-specific, uptake factors.
- Calculate “mass-limited” uptake factors that assume all of the chemical deposited onto the soil is taken up by an organism.

For this risk assessment, the calculation of “mass-limited” uptake factors has been determined to be the most reasonable option and will been performed. Maximum (mass-limited) uptake factors based on simple conservation of mass (that is, that result in transfer of 100 % of the deposited chemical into the receiving organism, but no more) can be calculated. These calculations can be shown to be a function of the soil density and the plant yield. Since the soil density is dependent on the soil depth, and since the root-zone and untilled soil depths apply to the plant concentration due to root uptake, separate determinations of the soil-to-plant, mass-limited uptake factor must be made for these two depths.

The initial soil-to-plant, mass-limited uptake factor (i.e., before adjustments are made for the length of operation for the facility and to divide aboveground and belowground produce) is calculated as:

$$\text{Initial Uptake Factor} = \text{Soil Density} \div \text{Plant Yield}$$

where:

| | | |
|---|---|---|
| Initial Uptake Factor | = | initial calculation of soil-to-plant uptake factor (kg soil/$m^2$ per kg DW plant/$m^2$). |
| Soil Density | = | soil density (kg soil/$m^2$), calculated as bulk density (in kg soil/$m^3$) times soil depth (in meters) (that is, mass per area for a specific depth). For example, using a soil bulk density of 1.3 g/$cm^3$ (1300 kg/$m^3$) and a soil depth of 15 cm (0.15 m), the soil density is (1300 kg/$m^3$) (0.15 m) = 195 kg/$m^2$. The soil density for tilled soil (that is, at the 20 cm depth) is (1300 kg/$m^3$) (0.2 m) = 260 kg/$m^2$. Both soil depths are used to estimate separate mass-limited uptake factors. |
| Plant Yield | = | yield for the plant (kg DW plant/$m^2$). Plant yields used are 2.24 kg/$m^2$ for aboveground produce (EPA 2005), 0.25 kg/$m^2$ for belowground produce (USDA 2009; Baes et. al. 1984), 0.15 kg/$m^2$ for forage (Wisiol 1984), 0.8 kg/$m^2$ for silage (EPA 2005), and 0.25 kg/$m^2$ for grain (Baes et al. 1984, Figure 4.14); see Table 6-4. |

As seen above, the initial soil-to-plant, mass-limited uptake factor is a function of the soil density (which is dependent on the depth of soil) and the plant yield. These mass-limited uptake factors assume that:

- In one growing season, the plant takes up all of the chemical deposited over 40 years.
- The plant concentrates all of the deposited chemical into the aboveground edible portion of the plant.

These assumptions directly contradict other assumptions recommended in the risk assessment guidance (EPA 2005):

- If the plant takes up the entire deposited chemical in one growing season, a human receptor cannot be exposed to this concentration for the recommended exposure durations (which are longer than one year for the various adult receptors).
- If plants take up all of the deposited chemical in the aboveground portion, the concentration in the belowground portion (i.e., root vegetables) must be zero.

To prevent this contradiction, reasonable maximum uptake factors can be calculated using the following assumptions:

- The plants take up one year's worth of deposition each growing season so that for each year of exposure, the plants take up all the deposition available that year.
- The plants take up one-half of the available chemical into the edible aboveground portion and one-half into the edible belowground portion.

Using these assumptions, reasonable maximum uptake factors can be calculated as:

$$\text{Mass-limited Uptake Factor} = \text{Initial Uptake Factor} \times \text{Modifying Factor}$$

where:

| | | |
|---|---|---|
| Mass-limited Uptake Factor | = | final mass-limited, soil-to-plant uptake factor (kg soil/$m^2$ per kg DW plant/$m^2$) |
| Initial Uptake Factor | = | initial estimate of uptake factor (Soil Density ÷ Plant Yield) |
| Modifying Factor | = | adjustments necessary for aboveground versus belowground portions of the plant and for operating duration of the facility that is producing emissions |

There are two types of modifying factors used to estimate the mass-limited uptake factor; these modifying factors are dependent on the type of plant (e.g., produce, forage, silage, and grain). One type of modifying factor is applied to plant types that have both aboveground and belowground concentrations. For produce, a modifying factor of 1/2 is applied to aboveground produce due to root uptake, and a modifying factor of 1/2 is applied to belowground produce due to root uptake (so as to equally divide the mass-limited uptake factor between aboveground and belowground produce due to root uptake). However, this modifying factor related to aboveground as compared to belowground is not applied to forage, silage, or grain, since the edible portions of these plant types are all totally aboveground. The second type of modifying factor (a modifying factor of 1/40) is used to apportion the depositions over the 40-year duration of the facility. This second type of modifying factor is applied to produce, silage, and grain because these products will be harvested and the chemicals in them removed from the soil. This 40-year modifying factor is not applied to forage because some of the forage will remain and decay in place, thus returning the chemicals to the soil. Therefore, the modifying factors (combining the two types of modifying factors, as appropriate) are:

- 1/80 for aboveground produce due to root uptake (1/2 × 1/40)
- 1/80 for belowground produce due to root uptake (1/2 × 1/40)
- 1 for forage (no modifying factor applied)
- 1/40 for silage (1/2 modifying factor not applied)
- 1/40 for grain (1/2 modifying factor not applied)

All of the modifying factors will be used for human health exposure pathways. In contrast, ecological receptors are assumed to consume only forage, so a modifying factor is not used for ecological receptors.

Soil-to-plant, mass-limited uptake factors are calculated in Appendix A. The final step in this mass-limited uptake factor approach is to compare the uptake factors as calculated per the HHRAP guidance (EPA 2005) to these calculated mass-limited uptake factors, on a chemical-by-chemical basis for organic COPCs. The smaller of the two values will be used in the estimation of plant concentrations.

## 6.7 Other Media

Modeling for various animal products (such as wild game and fish) is also necessary for this risk assessment. However, since this modeling effort is slightly different for the human health risk assessment (HHRA), as opposed to the ERA, the modeling will be described in Section 7.1.7.5 for human health receptors and in Section 8.2.3.1 for ecological receptors.

## 6.8 Uncertainty in Fate and Transport Modeling

Uncertainties will be presented in the risk assessment for each aspect of the environmental fate and transport modeling. This includes air dispersion modeling, soil accumulation modeling, surface water modeling, sediment modeling, and plant accumulation modeling. The uncertainty assessment will be presented in the form of both text and tables summarizing the primary contributors and potential magnitude of uncertainties.

A variety of conservative assumptions are used throughout the modeling process to compensate for uncertainties. Some important sources of uncertainty in each type of modeling are summarized in the following sections.

### 6.8.1 Uncertainty in Air Dispersion Modeling

A number of sources of uncertainty exist in the air dispersion modeling, such as:

- Input values, such as stack emission characteristics
- Emission rates of individual COPCs and ROPCs
- Upset factor multipliers used to bound emissions of COPCs and ROPCs
- Meteorological data
- Accurate simulation of the atmospheric dispersion of emissions plume from each stack

Some of these uncertainties are based on the limited data available, such as estimated emission rates as described in Section 5.5. Other uncertainties become larger when the model is used at the limits of its normal application (for instance, in very complex terrain as distances from the source increase).

### 6.8.2 Uncertainty in Soil Accumulation Modeling

Estimating soil concentrations incorporates numerous uncertainties, which are inherent in the assumptions that are the basis for the calculations. Examples of uncertainty in the parameters would be soil mixing depth, soil bulk density, and volumetric water content, which are assigned a single value, but may vary widely over a relatively small area. The concentration of COPCs and ROPCs in soil will be subject to loss due to biotic and abiotic degradation; however, transformation and subsequent increase of secondary COPCs (that is, degradation products) will not be considered in the assessment. Transformation of ROPCs and formation of daughter products will be included in this assessment through the use of toxicity values (slope factors) that include daughter products.

### 6.8.3 Uncertainty in Surface Water Accumulation Modeling

Uncertainty in the estimation of surface water and fish concentrations of COPCs and ROPCs results from the assumptions used in the calculations. The equations assume that the water body reaches a steady-state condition; however, for application to the Columbia River and any other flowing surface water, this assumption is extremely conservative. Additionally, many of the equations used to model the fate of COPCs and ROPCs deposited into the water body greatly simplify the mechanisms occurring within such a dynamic system and may overestimate or underestimate the concentration of given COPCs and ROPCs in the surface water. It is also assumed that the maximum deposition of COPCs and ROPCs occurs over

the entire depositional area of the water body, which is a source of additional uncertainty and conservatism.

### 6.8.4 Uncertainty in Sediment Accumulation Modeling

There is uncertainty in assigning COPCs exclusively to either water column or bed sediment for purposes of estimating fish-tissue concentrations as described in the HHRAP and concentrations of other organisms as described in the SLERAP. The problem is that this approach to partitioning COPCs in the aquatic environment may not reflect the multiple pathways by which different kinds of organisms are potentially exposed to any given contaminant.

The EPA approach estimates concentrations of organisms using bioconcentration factors (*BCF*s) and dissolved water concentrations for COPCs with low values for $K_{ow}$, bioaccumulation factors (*BAF*s) and whole-water concentrations for COPCs with moderate values for $K_{ow}$, and *BAF*s and sediment concentrations for COPCs with high values for $K_{ow}$. The uncertainty lies in the source and meaning of the different biotransfer factors used for the different media. If the EPA (2005) biotransfer factors do not incorporate all the pathways to all organisms from the single medium where each COPC is assumed to predominate, then the exposure will be underestimated. It is unlikely that the EPA transfer factors account for all pathways relevant to all fish.

Fish take up contaminants into their tissue via the water in contact with their gills and via the ingestion of water, abiotic particulates, and biota. Some organisms will be primarily exposed by one pathway, while others will be exposed over multiple pathways:

- Dissolved contaminants are primarily taken up across the gill membrane; thus, all organisms living in the water column will be exposed to dissolved contaminants.
- Filter-feeding organisms, which usually live in the water column, will also be exposed to contaminants bound to suspended particulates that they filter out of the water and ingest.
- Sediment-ingesting organisms that live in the water column will also be exposed to sediment contaminants by direct ingestion.
- Predatory fish, which are also water-column dwellers, will also be exposed to dissolved, particulate-bound, and sediment contaminants by ingesting prey that were so exposed, as well as by direct uptake from the water column and ingestion of suspended particulates.

In fact, there are probably few organisms that are exposed to only dissolved contaminants, perhaps only those that live in the water column and selectively feed on planktonic animals to the exclusion of abiotic particulates. Therefore, assigning each contaminant to a particular class of media (dissolved water, whole water, and bed sediment) based on its tendency to adsorb to particles or organic carbon, potentially neglects pathways from other media. Further discussion of uncertainty related to these pathways is presented in the ecological section (Section 8.6) of this work plan.

### 6.8.5 Uncertainty in Plant Accumulation Modeling

Calculation of COPC and ROPC concentrations in biota incorporates the uncertainties inherent in calculation of air and soil concentrations because the air and soil are the sources of COPCs and ROPCs to plants. Uncertainties also arise from the assumption that the location of maximum soil concentration is the location of exposure to biota over a multiple-year period. Additionally, although COPCs and ROPCs are incorporated into plants and animal tissue, it is assumed that the COPC and ROPC concentration in

soil does not decrease due to these processes. Assumptions of the animal feed ingestion rates introduce additional uncertainty because they are based on average rates.

Additional pathways, such as fugitive dust emissions or entrainment of soil in rainwater splash, may contribute to COPC and ROPC concentrations in biota. However, no equations are available to quantify these pathways. In addition, the chemical transport through inedible portions of plants (such as corn husks) may contribute to COPC and ROPC concentrations in edible portions of plants (such as corn kernel). Transfer factors for this type of chemical transport are not available.

## 6.9 Summary of Environmental Modeling

Air dispersion modeling will be used to determine COPC- and ROPC-specific concentrations and deposition rates resulting from emissions of the WTP. The assessment area extends to a 50-km radius from the WTP. These results will be used as input into the human health and ecological risk assessments.

The CALPUFF model will be used for the air-quality modeling task. The WTP emissions, as determined by the design engineers, and 5 years of upper air and Hanford Site meteorological data collected by the Hanford Site Meteorological Station network will be used as input into the model. The COPC and ROPC-specific concentrations and deposition rates will be calculated at a gridded network of receptors and at specific sensitive receptors identified by the risk assessment analysts.

Fate and transport modeling will be used to estimate COPC and ROPC concentrations in various exposure media (air, soil, surface water, sediment, plants, and animal tissue). This modeling effort will utilize assumed emissions rates with a combination of site-specific and default parameters to describe the movement of COPCs and ROPCs through the environment. This modeling is predictive and cannot be confirmed by sampling environmental media since the emissions source does not yet exist. The uncertainty associated with this predictive modeling is addressed through the use of conservative assumptions whenever possible. Estimated media concentrations resulting from this modeling effort will be used in the exposure assessment for the human health (Section 7) and ecological (Section 8) risk assessments. Environmental modeling will be the same for the PRA and final risk assessment (FRA) with the possible inclusion of additional site-specific modeling parameters in the FRA.

## 6.10 References

### 6.10.1 Project Documents

CCN 097844, *Discuss and Resolve the Outstanding Risk Assessment Issues in the Risk Assessment Work Plan (RAWP)*, Meeting minutes from a meeting held on 9 September 2004 between WTP, US Environmental Protection Agency, Region 10, and Washington Department of Ecology, in Seattle, Washington, USA.

CCN 194345, meeting minutes, D. Blumenkranz (WTP) to C. Bowman (Ecology) and others, *Modeling Particle-bound Constituents in CalPuff*, 25 June 2009.

24590-CM-HC4-HKYM-00001-01-00002, Rev 00A, *Modeling Verification and Methods Report, River Protection Project, Hanford Tank Waste Treatment and Immobilization Plant.*

24590-WTP-HPC-M30T-00002, Rev 00B, *WTP Stack Parameters and Flow*.

24590-WTP-RPT-ENV-08-001, Rev 1, *Hanford Tank Waste Treatment and Immobilization Plant Risk Assessment Air Quality Modeling Protocol.*

24590-WTP-RPT-ENV-13-001, Rev 0, *CALPOST Data Evaluation to Support the Environmental Risk Assessment*.

### 6.10.2 Codes and Standards

40 CFR 51. App. W. "Guideline on Air Quality Models", US Environmental Protection Agency, *Code of Federal Regulations*, as amended.

### 6.10.3 Other Documents

Bonneville Power Administration. 2001. *Modeling Protocol, Regional Air Quality Modeling Study*, Bonneville Power Administration.

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PNNL. 2006. *Hanford Site Environmental Report for Calendar Year 2005*, PNNL-15892, September 2006. Pacific Northwest National Laboratory, Richland, Washington.

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Wischmeier WH and Smith DD. 1978. *Predicting Rainfall Erosion Losses - A Guide to Conservation Planning*, AH-537, US Department of Agriculture, Science and Education Administration, Washington, DC.

Wisiol K. 1984. "Estimating Grazingland Yield from Commonly Available Data," in *J. Range Mgmt*., Volume 37, Issue 5, p 471–475, September 1984.

**Table 6-1 CALMET Model Input Assumptions**

| Level | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 |
|---|---|---|---|---|---|---|---|---|---|
| Cell face heights (m) | 20 | 40 | 80 | 160 | 320 | 670 | 1400 | 2600 | 4000 |
| BIAS | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| Other CALMET parameters | IEXTRP | R1 | R2 | RMAX1 | RMAX2 |
|---|---|---|---|---|---|
| | 4 | 10 | 12 | 12 | 12 |

| Scavenging Coefficient For Wet Deposition | | | |
|---|---|---|---|
| **Species** | **Species Abbreviation** | **Liquid Precipitation** | **Frozen Precipitation** |
| Pretreatment Facility Vapor Concentration | PTFV | 0.00017 | 0.00006 |
| Pretreatment Facility Particulate Concentration (1 micron) | PTFP1 | 0.00005 | 0.000017 |
| Pretreatment Facility Particulate Concentration (2.5 micron) | PTFP25 | 0.00005 | 0.000017 |
| LAW Facility Vapor Concentration | LAWV | 0.00017 | 0.00006 |
| LAW Facility Particulate Concentration (1 micron) | LAWP1 | 0.00005 | 0.000017 |
| LAW Facility Particulate Concentration (2.5 micron) | LAWP25 | 0.00005 | 0.000017 |
| HLW Facility Vapor Concentration | HLWV | 0.00017 | 0.00006 |
| HLW Facility Particulate Concentration (1 micron) | HLWP1 | 0.00005 | 0.000017 |
| HLW Facility Particulate Concentration (2.5 micron) | HLWP25 | 0.00005 | 0.000017 |

Reference: 24590-WTP-RPT-ENV-08-001, Rev 1, *Hanford Tank Waste Treatment and Immobilization Plant Risk Assessment Air Quality Modeling Protocol*, Supplement 5

**Table 6-2 Site-Specific Parameters Used in Modeling Soil Concentrations**

| Parameter | Value | Units | Source or Reference |
|---|---|---|---|
| Total watershed area receiving deposition ($A_L$) | $3.927\times10^9$ | $m^2$ | Value estimated as half of the study area |
| Soil bulk density (*BD*) | 1.3 | $g/cm^3$ | Halvorson JJ, McCool DK, King LG, and Gatto LW. 1998. *Ground Freezing Effects on Soil Erosion of Army Training Lands. Part 2. Overwinter Changes to Tracked-Vehicle Ruts*, Yakima Training Center, Washington, Special Report 98-8. Cold Regions Research and Engineering Laboratory, US Army Corps of Engineers. |
| USLE cover management factor (*C*) | 0.1 | unitless | Table B-4-13 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm (dense vegetative cover assumed). |
| Soil enrichment ratio (*ER*) | Inorganics: 1<br>Organics: 3 | unitless | Table B-1-3 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Average annual evapotranspiration ($E_v$) | 16.8 | cm/yr | Wisiol K. 1984. "Estimating Grazingland Yield from Commonly Available Data," in J. Range Mgmt., Volume 37, Issue 5, p 471-475, September 1984. |
| Average annual irrigation (*I*) | 0 | cm/yr | assumed value |
| USLE erodibility factor (*K*) | 0.39 | ton/acre | Table B-4-13 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| USLE length-slope factor (*LS*) | 1.5 | unitless | Table B-4-13 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |

**Table 6-2 Site-Specific Parameters Used in Modeling Soil Concentrations**

| Parameter | Value | Units | Source or Reference |
|---|---|---|---|
| Average annual precipitation (*P*) | 18.47 | cm/yr | Average annual precipitation computed from Hanford Environmental Reports, years 2002 through 2006;<br>PNNL. 2003. *Hanford Site Environmental Report for Calendar Year 2002*. PNNL-14295, Pacific Northwest National Laboratory, Richland, Washington, September 2003.<br>PNNL. 2004. *Hanford Site Environmental Report for Calendar Year 2003*. PNNL-14687, Pacific Northwest National Laboratory, Richland, Washington, September 2004.<br>PNNL. 2005. *Hanford Site Environmental Report for Calendar Year 2004*. PNNL-15222, Pacific Northwest National Laboratory, Richland, Washington, September 2005.<br>PNNL. 2006. *Hanford Site Environmental Report for Calendar Year 2005*. PNNL-15982, Pacific Northwest National Laboratory, Richland, Washington, September 2006.<br>PNNL. 2007. *Hanford Site Environmental Report for Calendar Year 2006*. PNNL-16623, Pacific Northwest National Laboratory, Richland, Washington, September 2007. |
| USLE supporting practice factor (*PF*) | 1 | unitless | Table B-4-13 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Ideal Gas Law Constant (*R*) | 0.08205746 | L·atm/ mol·°K | Wikimedia Foundation, Inc., St. Petersburg, Florida (http://en.wikipedia.org/wiki/Universal_gas_constant, accessed July 2006) |
| USLE rainfall (or erosivity) factor (*RF*) | 20 | $yr^{-1}$ | Site-specific value from Figure 1 of Wischmeier and Smith 1978. |
| Average annual surface runoff from pervious areas (*RO*) | 2.5 | cm/yr | Estimated: assumes the majority of rainfall recharges or evaporates |

**Table 6-2 Site-Specific Parameters Used in Modeling Soil Concentrations**

| Parameter | Value | Units | Source or Reference |
|---|---|---|---|
| Ambient temperature ($T_a$) | 286 | °K | Average ambient air temperature computed from Hanford Environmental Reports, years 2002 through 2006;<br>PNNL. 2003. *Hanford Site Environmental Report for Calendar Year 2002*. PNNL-14295, Pacific Northwest National Laboratory, Richland, Washington, September 2003.<br>PNNL. 2004. *Hanford Site Environmental Report for Calendar Year 2003*. PNNL-14687, Pacific Northwest National Laboratory, Richland, Washington, September 2004.<br>PNNL. 2005. *Hanford Site Environmental Report for Calendar Year 2004*. PNNL-15222, Pacific Northwest National Laboratory, Richland, Washington, September 2005.<br>PNNL. 2006. *Hanford Site Environmental Report for Calendar Year 2005*. PNNL-15982, Pacific Northwest National Laboratory, Richland, Washington, September 2006.<br>PNNL. 2007. *Hanford Site Environmental Report for Calendar Year 2006*. PNNL-16623, Pacific Northwest National Laboratory, Richland, Washington, September 2007. |
| Water body temperature ($T_{wk}$) | 285 | °K | Average median Columbia River temperature (Vernita Bridge measurement) computed from Hanford Environmental Reports, years 2002 through 2006;<br>PNNL. 2003. *Hanford Site Environmental Report for Calendar Year 2002*. PNNL-14295, Pacific Northwest National Laboratory, Richland, Washington, September 2003.<br>PNNL. 2004. *Hanford Site Environmental Report for Calendar Year 2003*. PNNL-14687, Pacific Northwest National Laboratory, Richland, Washington, September 2004.<br>PNNL. 2005. *Hanford Site Environmental Report for Calendar Year 2004*. PNNL-15222, Pacific Northwest National Laboratory, Richland, Washington, September 2005.<br>PNNL. 2006. *Hanford Site Environmental Report for Calendar Year 2005*. PNNL-15982, Pacific Northwest National Laboratory, Richland, Washington, September 2006.<br>PNNL. 2007. *Hanford Site Environmental Report for Calendar Year 2006*. PNNL-16623, Pacific Northwest National Laboratory, Richland, Washington, September 2007. |
| Empirical intercept coefficient ($a$) | 0.6 | unitless | Table B-4-14 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |

**Table 6-2 Site-Specific Parameters Used in Modeling Soil Concentrations**

| Parameter | Value | Units | Source or Reference |
|---|---|---|---|
| Empirical slope coefficient ($b$) | 0.125 | unitless | Table B-4-14 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Fraction of organic carbon in soil ($f_{oc,s}$) | 0.0044 | unitless | Average TOC. CCN 150854, E-mail from Jerry Yokel, Washington Department of Ecology, to David Blumenkranz, WTP, *Ecology Sample Results (soil TOC and pH)*, January 02, 2007, Richland, Washington. |
| Soil volumetric water content ($\theta_{sw}$) | 0.2 | $mL/cm^3$ | Table B-1-3 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Solids particle density ($\rho_{soil}$) | 2.7 | $g/cm^3$ | Table B-1-6 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |

**Table 6-3 Site-Specific Parameters Used in Modeling Surface Water and Sediment**

| Parameter | Value | Units | Source or Reference |
|---|---|---|---|
| Area of impervious watershed receiving deposition ($A_I$) | 0 | $m^2$ | Estimated: assumes deposited COPCs are all mixed with soil |
| Total watershed area receiving deposition ($A_L$) | $3.927\times10^9$ | $m^2$ | Value estimated as half of the study area |
| Water body surface area ($A_w$) | $3.652\times10^7$ | $m^2$ | PNNL. 2005. *Hanford Site National Environmental Policy Act (NEPA) Characterization*, PNL-6415, Rev 17, September 2005, Pacific Northwest National Laboratory, Richland, Washington. (Sect. 4, pg 41, footnote a) |
| Soil bulk density (*BD*) | 1.3 | $g/cm^3$ | Halvorson JJ, McCool DK, King LG, and Gatto LW. 1998. *Ground Freezing Effects on Soil Erosion of Army Training Lands. Part 2. Overwinter Changes to Tracked-Vehicle Ruts*, Yakima Training Center, Washington, Special Report 98-8. Cold Regions Research and Engineering Laboratory, US Army Corps of Engineers. |
| USLE cover management factor (*C*) | 0.1 | unitless | Table B-4-13 of EPA, 2005, Human Health Risk Assessment *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. (dense vegetative cover assumed) |
| Bed sediment concentration ($C_{BS}$) | 1 | $g/cm^3$ | Table B-4-22 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| USLE erodibility factor (*K*) | 0.39 | ton/acre | Table B-4-13 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Gas-Phase Transfer Coeff. ($K_G$) | 36500 | m/yr | Calculated value per Table B-4-21 in EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| USLE length-slope factor (*LS*) | 1.5 | unitless | Table B-4-13 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |

**Table 6-3 Site-Specific Parameters Used in Modeling Surface Water and Sediment**

| Parameter | Value | Units | Source or Reference |
|---|---|---|---|
| USLE supporting practice factor (*PF*) | 1 | unitless | Table B-4-13 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Ideal Gas Law Constant (*R*) | 0.08205746 | L-atm/ mol-°K | Wikimedia Foundation, Inc., St. Petersburg, Florida (http://en.wikipedia.org/wiki/Universal_gas_constant, accessed July 2006) |
| USLE rainfall (or erosivity) factor (*RF*) | 20 | 1/yr | Site-specific value from Figure 1 of Wischmeier and Smith 1978. |
| Average annual surface runoff from pervious areas (*RO*) | 2.5 | cm/yr | Estimated: assumes the majority of rainfall recharges or evaporates |
| Total suspended solids concentration (*TSS*) | 10 | mg/L | Table B-4-16 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Water body temperature ($T_{wk}$) | 285 | °K | Average median Columbia River temperature (Vernita Bridge measurement) computed from Hanford Environmental Reports, years 2002 through 2006;<br>PNNL. 2003. *Hanford Site Environmental Report for Calendar Year 2002*. PNNL-14295, Pacific Northwest National Laboratory, Richland, Washington, September 2003.<br>PNNL. 2004. *Hanford Site Environmental Report for Calendar Year 2003*. PNNL-14687, Pacific Northwest National Laboratory, Richland, Washington, September 2004.<br>PNNL. 2005. *Hanford Site Environmental Report for Calendar Year 2004*. PNNL-15222, Pacific Northwest National Laboratory, Richland, Washington, September 2005.<br>PNNL. 2006. *Hanford Site Environmental Report for Calendar Year 2005*. PNNL-15982, Pacific Northwest National Laboratory, Richland, Washington, September 2006.<br>PNNL. 2007. *Hanford Site Environmental Report for Calendar Year 2006*. PNNL-16623, Pacific Northwest National Laboratory, Richland, Washington, September 2007. |
| Average volumetric flow rate through water body ($V_{fx}$) | $1.060 \times 10^{11}$ | $m^3$/yr | PNNL. 2002. *An Initial Assessment of Hanford Impact Performed with the System Assessment Capability*, PNNL-14027, Pacific Northwest National Laboratory. Richland, Washington (based on 3360 $m^3$/sec for Priest Rapids) |

**Table 6-3 Site-Specific Parameters Used in Modeling Surface Water and Sediment**

| Parameter | Value | Units | Source or Reference |
|---|---|---|---|
| Empirical intercept coefficient ($a$) | 0.6 | unitless | Table B-4-14 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Empirical slope coefficient ($b$) | 0.125 | unitless | Table B-4-14 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Depth of upper benthic sediment layer ($d_{bs}$) | 0.03 | m | Table B-4-15 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Depth of water column ($d_{wc}$) | 8.65632 | m | Columbia Basin Research. 1996. Columbia River Salmon Passage Model CriSP.1.5 Theory, *Calibration & Validation Manual*, Copyright © 1996. Available at http://www.cbr.washington.edu/crisp/models/crisp1manual/theory/theory.mif14.html (Hanford Reach full pool depth at the downstream end of the segment, accessed July 2006.) |
| Temperature correction factor ($\theta$) | 1.026 | unitless | Table B-4-19 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Bed sediment porosity ($\theta_{bs}$) | 0.6 | L(water)/ L(sediment) | Table B-4-16 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Soil volumetric water content ($\theta_{sw}$) | 0.2 | $mL/cm^3$ | Table B-1-3 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Current velocity ($u$) | 1.37 | m/s | Columbia Basin Research. 1996. Columbia River Salmon Passage Model CriSP.1.5 Theory, *Calibration & Validation Manual*, Copyright © 1996. Available at http://www.cqs.washington.edu/crisp/models/crisp1manual/theory/theory.mif14.html. (Accessed in 2002.) Used John Day Free flow rate of 4.5 ft/sec. |

**Table 6-4 Site-Specific Modeling Parameters for Estimating Exposure Point Concentrations in Plants**

| Parameter | Value | Units | Source or Reference |
|---|---|---|---|
| Fraction of COPC wet deposition that adheres to plant surfaces ($F_W$) | 0.2 for anions<br>0.6 for cations and organics | unitless | Table B-3-7 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Interception fraction of the edible portion of plant ($Rp$) | 0.39 | unitless | Section 5.3.1.1 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Interception fraction of the edible portion of forage ($Rp_{forage}$) | 0.5 | unitless | Section 5.3.1.1 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Interception fraction of the edible portion of fruit ($Rp_{fruit}$) | 0.053 | unitless | Section 5.3.1.1 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Interception fraction of the edible portion of silage ($Rp_{silage}$) | 0.46 | unitless | Section 5.3.1.1 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Interception fraction of the edible portion of vegetation ($Rp_{veg}$) | 0.982 | unitless | Section 5.3.1.1 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC, USA. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Length of plant exposure to deposition per harvest ($Tp$) | 0.164 | yr | Calculated value per Eq. 5-16 in EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Length of forage exposure to deposition per harvest ($Tp_{forage}$) | 0.12 | yr | Section 5.4.1.2 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Length of silage exposure to deposition per harvest ($Tp_{silage}$) | 0.16 | yr | Section 5.4.1.2 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |

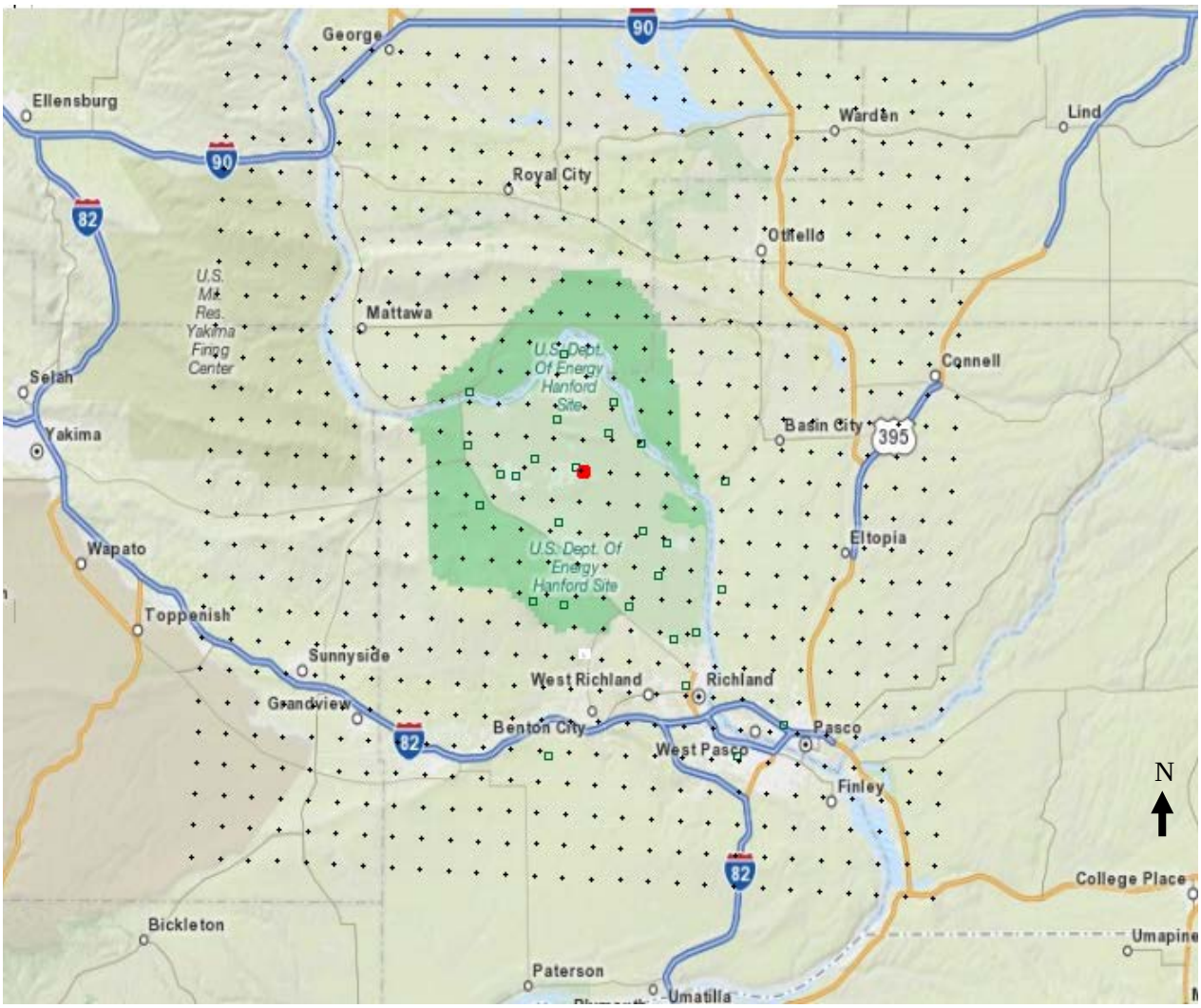
**Table 6-4 Site-Specific Modeling Parameters for Estimating Exposure Point Concentrations in Plants**

| Parameter | Value | Units | Source or Reference |
|---|---|---|---|
| Empirical correction factor for aboveground produce ($VG_{ag}$) | 0.01 for log $K_{ow}$ > 4<br>1.0 for log $K_{ow}$ < 4 | unitless | Section 5.3.2.1 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Empirical correction factor for forage ($VG_{ag_forage}$) | 1 | unitless | Section 5.4.2.1 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Empirical correction factor for silage ($VG_{ag_silage}$) | 0.5 | unitless | Section 5.4.2.1 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Correction factor for belowground produce ($VG_{rootveg}$) | 0.01 for log $K_{ow}$ > 4<br>1.0 for log $K_{ow}$ < 4 | unitless | Section 5.3.3 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Yield or standing crop biomass of the edible portion of plant ($Yp$) | 2.24 | kg DW/$m^2$ | Section 5.3.1.4 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Yield or standing crop biomass of the edible portion of below-ground produce ($Yp_{bg}$) | 0.25 | kg DW/$m^2$ | A yield of 600 cwt (WW)/acre (6.72 kg/$m^2$) was assumed based on USDA 2009 data for potatoes and onions. A conversion factor of 0.173 kg(DW)/kg(WW) (Baes et. al. 1984, Table 2.3, potato and onion average) is applied resulting a dry weight yield of 1.17 kg/$m^2$ |
| Yield or standing crop biomass of the edible portion of grain ($Yp_{grain}$) | 0.25 | kg DW/$m^2$ | Baes et al. 1984, Figure 4.14 |
| Yield or standing crop biomass of the edible portion of forage ($Yp_{forage}$) | 0.15 | kg DW/$m^2$ | Wisiol K. 1984. "Estimating Grazingland Yield from Commonly Available Data," in J. Range Mgmt., Volume 37, Issue 5, p 471-475, September 1984.)<br>$Yp_{forage}$ = (1,500 kg/ha) × (1 ha / 10,000 $m^2$)<br>= 0.15 kg/$m^2$ dry weight. |

**Table 6-4 Site-Specific Modeling Parameters for Estimating Exposure Point Concentrations in Plants**

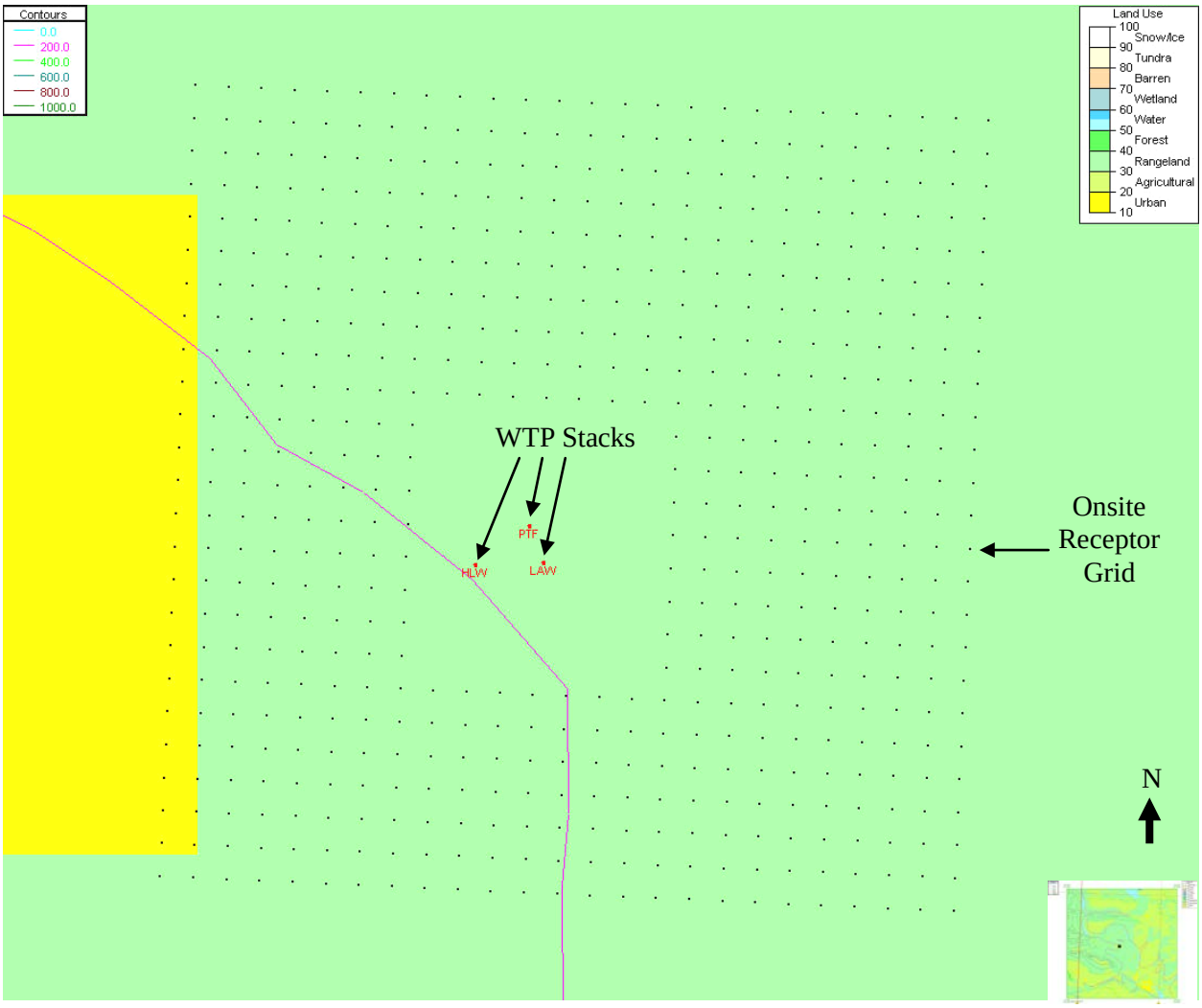
| Parameter | Value | Units | Source or Reference |
|---|---|---|---|
| Yield of the edible portion of fruit ($Yp_{fruit}$) | 0.25 | kg DW/m$^2$ | Section 5.3.1.4 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Yield or standing crop biomass of the edible portion of silage ($Yp_{silage}$) | 0.8 | kg DW/m$^2$ | Section 5.4.1.4 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Yield or standing crop biomass of the edible portion of vegetation ($Yp_{veg}$) | 5.66 | kg DW/m$^2$ | Section 5.3.1.4 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Plant surface loss coefficient ($kp$) | 18 | yr$^{-1}$ | Section 5.3.1.2 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |
| Density of air ($\rho_{air}$) | 0.0012 | g/cm$^3$ | Table B-4-21 of EPA, 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm. |

**Figure 6-1 CALMET Domain**




Symbols:
• indicates receptor grid node
□ indicates met station
• indicates WTP location

**Figure 6-2 Onsite Receptor Locations**




Symbols:
• indicates receptor grid node
• indicates WTP location

**Figure 6-3 Offsite Receptor Locations**

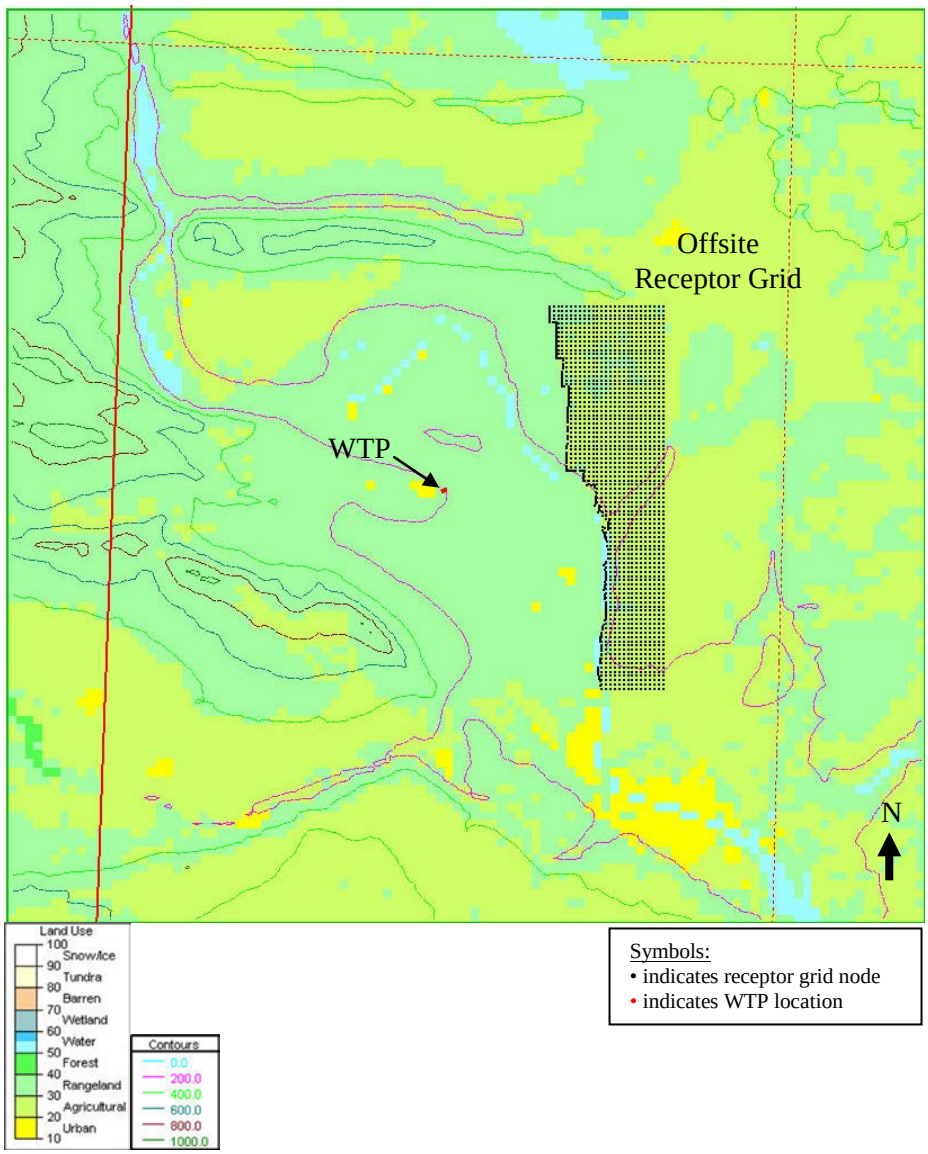


**Figure 6-4 Gable Mountain Receptor Locations**

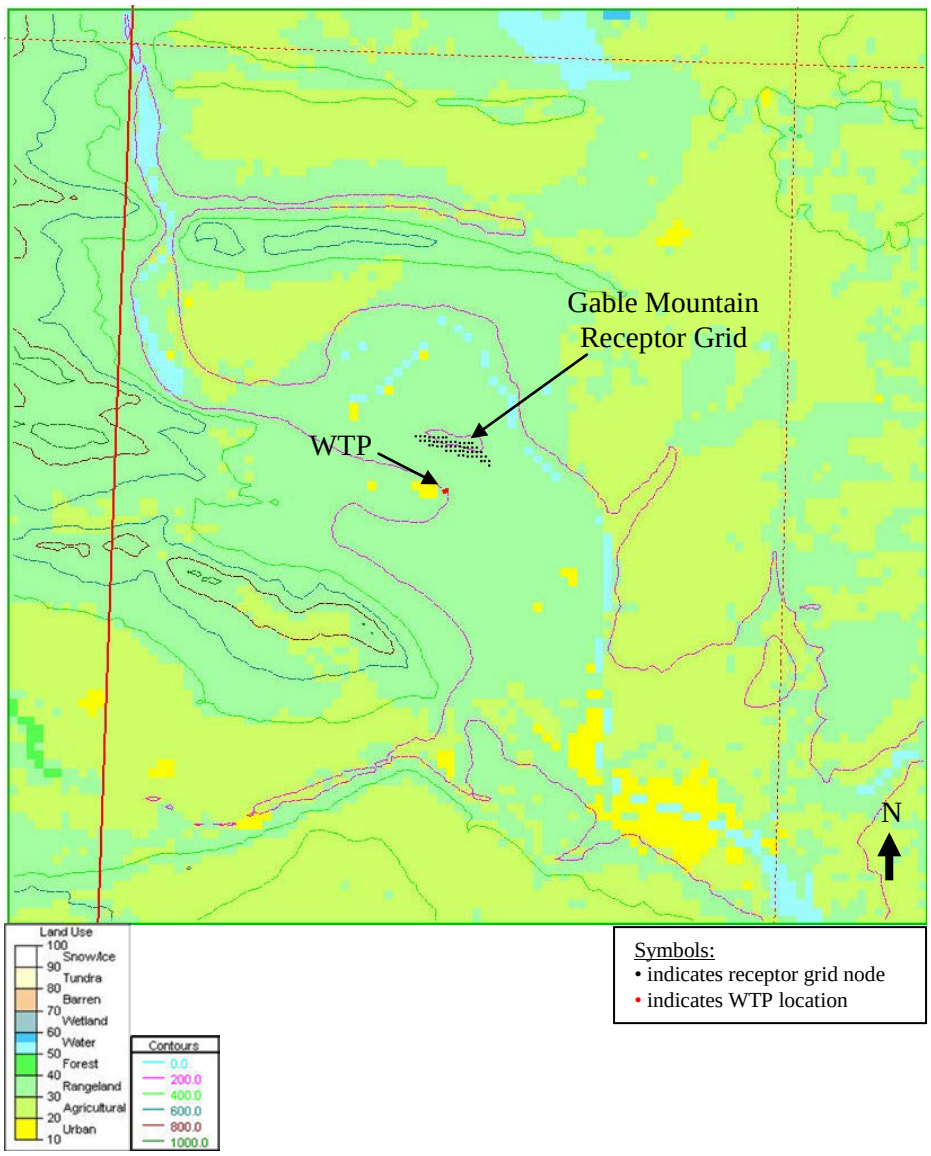


**Figure 6-5 Columbia River Receptor Locations**

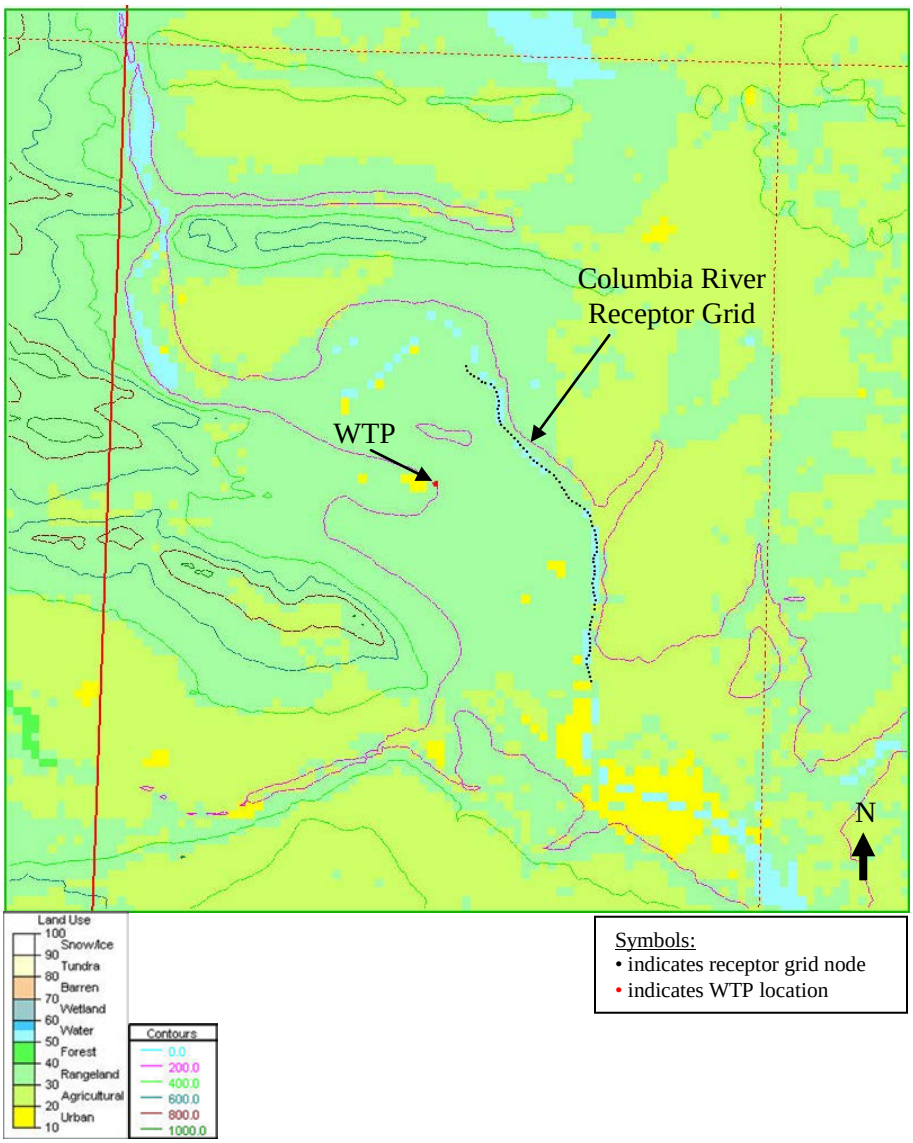


**Figure 6-6 Hunter-Gatherer Current Receptor Locations**

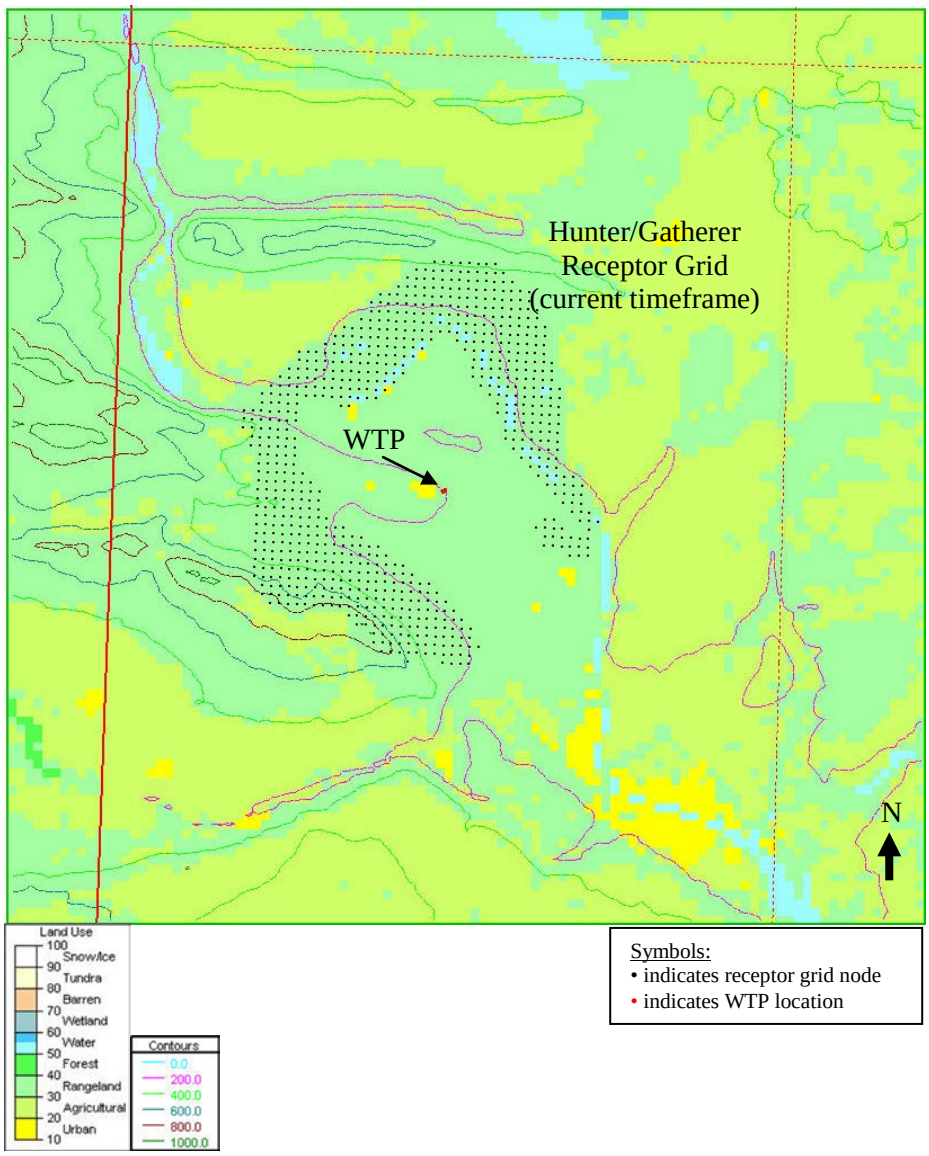


**Figure 6-7 Hunter-Gatherer Future Receptor Locations**

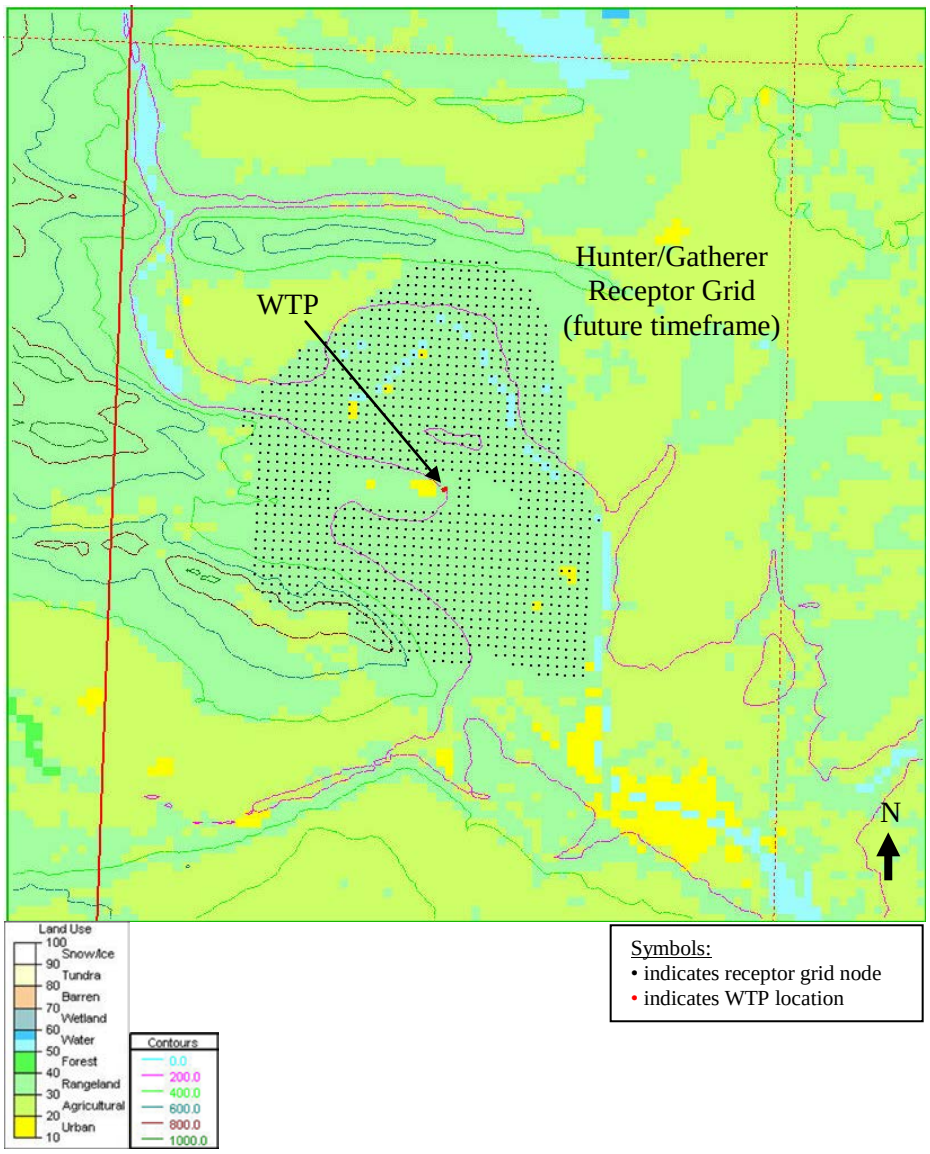
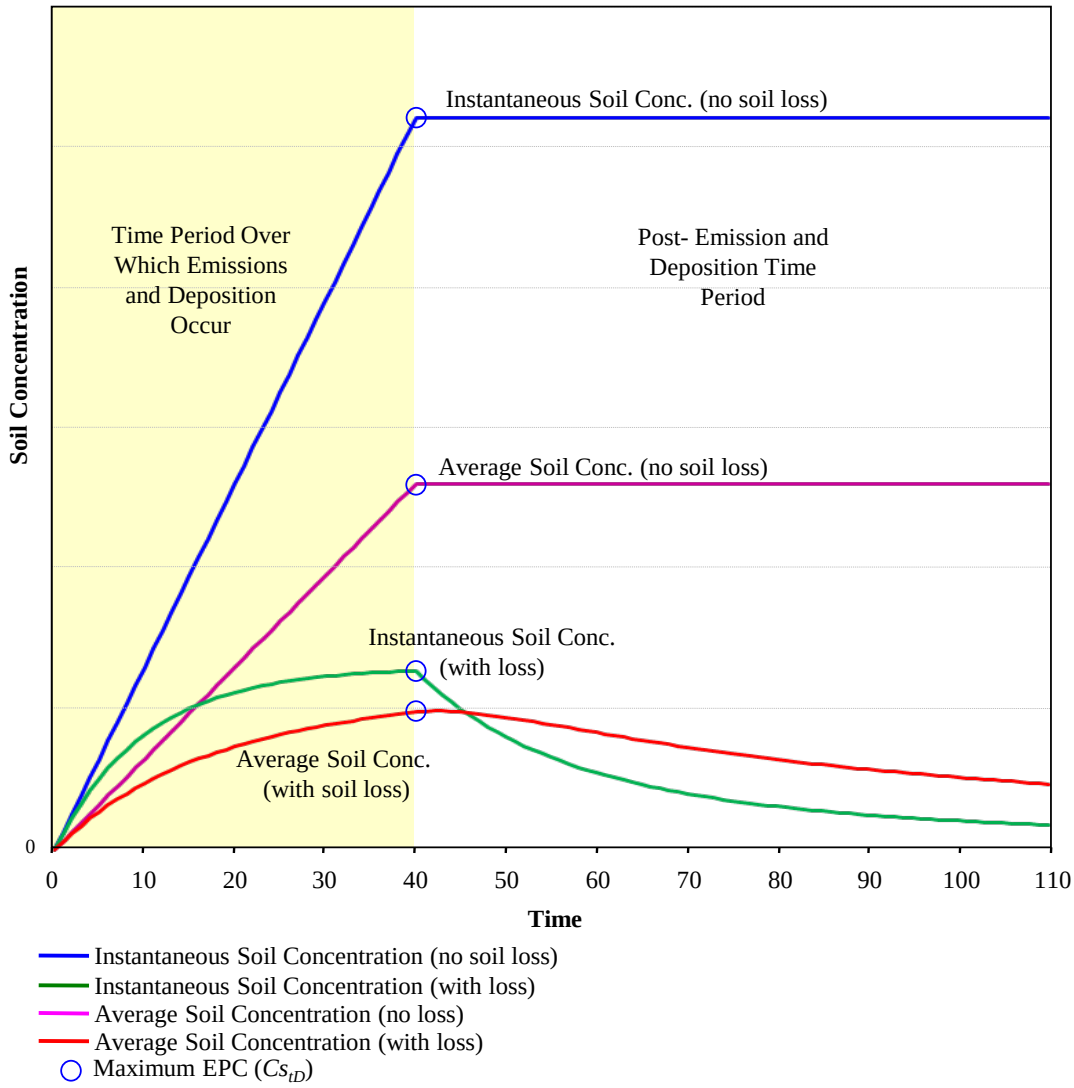
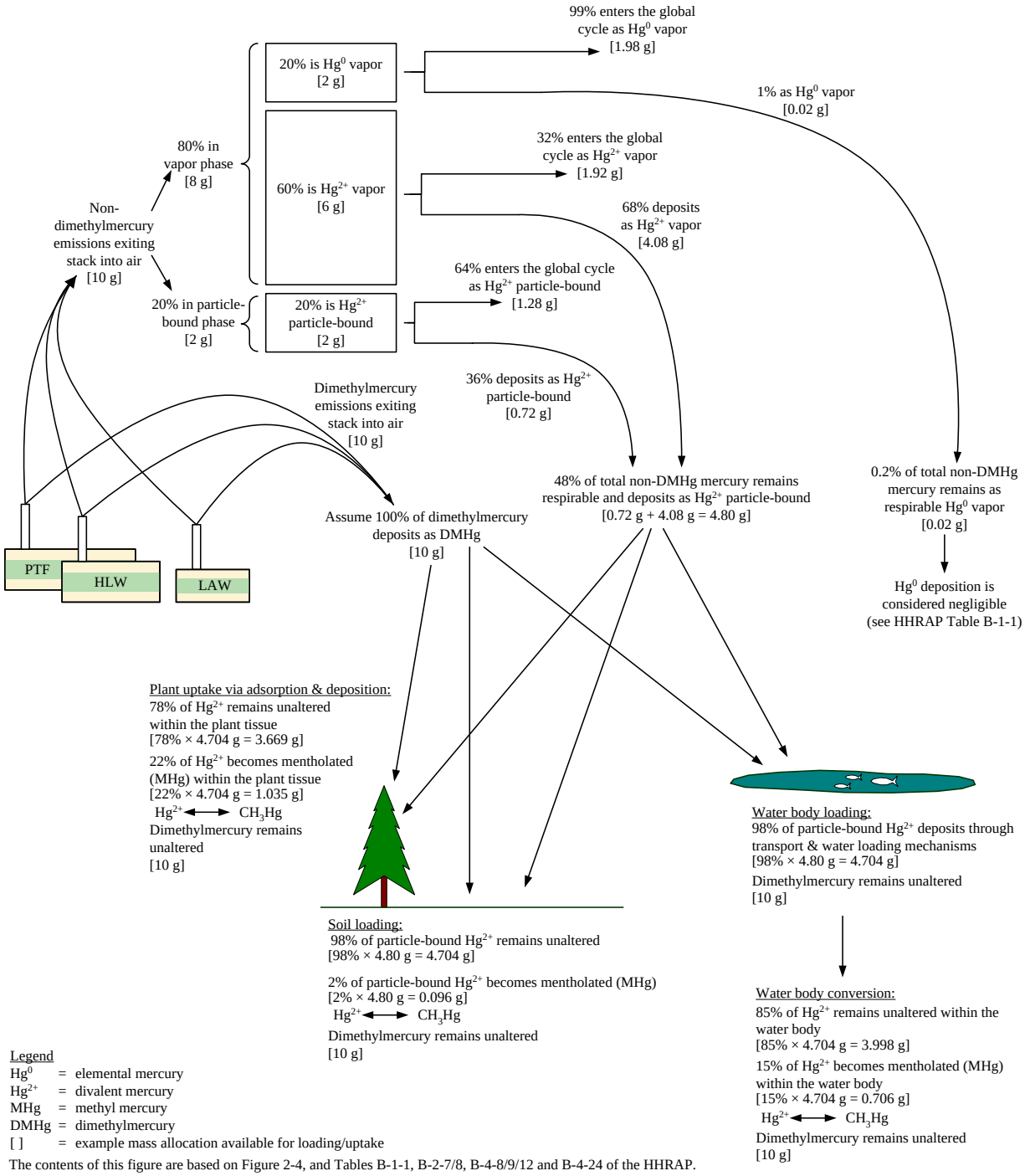


**Figure 6-8 Soil Concentration with Time**



## Figure 6-9 Mercury Emissions Fate and Transport Assumptions




The contents of this figure are based on Figure 2-4, and Tables B-1-1, B-2-7/8, B-4-8/9/12 and B-4-24 of the HHRAP.

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# 7 Screening Human Health Risk Assessment

The HHRA process includes four fundamental components: (1) data evaluation, (2) exposure assessment, (3) toxicity assessment, and (4) risk characterization. The data evaluation step is the selection of COPCs and ROPCs discussed in Section 4 of this work plan, and the quantification of emissions discussed in Section 5. Each of the remaining three components is discussed below:

- Exposure assessment – Section 7.1
- Toxicity assessment – Section 7.2
- Risk characterization – Section 7.4

The SLRA is designed to identify, early in the process, any potential risks associated with the WTP. The SLRA has been designed to overestimate potential risks by using conservative exposure assumptions combined with conservative toxicity values. The HHRA is one part of the SLRA that focuses on human health.

## 7.1 Exposure Assessment

Exposure assessment is the process of estimating the magnitude, frequency, duration, and type of potential exposures to COPCs and ROPCs. This section presents the exposure scenarios and approach for conducting the quantitative exposure assessment.

A human health conceptual exposure model identifies exposure scenarios that are defined by potentially exposed populations and exposure pathways. The conceptual exposure model used for this work plan is shown as Table 7-1 and was developed from guidance and information obtained from theHHRAP (EPA 2005a), the *Screening Assessment and Requirements for a Comprehensive Assessment: Columbia River Comprehensive Impact Assessment* (DOE-RL 1998), and discussions with DOE-ORP, Ecology, and EPA.

The conceptual exposure model focuses on identifying complete exposure pathways for potentially exposed populations. An exposure pathway is the means through which an individual may come in contact with a chemical in the environment. Exposure pathways are determined by:

- Environmental conditions (such as location of receptors, vegetative cover, and wind speed and direction)
- The potential for chemical migration through environmental media (such as soil, vegetation, or air)
- Lifestyles and work activities of potentially exposed populations

Although several potential pathways may exist, not all may be complete. For a pathway to be complete, all of the following four factors must exist:

- COPC or ROPC release into the environment
- Release and transport mechanism (such as deposition to soil) that moves the COPC or ROPC from the source to other locations
- Point of contact for receptors to be exposed to the affected media

- Exposure pathway, such as breathing vapors or ingesting affected media

These four factors were considered in the development of the conceptual exposure model. The sources of COPC and ROPC release are the WTP stacks. Transport processes, potential points of contact, and complete exposure pathways are identified to formulate exposure scenarios that will be the focus of the quantitative risk assessment. The process of exposure assessment is detailed in the following subsections.

### 7.1.1 Characterization of Exposure Setting

Characterizing the exposure setting is the first step in identifying potentially exposed populations. This characterization includes identifying the location of human populations within the study area and types of activities that can be expected under current and reasonable potential future land use.

The study area is defined as the area within a 50-km radius of the WTP. While it is possible for human populations to be exposed beyond this 50-km radius, the concentration of airborne and deposited emissions will be orders of magnitude less than those within the study area, essentially approaching zero. The EPA (2005a) reports that at other facilities, the most significant deposition of airborne emissions has been observed within a 3-km radius of a source. The Hanford site boundary extends approximately 9 km to 28 km from the WTP. The Columbia River is located approximately 8 km to more than 20 km from the WTP. Therefore, the potential for offsite impacts is expected to be minimal; however, because of the importance of the Columbia River as a potable water and recreational resource, it will be included in the quantitative risk assessment. Currently, no residential receptors are present on the Hanford site, nor are there likely to be any in the near future (i.e., within the next 50 years). Game animals that graze on Hanford site property and plants that grow on Hanford site property may be harvested by American Indians living off-site.

Characterization of the exposure setting includes the following:

- Characterization of the physical setting, including location of important physical features such as Gable Mountain, surface water bodies, and watersheds
- Characterization of potentially exposed populations, including identifying the location and activity patterns of current populations relative to the facility, determining plausible future land use, and identifying subpopulations of potential concern

Characterization will concentrate on the areas potentially most impacted by emissions, based on the results of the air-dispersion modeling and will include both current and future land use. The exposure assessment will focus on six locations of interest (see Figure 7-1):

- **Onsite ground maximum** – location of maximum predicted ground-level concentrations of both airborne and deposited emissions. This location generally represents worst-case human and ecological exposures because very few receptors are expected to be present here. The onsite ground maximum location is a 100 m × 100 m area represented by the point or points predicted to have the highest concentration of airborne and deposited emissions. The grid covers the area in the immediate region downwind from WTP. Receptors who are likely to receive exposures at this location are limited to onsite workers.
- **Hanford offsite** – location of predicted ground-level concentrations of both airborne and deposited emissions outside the Hanford Site boundary. This location represents a more plausible location for most human receptors and is an important point of compliance. The grid spans a region that is

predominately downwind of the WTP, adjacent to the southeastern border of the site. For conservatism, the residence of all receptors is assumed to occupy this grid.

- **Gable Mountain maximum** – location of maximum predicted ground-level concentrations of both airborne and deposited emissions at Gable Mountain. This location is included due to its importance to American Indian populations in the Oregon-Washington area. For purposes of assessing potential risks due to WTP emissions, Gable Mountain represents a site of tribal ceremonial activities and as such, receptor exposure is generally of a limited duration.
- **Columbia River maximum** – location of maximum predicted ground-level concentrations of both airborne and deposited emissions at the Columbia River. This location is used to evaluate potential risks to aquatic ecological receptors, as a source of potable water for human receptors, and as a source of fish for human receptors. The Columbia River receptor grid is predominately downwind of the WTP in the eastern region of the Hanford Site.
- **Hunter/Gatherer Area I (current timeframe)** – location of predicted ground-level concentrations of both airborne and deposited emissions for grazing game and native plant species. The subsistence resident American Indian is assumed to consume food (wild game, wildfowl, wildfowl eggs, and wild plants) harvested on site. Although the home range of most game and wild fowl is located primarily along the riparian corridor of the Columbia River, deer and other game may browse anywhere on site. The current exposure scenario hunting/gathering area (shown in Figure 7-2) includes the Hanford Reach National Monument and Gable Mountain. This area includes the portions of the Hanford site that could be used for grazing by game animals and wild fowl, and most likely to be used by people for collecting wild plants. It excludes the interior area of the Hanford Site.
- **Hunter/Gatherer Area II (future timeframe)** – expanded location of predicted ground-level concentrations of both airborne and deposited emissions for grazing game and native plant species. The hunter/gatherer receptor grid in the future timeframe is presumed to exist after shutdown of the WTP. Like the current hunting/gathering area assessed in current scenarios, this future area includes the riparian zones along the Columbia River, where game animals and important wild plants are likely to be present, and Gable Mountain. This future area also includes the area of maximum contaminant concentrations near the WTP (see Figure 7-2). This future hunting/gathering area is intended to provide a more conservative estimate of potential exposure and risk by including the area where concentrations are at their maximum. The hunter/gatherer area receptor grid does not include the exclusion area located at Hanford's central plateau or the industrial area east of the plateau. Note that because receptor access to this area is not assumed to occur until after WTP shutdown, exposure to airborne contaminants will not occur since there are no emissions.

Using the highest discrete values of $Cyv$, $Cyp_1$, $Cyp_{2.5}$, $Dydv$, $Dydp_1$, $Dydp_{2.5}$, $Dywv$, $Dywp_1$, and $Dywp_{2.5}$ from the offsite, onsite maximum, Columbia River, and Gable Mountain grids will certainly result in the highest degree of conservatism. However, in large receptor exposure grids such as the offsite grid, the corresponding exposures to such extreme deposition and air concentrations are improbable and could result in risk estimates that are highly improbable as well. As a result, the data from CALPOST were evaluated to provide some quantitative information with regard to the potential exposure to extreme air concentrations and depositions, and to aid in determining the appropriate CALPOST data for input in the risk assessment (24590-WPT-RPT-ENV-13-001). It was concluded that the 90th percentile of the predicted ground-level concentrations of both airborne and deposited emissions values for the Hanford offsite receptor exposure grid will sufficiently characterize exposures at that location. The appropriateness of using average deposition values to represent exposures to foods from the hunter/gatherer area was also evaluated. For the very large hunter/gatherer areas (where average exposures are of concern) the distribution-free 95 % upper confidence limit of the median provides a sufficiently conservative estimate of air concentration and deposition. For more localized exposure grids

such as at the onsite ground maximum, Columbia River, and Gable Mountain, the maximum predicted deposition or concentration provides a conservative estimate of exposure.

Figure 7-1 shows the locations of interest that will be the focus of the exposure assessment.

The subsistence resident American Indian is assumed to consume food (wild game, wildfowl, wildfowl eggs, and wild plants) harvested on site. The hunting and gathering areas for the subsistence resident American Indian are based on the following assumptions:

- The various types of plants eaten or used for ceremonial or medicinal purposes are collected from a variety of habitats (such as river corridor, foothills and mountains, meadow, and shrub-steppe). The exact collecting locations and types of plants collected are unknown; however, it is known that Gable Mountain is important for ceremonial activities, and plants are collected approximately once per year at the McGee ranch west of the 200 Areas.
- While onsite hunting is currently limited to the area north of the Columbia River, deer and other game may browse anywhere on site.
- The home range of deer at Hanford is located primarily along the riparian corridor of the Columbia River.
- The traditional subsistence lifestyle is a communal lifestyle; therefore, the hunting and gathering area must support more than a single individual or even a single family.
- A conservative scenario should include the locations of maximum emissions concentrations (ground maximum), and the locations of maximum emissions concentrations where it is known that some important activities occur (Gable Mountain maximum, Columbia River maximum).

To meet these needs, two hunting/gathering areas have been identified as described above. The current exposure scenario hunting/gathering area (shown in Figure 7-2) includes the Hanford Reach National Monument and Gable Mountain. This area includes the portions of the Hanford site most likely to be used for grazing by game animals, and most likely to be used by tribal members for collecting wild plants. The future exposure scenario hunting/gathering area (shown in Figure 7-2) includes the entire Hanford site excluding the 200 Area industrial zones. Like the current hunting/gathering area, this future area includes the riparian zones along the Columbia River, where game animals and important wild plants are likely to be present, and Gable Mountain. This future area also includes the area of maximum contaminant concentrations near the WTP. This future hunting/gathering area is intended to provide a more conservative estimate of potential exposure and risk by including the area where concentrations are at their maximum but food gathering activities are not likely to occur. For both of these hunting/gathering areas (see Figures 6-6 and 6-7) contaminant concentrations in soil will be estimated from an average soil concentration computed using the 95 % upper confidence limit of the median deposition. These soil concentrations will in turn be used to calculate contaminant concentrations in plant and animal tissue.

Receptors, locations, scenarios, and exposure pathways are summarized in Table 7-1.

### 7.1.2 Identification of Receptor Types

EPA (2005a) recommends that the following receptor types be evaluated for assessing potential risks from thermal treatment facilities:

- Resident (adult and child)

- Subsistence farmer (adult and child)
- Subsistence fisher (adult and child)
- Nursing infant
- Acute risk

The nursing infant scenario is recommended by EPA guidance (EPA 2005a) to address specific concerns regarding exposure to polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzo-p-furans (PCDFs) because these chemicals are known to accumulate in human milk. The EPA guidance recommends inclusion of the nursing infant only for PCDDs/PCDFs; however, coplanar PCBs will also be evaluated for this pathway in the SLRA due to their potential to behave, physiologically, like PCDDs/PCDFs. Because radionuclides are a major component of the waste to be treated at the facility, several ROPCs will also be evaluated for this pathway. The ROPCs strontium (Sr-90), iodine (I-129), and cesium (Cs-134 and Cs-137) will be evaluated for the nursing infant scenario. These radionuclides were selected because they are potentially present in the waste, are judged to have the highest potential for accumulation in milk, and due to their toxicity (CCN 064327). No other ROPCs will be evaluated for the nursing infant scenario, because other ROPCs have not been shown to accumulate in human milk. Not addressing additional COPCs for the nursing infant scenario is a nonconservative uncertainty in the risk assessment. If the potential risks or hazards for other COPCs in other exposure pathways approach unacceptable levels, and the data are available to evaluate infant exposures, further consideration will be given to incorporating those COPCs into the nursing infant scenario. Nursing infant exposure will be evaluated based on intakes for the Hanford site industrial worker, resident adult, resident subsistence farmer adult, resident subsistence fisher adult, and resident subsistence American Indian adult.

For purposes of this workplan, special subpopulations are defined as individual human beings or subsets of the general population that may potentially be at higher risk due to lifestyle activities that cause higher exposures to COPCs and ROPCs. To address potentially sensitive subpopulations, the following additional exposure scenarios will be evaluated:

- Hanford site industrial worker
- A resident subsistence American Indian (adult and child)

Workers employed at the WTP will not be included in the risk assessment because other regulations exist for occupational exposures within the WTP boundary (e.g., DOE standards for occupational safety and health). However, because of the WTP's location within the Hanford site, the closest and most likely receptors are other Hanford site workers located outside the WTP boundary. Therefore, the Hanford site industrial worker scenario will be included in the risk assessment.

American Indian tribes (Nez Perce Tribe, Yakama Indian Nation, and CTUIR) ceded the land currently occupied by the US government's Hanford site. However, these tribes retained the rights to the use of resources on this land. Representatives of these tribes have expressed a desire to be able to use this land to conduct a traditional lifestyle. A wide range of possible tribal activities related to traditional lifestyles exists. The resident subsistence American Indian scenario will address a variety of potential exposures associated with food gathering (including hunting, fishing, and plant gathering), as well as cultural and social activities (e.g., use of a sweat lodge).

The exposure scenarios included in the quantitative risk assessment are designed to cover a wide range of possible receptor activities, age groups, and lifestyles. These receptors represent the most highly exposed

populations that could work or live near the Hanford site, including adult workers, adult and child residents and farmers, and American Indians living a traditional lifestyle. The exposure assessment and risk characterization results for the selected receptors are designed to be protective of other populations and special subpopulations of interest. For example, the resident child receptor provides a bounding estimate for other child activities, such as children attending school or daycare. This scenario assumes a high level of potential exposure (e.g., the resident child is present 7 days per week, 24 hours per day and ingests homegrown produce) at an offsite location of elevated contaminant concentration. Hence, risk-management decisions based on these conservative assumptions will be protective of other child populations (e.g., at a school or daycare center where exposure would be less because a child may be present 5 days per week for less than 12 hours per day). Other possible special subpopulations are being evaluated by identifying their locations and determining whether they are located in areas that are potentially at risk from WTP emissions. Figure 7-3 provides a map showing (1) the location of the WTP, (2) the locations of potential receptor populations (such as cities), and (3) locations of potentially sensitive subpopulations (such as daycare centers and preschools, schools, hospitals, and nursing homes). Figure 7-4 summarizes the receptors, locations, exposure scenarios, and exposure pathways.

### 7.1.3 Description of Exposure Scenarios

Exposure scenarios to be addressed by the risk assessment are described in more detail below and are summarized in Table 7-1. Exposure scenarios are defined for current and future land-use conditions. For the SLRA, *current* and *future* are defined as follows:

**Current Land Use.** For this work plan, *current* is defined as the 40-year operating lifetime of the WTP, anticipated to begin by 2019. This period corresponds approximately to the period addressed by the *Final Hanford Comprehensive Land-Use Plan (CLUP) Environmental Impact Statement* (DOE 1999) of at least 50 years from publication of the Record of Decision (ROD) (Federal Register 1999 [64 FR 61615]), that is, 1999 through 2049.

Current land use within the 50 km study area is characterized based on aerial photographs, zoning maps, land development plans, and information presented in the CLUP and the preferred land use alternative identified in *Record of Decision (ROD): Hanford Comprehensive Land-Use Plan Environmental Impact Statement* (64 FR 61615).

Figure 7-5 shows existing land use within the study area as of 1996 (DOE 1999). Figure 7-6 shows current (i.e., over approximately the next 40 years) land use on the Hanford Site as defined by the CLUP. Representatives of the Nez Perce Tribe, Yakama Indian Nation, and CTUIR are also being consulted in evaluating current land use in the study area.

**Future Land Use.** For this work plan, future exposure scenarios begin after WTP shutdown (following 40 years of operation). For example, the future resident subsistence farmer is assumed to be exposed from year 40 to year 80.

Plausible future land use is characterized based on information presented in the documents listed above. Representatives of the Nez Perce Tribe, Yakama Indian Nation, and CTUIR are also being consulted in evaluating future land use in the study area.

In addition to the information in DOE 1999, DOE has indicated that:

- The 200 Areas (a.k.a. central plateau) will remain industrial past the 50-year timeframe of the CLUP because mixed waste has been, and will continue to be, buried there as a result of remedial activities at the rest of the Hanford site.
- There will not be any onsite residential development (within the Hanford site boundary) in the foreseeable future

Both current and future land-use assumptions must also consider the newly created Hanford Reach National Monument, which includes the Saddle Mountain National Wildlife Refuge north of the Columbia River and the Fitzner-Eberhardt Arid Lands Ecology (FEALE) Reserve in the western portion of the Hanford Site.

Within these timeframes, exposure scenarios may be classified as being either plausible or worst-case as defined below.

Plausible exposure scenarios represent receptors that currently exist, or may reasonably be expected to exist, at a given location. For example, workers are currently present in the 200 Areas; therefore, the Hanford site industrial worker is a current plausible exposure scenario at that location. This does not mean that the exposure scenario as described here (a worker present at the point of maximum emissions concentration, 8 hours per day, 250 days per year, for 20 years) portrays actual current workers, but rather, that this type of receptor (onsite worker) is plausible at that location.

Worst-case exposure scenarios represent receptors that are not reasonably expected to exist now or in the future at the specified location. For example, a resident subsistence farmer will be evaluated as a future worst-case receptor at the point of elevated offsite emissions concentrations, but it is unlikely that such a receptor (a totally self-sustaining farmer) will ever exist at this location.

#### 7.1.3.1 Hanford Site Industrial Worker

**General Description**

This receptor is an adult worker employed near the WTP and living offsite. Workers employed at the WTP will not be included in the risk assessment because other regulations exist for occupational exposures within the WTP boundary (such as DOE standards for occupational safety and health). The Hanford site industrial worker will be evaluated using occupational exposure assumptions primarily from DOE-RL 1995 and residential exposure assumptions primarily from EPA (2005a, 2003, CCN 063810, CCN 063807, CCN 063805, CCN 064331, CCN 063806, CCN 063816), as described in Section 7.1.6.

**Exposure Pathways**

The Hanford site industrial worker is exposed on site (during work hours) through inhalation of emissions, ingestion of soil, inhalation of resuspended soil, ingestion of drinking water, and external exposure to radionuclides in air and soil. This worker is also assumed to be exposed while at home through these same pathways and through ingestion of homegrown produce.

**Exposure Location**

This receptor is assumed to work at the onsite ground maximum. The onsite ground maximum location is a 100 m × 100 m area (defined in Section 6.1) represented by the point or points predicted to have the highest concentration of airborne and deposited emissions. This receptor is assumed to live within the Hanford offsite receptor exposure grid. This offsite location is a 500 m × 500 m area represented by the

90th percentile of airborne and deposited emissions. The Hanford site industrial worker is assumed to obtain drinking water from the Columbia River maximum. Exposure of a Hanford site industrial worker is considered a plausible scenario since workers are present in this area and may live off site.

**Current Exposure Timeframe**

The Hanford site industrial worker is assumed to have a 20-year career. It is also assumed that a retired worker will live within the offsite location for 10 years after retirement. The retiree is assumed to be exposed through the same exposure pathways experienced by a worker while at home when not working. However, it is not necessarily assumed that the worker and the retiree are the same individual. A conservative approach to assessing potential risk is to assume that both the worker and retiree are exposed during the time of maximum potential soil concentration. Recall from Section 6.2 that the time of maximum potential soil concentration occurs at plant shutdown after 40 years of deposition. Thus, it is assumed that for a plant operating from year 0 to year 40, the Hanford site industrial worker is assumed in the current scenario to be exposed from years 20 to 40. Likewise, the retired worker is assumed to be exposed from year 30 to 40 (refer to Figure 7-7). For simplicity, the risk to the worker and retiree will be added and compared to thresholds. If an exceedance occurs, the individual risks of the worker and retiree will be reported and compared to applicable thresholds.

The worker is assumed to be at work (onsite maximum) for 8 hours each weekday (excluding holidays and yearly vacation), resulting in an exposure frequency of 250 days/year. The worker is assumed to be at home (offsite) during the remaining 16 hours of the workday. Additionally, the worker is assumed to spend a cumulative 100 days at home (offsite) each year during weekends. It is assumed that the worker is neither at work nor home for 15 days/year, presumably on holiday and/or vacation outside of the area of assessment. Unlike the worker, the retiree is assumed to spend 24 hours/day, 350 days/year at home and 15 days/year on holiday and/or vacation outside of the area of assessment.

**Future Exposure Timeframe**

As stated previously, the Hanford site industrial worker is assumed to have a 20-year exposure and the retiree a 10-year exposure. From Section 6.2 the time of maximum potential soil concentration occurs at plant shutdown after 40 years of deposition, and that soil concentrations then gradually decrease due to soil loss. The conservative approach is to assume that both the worker and retiree are exposed immediately after plant shutdown. The Hanford site industrial worker is assumed to be exposed from years 40 to 60 in the future scenario. The retired worker is assumed to be exposed from year 40 to 50.

Worker and retiree exposure time (hours per day) and frequency (days per year) are the same as the current exposure scenario.

#### 7.1.3.2 Nursing Infant of Hanford Site Industrial Worker

**General Description**

The nursing infant of the Hanford site industrial worker is the infant of the worker described above.

**Exposure Pathways**

The nursing infant of the Hanford site industrial worker is assumed to be exposed to PCDDs/PCDFs, PCBs, and four ROPCs through ingestion of breast milk from the worker exposed through:

- Inhalation of emissions, ingestion of soil, inhalation of resuspended soil, ingestion of drinking water, and external exposure to radionuclides in air and soil while at work

- Inhalation of emissions, ingestion of soil, inhalation of resuspended soil, ingestion of drinking water, external exposure to radionuclides in air and soil, and ingestion of homegrown produce while at home

**Exposure Location and Timeframe**

The nursing infant of the Hanford site industrial worker is assumed to reside with the worker described above at the Hanford offsite receptor exposure grid. It is assumed that the mother continues to work during the period of nursing and is exposed to the onsite maximum during the workday, and is exposed offsite while at home. Likewise, the timeframe for exposure of the lactating mother is assumed to be the same as the current worker (years 20 to 40). The timeframe for exposure of the lactating mother in a future exposure scenario is assumed to be the same as the future worker (years 40 to 60). Exposure assessment for the nursing infant does not include an assessment of intake from retired individuals. The current and future exposure of a nursing infant of the Hanford site industrial worker is considered a plausible scenario since workers are present in this area and may live (be a resident) off site.

#### 7.1.3.3 Resident

**General Description**

The resident is assumed to live, work, and play at a single location and, thus, is assumed to be home 24 hours per day, 350 days per year, with 2 weeks per year on vacation or otherwise away from the home. This receptor is assumed to have a garden that supplies fruit and vegetables. Both an adult and a child resident will be evaluated using EPA default (2005a) and other EPA-recommended (CCN 063810, CCN 063807, CCN 063805, CCN 063806) exposure assumptions described in Section 7.1.6.

**Exposure Pathways**

The resident (adult and child) is assumed to be exposed through direct inhalation of airborne emissions, ingestion of soil, inhalation of resuspended soil, ingestion of drinking water, external exposure to radionuclides in air and soil, and ingestion of homegrown produce. This receptor is assumed to have a garden that supplies homegrown fruit and vegetables. The resident is assumed to obtain drinking water from the point of maximum concentration in the Columbia River.

**Exposure Location**

The closest resident at the time of this work plan (2013) is located more than 20 km from the WTP. However, in this work plan, *current* is defined as the 40-year operating lifetime of the WTP (approximately beginning in 2019). Therefore, it is assumed that a current resident may be located within the Hanford offsite receptor exposure grid sometime during this 40-year period. This offsite location is a 500 m × 500 m area represented by the 90th percentile of airborne and deposited emissions. The resident is assumed to obtain drinking water from the point of maximum concentration in the Columbia River. Exposure of a resident at the Hanford offsite maximum is considered a plausible scenario since residents are present outside the site boundary and residential development could occur within the offsite grid within the next 40 years.

**Current Exposure Timeframe**

The resident is assumed to have a 30-year exposure duration. Since the time of maximum potential soil concentration occurs at plant shut down after 40 years of deposition, it is assumed that for a plant operating from year 0 to 40, the resident is assumed to be exposed from years 10 to 40 in the current

scenario. Likewise, the resident child, who has a 6-year exposure duration, is assumed to be exposed from years 34 to 40 in the current scenario (child and adult are not necessarily the same individual).

The resident (adult and child) is assumed to be exposed 24 hours/day, 350 days/year (15 days/year on holiday and/or vacation outside of the area of assessment).

**Future Exposure Timeframe**

Since the resident is assumed to have a 30-year exposure duration, the future resident is assumed to be exposed from years 40 to 70. The resident child, who has a 6-year exposure duration, is assumed to be exposed from years 40 to 46 in the future scenario (child and adult are not necessarily the same individual).

The resident (adult and child) exposure time (hours per day) and frequency (days per year) are the same as the current exposure scenario.

#### 7.1.3.4 Nursing Infant of Resident

**General Description**

The nursing infant of the resident is the infant of the adult resident described above.

**Exposure Pathways**

The nursing infant of the resident is assumed to be exposed to PCDDs/PCDFs, PCBs, and four ROPCs through ingestion of breast milk from the adult resident exposed through inhalation of emissions, ingestion of soil, inhalation of resuspended soil, ingestion of drinking water, and ingestion of homegrown produce.

**Exposure Location and Timeframe**

The nursing infant of the resident is assumed to reside with the resident described above at the Hanford offsite location during the same period of resident exposure. The timeframe for exposure of the lactating mother is assumed to be the same as the current resident (years 10 to 40). The timeframe for exposure of the lactating mother in a future exposure scenario is assumed to be the same as the future resident (years 40 to 70). The current and future exposure of a nursing infant of the resident within the Hanford offsite receptor exposure grid is considered a plausible scenario since residents are present outside the site boundary and residential development could occur offsite within the next 40 years.

#### 7.1.3.5 Resident Subsistence Farmer

**General Description**

The resident subsistence farmer is assumed to live, work, and play at a single location (i.e., the resident farmer is assumed to be at home 24 hours per day, 350 days per year, with 2 weeks per year on vacation or otherwise away from the home). This receptor is assumed to maintain a farm that supplies his or her produce (fruit and vegetable), meat (beef, pork, and poultry), dairy products, and eggs. Both an adult and a child resident subsistence farmer will be evaluated using EPA default (2005a) and other EPA-recommended (CCN 063807, CCN 064331, CCN 063806, CCN 063804) exposure assumptions described in Section 7.1.6.

**Exposure Pathways**

The resident subsistence farmer (adult and child) is assumed to be exposed through inhalation of emissions, ingestion of soil, inhalation of resuspended soil, ingestion of drinking water, external exposure to radionuclides in air and soil, and ingestion of homegrown produce, beef, pork, poultry, dairy products, and eggs.

**Exposure Location**

The closest resident at the time of this work plan (2011) is located more than 20 km from the WTP. However, in this work plan, *current* is defined as the 40-year operating lifetime of the WTP (beginning in approximately 2019). Therefore, it is assumed that a current resident subsistence farmer may be located at the Hanford offsite receptor exposure grid. This offsite location is a 500 m × 500 m area represented by the 90th percentile of airborne and deposited emissions. The resident subsistence farmer is assumed to obtain drinking water from the point of maximum concentration in the Columbia River. Exposure of a resident subsistence farmer at the Hanford offsite location is considered a worst-case scenario because, while resident farmers may be present outside the site boundary, the defined exposure scenario (i.e., a farmer producing his or her own food, as described in Section 7.1.6.2, within a 500 m × 500 m area) is unlikely.

**Current Exposure Timeframe**

The resident subsistence farmer is assumed to have a 40-year exposure duration. Since the time of maximum potential soil concentration occurs at plant shut down after 40 years of deposition, it is assumed that for a plant operating from year 0 to 40, the resident subsistence farmer is assumed to be exposed throughout the entire duration of plant operation (years 0 to 40). The resident subsistence farmer child, who has a 6-year exposure duration, is assumed to be exposed from years 34 to 40 in the current scenario (child and adult are not necessarily the same individual).

The resident subsistence farmer (adult and child) is assumed to be exposed 24 hours/day, 350 days/year (15 days/year on holiday and/or vacation outside of the area of assessment).

**Future Exposure Timeframe**

Since the resident subsistence farmer is assumed to have a 40-year exposure duration, the future resident subsistence farmer is assumed to be exposed from years 40 to 80. The resident subsistence farmer child, who has a 6-year exposure duration, is assumed to be exposed from years 40 to 46 in the future scenario (child and adult are not necessarily the same individual).

The resident subsistence farmer (adult and child) exposure time (hours per day) and frequency (days per year) are the same as the current exposure scenario.

#### 7.1.3.6 Nursing Infant of Resident Subsistence Farmer

**General Description**

The nursing infant of the resident subsistence farmer is the infant of the adult resident subsistence farmer described above.

**Exposure Pathways**

The nursing infant of the resident subsistence farmer is assumed to be exposed to PCDDs/PCDFs, PCBs, and four ROPCs through ingestion of breast milk from the adult resident subsistence farmer exposed through inhalation of emissions, ingestion of soil, inhalation of resuspended soil, ingestion of drinking water, and ingestion of homegrown produce, beef, pork, poultry, dairy products, and eggs.

**Exposure Location**

The nursing infant of the resident subsistence farmer is assumed to reside with the resident subsistence farmer described above within the Hanford offsite receptor exposure grid during the same period of resident subsistence farmer exposure. The current and future exposure of a nursing infant of the resident subsistence farmer at the Hanford offsite location is considered a worst-case scenario because, while resident farmers may be present outside the site boundary, the defined exposure scenario (i.e., an infant nursed for one year by a farmer producing her own food at a single grid node) is unlikely.

#### 7.1.3.7 Resident Subsistence Fisher

**General Description**

The resident subsistence fisher scenario is the same as the resident scenario with the addition of fish ingestion. This receptor is assumed to live, work, and play at a single location (i.e., the resident fisher is assumed to be at home 24 hours per day, 350 days per year, with 2 weeks per year on vacation or otherwise away from the home). This receptor is assumed to have a garden that supplies fruit and vegetables and to obtain fish from the Columbia River. Both an adult and a child resident subsistence fisher will be evaluated using the EPA default (2005a) and other EPA-recommended (CCN 063810, CCN 063807, CCN 063805, CCN 063806) exposure assumptions described in Section 7.1.6.

**Exposure Pathways**

The resident subsistence fisher (adult and child) is assumed to be exposed through inhalation of emissions, ingestion of soil, inhalation of resuspended soil, ingestion of drinking water, external exposure to radionuclides in air and soil, ingestion of homegrown produce, and ingestion of locally caught fish.

**Exposure Location**

The closest resident at the time of this work plan (2013) is located more than 20 km from the WTP. However, for this risk assessment work plan (RAWP), *current* is defined as the 40-year operating lifetime of the WTP (beginning in approximately 2019). Therefore, it is assumed that a current resident subsistence fisher may be located at the Hanford offsite receptor exposure grid sometime during this 40-year period. This offsite location is a 500 m × 500 m area represented by the 90th percentile of airborne and deposited emissions. The resident subsistence fisher is assumed to obtain drinking water and fish from the point of maximum concentration in the Columbia River. Exposure of a resident subsistence fisher within the Hanford offsite exposure grid is considered a worst-case scenario because, while residents might be present outside the site boundary and fish the Columbia River, the defined exposure scenario (i.e., a fisher growing fruit and vegetables and ingesting locally caught fish every day) is unlikely.

**Current Exposure Timeframe**

The resident subsistence fisher is assumed to have a 30-year exposure duration. Since the time of maximum potential soil concentration occurs at plant shut down after 40 years of deposition, it is assumed

that for a plant operating from year 0 to 40, the resident subsistence fisher is assumed to be exposed from years 10 to 40. The resident subsistence fisher child, who has a 6-year exposure duration, is assumed to be exposed from years 34 to 40 in the current scenario (child and adult are not necessarily the same individual).

The resident subsistence fisher (adult and child) is assumed to be exposed 24 hours/day, 350 days/year (15 days/year on holiday and/or vacation outside of the area of assessment).

**Future Exposure Timeframe**

Since the resident subsistence fisher is assumed to have a 30-year exposure duration, the future resident subsistence fisher is assumed to be exposed from years 40 to 70. The resident subsistence fisher child, who has a 6-year exposure duration, is assumed to be exposed from years 40 to 46 in the future scenario (child and adult are not necessarily the same individual).

The resident subsistence fisher (adult and child) exposure time (hours per day) and frequency (days per year) are the same as the current exposure scenario.

#### 7.1.3.8 Nursing Infant of Resident Subsistence Fisher

**General Description**

The nursing infant of the resident subsistence fisher is the infant of the adult resident subsistence fisher described above.

**Exposure Pathways**

The nursing infant of the resident subsistence fisher is assumed to be exposed to PCDDs/PCDFs, PCBs, and four ROPCs via ingestion of breast milk from the adult resident subsistence fisher exposed via inhalation of emissions, ingestion of soil, inhalation of resuspended soil, ingestion of drinking water, external exposure to radionuclides in air and soil, ingestion of homegrown produce, and ingestion of locally caught fish.

**Exposure Location**

The nursing infant of the resident subsistence fisher is assumed to reside with the resident subsistence fisher described above within the Hanford offsite receptor exposure grid during the same period of resident subsistence fisher exposure. The current and future exposure of a nursing infant of the resident subsistence fisher within the Hanford offsite location is considered a worst-case scenario as described above for the resident subsistence fisher.

#### 7.1.3.9 Resident Subsistence American Indian

**General Description**

The resident subsistence American Indian refers to the American Indian hunter-gatherer exposure scenario originally developed for the *Final Tank Closure and Waste Management Environmental Impact Statement for the Hanford Site* (TC&WM EIS, DOE 2012), adapted for the WTP risk assessment. The American Indian scenario exposure parameters presented in the TC&WM EIS were used where available. Other parameters were taken from the *Tank Waste Remediation System, Hanford Site, Richland, Washington, Final Environmental Impact Statement* (TWRS EIS, DOE 1996) or from EPA's *Exposure*

*Factors Handbook* (EFH[1], EPA 1997a). Children's exposure parameters were developed by proportioning the child caloric intake reported in EPA guidance (*Child-Specific Exposure Factors Handbook* [CSEFH, EPA 2008]), according to the various proportions of meat, vegetable, roots, etc. in the diet of the adult American Indian as reported in the TC&WM EIS.

The resident subsistence American Indian is assumed to live a traditional subsistence lifestyle. The traditional lifestyles of the Nez Perce, Yakama Indian Nation, and CTUIR were historically based on a seasonal cycle of travel among hunting, plant gathering, and fishing areas. The most common foods were salmon, roots (including camas bulb, bitterroot, wild carrot, and wild potato), berries (including service berries, gooseberries, huckleberries, chokecherries, and wild strawberries), deer, and elk. Each of these foods was collected in different locations during different seasons. The seasonal cycle of food gathering encompassed a large area including the lowlands along the Columbia River and its tributaries, foothills, and prairies, and higher mountainous areas. Presently, tribal members may hunt in areas such as the North Slope (a.k.a. Saddle Mountain National Wildlife Refuge, north of the Columbia River), fish near the Vernita bridge (where the Columbia River enters the western boundary of the Hanford site), and occasionally gather food at sites such as the McGee ranch (south of the Columbia River at the western boundary of the Hanford site). Members of the three tribes potentially impacted at Hanford would be individuals pursuing a traditional lifestyle. The traditional lifestyle of these three tribes is heavily dependent on fish (primarily salmon) in addition to game and plants; therefore, a separate hunter/gatherer and fisher would not exist. A more realistic receptor is a combination hunter/gatherer/fisher.

The resident subsistence American Indian is assumed to hunt and fish to supply his or her meat (game and wildfowl), egg (from wildfowl), and fish needs, and to gather native plants to supply his or her fruit and vegetable needs. Both an adult and a child resident subsistence American Indian will be evaluated.

**Exposure Pathways**

The resident subsistence American Indian (adult and child) is assumed to be exposed through inhalation of emissions; ingestion of soil; inhalation of resuspended soil; ingestion of drinking water; external exposure to radionuclides in air and soil; and ingestion of wild plants, game, wildfowl, fish, and wildfowl eggs. The consumption of livestock, dairy products, and domestic produce does not occur in this scenario. In addition to these pathways, the resident subsistence American Indian is also assumed to be exposed through inhalation and dermal exposure to resuspended constituents from water in a sweat lodge.

**Exposure Location**

The resident subsistence American Indian is assumed to live offsite within the Hanford offsite receptor exposure grid, consume fish from the Columbia River maximum, and consume wild game, wildfowl, wildfowl eggs, and plants harvested on the hunting/gathering area. The resident subsistence American Indian is also assumed to obtain drinking water and water for use in a traditional sweat lodge from the Columbia River maximum. The locations for each of these activities are described in more detail in Section 7.1.1.

The resident subsistence American Indian is assumed to live a traditional subsistence lifestyle, based on seasonal hunting, plant gathering, and fishing. This receptor is assumed to live, work, and play at a single

[1] The 1997 version of the *Exposure Factors Handbook* was the version available at the time of the development of this RAWP. In October 2011, EPA released *Exposure Factors Handbook: 2011 Edition*. Data from this later version will be used to supersede like data from the 1997 version (as needed) as part of the FRA. This update is not anticipated to significantly alter the risk assessment results.

location continuously (i.e., the resident subsistence American Indian is assumed to be at home 24 hours per day, 365 days per year). It is assumed that the resident subsistence American Indian's home is located within the offsite receptor grid. This offsite location is a 500 m × 500 m area represented by the 90th percentile of airborne and deposited emissions.

It is also assumed that consumption of food (wild game, wildfowl, wildfowl eggs, and wild plants) from the hunter/gatherer location, and consumption of fish from the Columbia River maximum, occur every day (365 days/year). The current and future hunting and gathering areas are defined as described above and shown in Figure 7-2 (i.e., [1] the Hanford Reach National Monument and Gable Mountain, and [2] the entire Hanford site). The assumption is that the time spent hunting, gathering, and fishing is negligible relative to the time spent at the offsite location, hence, there is no exposure time or exposure frequency associated with those particular locations. Furthermore, it is also assumed that adults and children spend 1 hour/day in a sweat lodge at the applicable exposure locations (offsite), but that the water used for the sweat lodge is from the Columbia River maximum as described above. Table 7-1 should be consulted for a summary of pathways and associated exposure locations.

This approach is conservative because it includes the points of maximum concentration, expected to be located east of the 200 East Area, as well as the areas west and north of the 200 East Area where actual hunting, gathering, and fishing activities currently occur.

Exposure of the resident subsistence American Indian is considered a plausible scenario since (1) residents are present outside the site boundary and development could occur within the offsite location, and (2) American Indian people are presently allowed to access the Hanford site; however, this access is limited to individuals with security badges, and then only for limited purposes, such as religious purposes or to gather foods (approximately once per year) for ceremonies.

**Current Exposure Timeframe**

The resident subsistence American Indian is assumed to have a lifetime (70-year) exposure duration. Since the time of maximum potential soil concentration occurs at plant shut down after 40 years of deposition, it is assumed that for a plant operating from year 0 to 40, the resident subsistence American Indian is exposed throughout the period of plant operation and subsequent to plant shutdown (years 0 to 70). However, the resident subsistence American Indian child, who has a 6-year exposure duration, is assumed to be exposed from years 34 to 40 in the current scenario (child and adult are not necessarily the same individual).

**Future Exposure Timeframe**

Since the resident subsistence American Indian is assumed to have a 70-year exposure duration, the future resident subsistence American Indian is assumed to be exposed from years 40 to 110. The resident subsistence American Indian child, who has a 6-year exposure duration, is assumed to be exposed from years 40 to 46 in the future scenario (child and adult are not necessarily the same individual).

The resident subsistence American Indian (adult and child) exposure time (hours per day) and frequency (days per year) are the same as the current exposure scenario.

**Alternate Resident Subsistence American Indian, Scenario #1**

There are two alternate resident subsistence American Indian scenarios are included in this SLRA that are not necessarily endorsed by the DOE-ORP. These receptor scenarios were developed to more closely represent the lifestyle described by guidance documents issued by local tribes. The lifestyle and exposure

parameters of the first alternate resident subsistence American Indian are primarily based on data from *Exposure Scenario for CTUIR Traditional Subsistence Lifeways* (Harris and Harper 2004) and *Application of the CTUIR Traditional Lifeways Exposure Scenario in Hanford Risk Assessments* (Harris 2008). Other parameters were taken from the "A Native American Exposure Scenario" (Harris and Harper 1997) or from the EFH. Children's exposure parameters were developed by proportioning the child caloric intake reported in the CSEFH according to the various proportions of meat, vegetable, roots, etc. in the diet of the adult tribal member as reported in the guidance documents provided by the CTUIR. In order to assess the potential for exposure to a tribal member whose lifestyle differs from the resident subsistence American Indian described above, the uncertainty assessment of the PRA will include evaluation of the alternate resident subsistence American Indian exposure scenario described below.

The alternate resident subsistence American Indian #1 is assumed to live a traditional subsistence lifestyle, based on a seasonal hunting, plant gathering, and fishing. This receptor is assumed to live, work, and play at a single location (i.e., assumed to be at home 24 hours per day, 353 days per year), but spends one day a month away from home participating in ceremonial activities (assumed to occur at Gable Mountain[2]). It is assumed that the resident's home is located within the offsite grid. The alternate resident subsistence American Indian #1 (adult and child) is assumed to be exposed through inhalation of emissions, ingestion of soil, inhalation of resuspended soil, ingestion of drinking water, and ingestion of wild plants, game (including game organs), wild fowl and fish (including fish organs). The consumption of livestock, dairy products, and domestic produce does not occur in this scenario. The alternate resident subsistence American Indian #1 is assumed to obtain fish, drinking water and water for sweat lodge activities from the point of maximum concentration in the Columbia River. No credit is given for soil and air exposures away from home (e.g., while hunting, gathering, fishing, or for ceremonial and cultural activities outside of the assessment area), except for the time spent at ceremonial activities at Gable Mountain. The exposure timing of this scenario is the same as the resident subsistence American Indian scenario described above; adult exposures in the current timeframe are assessed over a 70-year duration that spans the 40 year operational period of WTP and 30 years following its shutdown. Child exposures are assessed through age 6. Future timeframe exposures are assumed to occur immediately after plant shutdown.

**Alternate Resident Subsistence American Indian, Scenario #2**

The alternate resident subsistence American Indian #2 refers to the second of two tribal lifestyles modeled in this exposure assessment that is not necessarily endorsed by the DOE-ORP. The lifestyle and exposure parameters of the second alternate resident subsistence American Indian are primarily based on data from *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment* (RUDOLFI Inc. 2007). Other parameters were taken from the EFH. Children's exposure parameters were developed by proportioning the child caloric intake reported in the CSEFH according to the various proportions of meat, vegetable, roots, etc., in the diet of the adult tribal member as reported in the guidance documents provided by the Yakama Nation. In order to assess the potential for exposure to a resident subsistence American Indian whose lifestyle differs from the resident subsistence American Indian receptor described above, the uncertainty assessment of the PRA will include evaluation of the alternate resident subsistence American Indian exposure scenario described below.

[2] The actual location of tribal ceremonial activities varies with the nature of the activity, and is considered confidential. Gable Mountain is chosen as the location for ceremonial activities in the WTP risk assessment because it is a place of significance and is in close proximity to WTP, making related exposures at this location conservative and bounding.

The alternate resident subsistence American Indian #2 is assumed to live a traditional subsistence lifestyle, based on consumption of homegrown produce and livestock, in addition to seasonal hunting, plant gathering, and fishing. This receptor is assumed to live, work, and play at a single location continuously (i.e., the resident subsistence American Indian is assumed to be at home 24 hours per day, 365 days per year). It is assumed that the resident's home is located within the offsite grid. The alternate resident subsistence American Indian #2 (adult and child) is assumed to be exposed through inhalation of emissions, ingestion of soil, inhalation of resuspended soil, ingestion of drinking water. This receptor's diet includes domestic produce and livestock, supplemented with wild produce, game, and fish. To fully bound the estimated risk, it is assumed that all domestic produce and livestock is homegrown, and that all wild produce and game is taken from the hunter/gatherer area. The alternate resident subsistence American Indian #2 is assumed to obtain fish, drinking water and water for sweat lodge activities from the point of maximum concentration in the Columbia River. No credit is given for soil and air exposures away from home (e.g., while hunting, gathering, fishing, or for ceremonial and cultural activities away from home). The exposure timing of this scenario is the same as the resident subsistence American Indian scenario described above; adult exposures in the current timeframe are assessed over a 70 year duration that spans the 40 year operational period of WTP and 30 years following its shutdown. Child exposures are assessed through age 6. Future timeframe exposures are assumed to occur immediately after plant shutdown.

#### 7.1.3.10 Nursing Infant of Resident Subsistence American Indian

**General Description**

The nursing infant of the subsistence tribal resident is the infant of the adult resident subsistence American Indian described above.

**Exposure Pathways**

The nursing infant of the resident subsistence American Indian is assumed to be exposed to PCDDs/PCDFs, coplanar PCBs, and four ROPCs through ingestion of breast milk from the adult resident subsistence American Indian exposed through inhalation of emissions; ingestion of soil; inhalation of resuspended soil; ingestion of drinking water; and ingestion of traditional foods. It is assumed that the mother does not participate in sweat lodge activities during the lactation period, and is therefore not exposed to pathways associated with the sweat lodge.

**Exposure Location**

The nursing infant of the resident subsistence American Indian is assumed to reside with the resident subsistence American Indian described above within the Hanford offsite location. The current and future exposure of a nursing infant of the resident subsistence American Indian within the Hanford offsite grid is considered a plausible scenario because residents are present outside the Hanford site boundary and development could occur at the offsite maximum point or points within the next 40 years.

#### 7.1.3.11 Acute Exposure

The EPA (2005a) recommends evaluating potential acute exposures in addition to the chronic exposures evaluated by previously described exposure scenarios. The acute exposure scenario includes direct inhalation of airborne COPC and ROPC emissions and exposure to external radiation from airborne ROPC emissions at the estimated maximum one-hour concentration. The receptor for the acute exposure scenario is located at the point of maximum one-hour concentration and is independent of land use.

Because the acute exposure scenario is based on the maximum-modeled concentration and assumes that a receptor will be present at the location of that maximum during the hour in which it occurs, this is considered a worst-case scenario. However, it may also represent a plausible location for the receptor.

This acute scenario is designed to evaluate the worst-case air concentration resulting from normal emissions combined with short-term meteorological conditions that result in higher than normal air concentrations. Acute emissions estimates include process upset and cell emissions in addition to normal stack emissions as described in Section 5. The acute scenario is not an accident (e.g., fire, explosion) scenario. Accident scenarios are evaluated in separate documents to support nuclear licensing requirements.

Because the WTP facilities do not handle waste outside the buildings (all waste management activities are conducted inside the negative pressure cells, in sealed transport containers, or doubly contained pipelines) and do not release uncontrolled emissions except through building ventilation systems, there are no "fugitive" emissions. The building cell ventilation systems are equipped only with HEPA filters to control particulate and do not control vapors. Within WTP, the process upset and cell emissions are those vapor emissions that leak from valve, connectors, etc. within the facility that are incidental to waste processing and considered uncontrolled, with the exception of HEPA filtration control of the particulate phase.

### 7.1.4 Exposure Pathways

Exposure pathways to be evaluated for each of these exposure scenarios are summarized in Table 7-1 and the conceptual exposure model (Figure 7-4). Both direct exposure to emissions and indirect exposure to other media (such as soil and food) contaminated by emissions will be evaluated. The following are direct exposure pathways to be included in the quantitative risk assessment:

- COPCs and ROPCs
  - Direct inhalation of emissions
- ROPCs only
  - External exposure to radionuclides in air

Indirect exposure pathways to be included in the quantitative risk assessment are as follows:

- COPCs and ROPCs
  - Ingestion of soil
  - Inhalation of resuspended soil
  - Ingestion of homegrown or wild gathered produce
  - Ingestion of homegrown beef, milk, chicken, eggs, and pork
  - Ingestion of wild game, wildfowl, and wildfowl eggs
  - Ingestion of drinking water
  - Ingestion of fish
  - Inhalation of vapors and suspended particulates in sweat lodge
  - Dermal absorption in the sweat lodge
- ROPCs only

  - External exposure to radionuclides in soil
- PCDDs/PCDFs, PCBs, and selected ROPCs (Sr-90, I-129, Cs-134, and Cs-137) only
  - Ingestion of breast milk

External radiation exposure will be quantitatively evaluated for radionuclides in air and soil. External radiation exposure is not expected to be significant for surface water because of the following:

- Distance from the WTP to the Columbia River will result in extremely low concentrations of ROPCs through deposition.
- ROPC concentrations in air near the WTP and in soil following deposition and accumulation will far exceed surface water concentrations.
- Exposure to air and soil is continuous, while potential exposure to surface water is intermittent.

Therefore, external radiation exposure will not be evaluated for surface water because the distance from the WTP to the Columbia River will result in extremely low concentrations of ROPCs through deposition compared with other media.

The EPA (2005a) has identified the following three exposure pathways, generally considered insignificant contributors to risk at thermal treatment facilities:

- Groundwater pathways
- Resuspended dust
- Dermal contact

Groundwater pathways are generally not significant contributors to risk from airborne emissions because exposure concentrations in groundwater following air dispersion, deposition, leaching, and groundwater dispersion are much less than concentrations in air, soil, and other media. Conditions at the Hanford site (i.e., low precipitation) will make the contribution to groundwater even less than at other sites. Therefore, exposure to groundwater will not be included in the quantitative risk assessment. However, surface water concentrations will be used to evaluate the ingestion of drinking water, as well as inhalation and dermal absorption from exposure in a sweat lodge.

Inhalation of resuspended dust can be an important exposure pathway at contaminated sites where the contaminant source is at the surface or in the soil, as explained in the air-dispersion modeling portion (Section 6) of this work plan. At these sites, dust resuspension generally represents the only source of inhalation exposure (unless significant volatiles are present). At sites such as the WTP where the source of COPCs and ROPCs is airborne emissions, direct, continuous inhalation of these emissions is a much more important exposure pathway than periodic inhalation of fugitive dust. Although it is generally considered insignificant at most sites, inhalation of resuspended dust will be included in the SLRA (CCN 064332) because of the dry, dusty conditions at the Hanford site.

Dermal exposure pathways (to soil, surface water, or air) will not be included in the SLRA, with the exception of exposures from participation in sweat lodge activities. This is a non-conservative assumption (i.e., it will underestimate exposure to contaminants in soil, surface water, and air) because dermal contact will occur. However, dermal exposure pathways have been identified as insignificant contributors to risk in numerous risk assessments prepared or reviewed, or both, by EPA for airborne

emissions from thermal treatment facilities (i.e., the amount that exposure is underestimated due to excluding this pathway is insignificant). If initial PRA results indicate that the soil ingestion pathway results in risks that are borderline for any plausible receptor, then the dermal exposure pathway may be included in the PRA. A discussion of the potential impact associated with exclusion of this minor pathway from the quantitative risk assessment will be included in the uncertainty assessment of the PRA.

### 7.1.5 Quantification of Exposure

The following subsections provide the equations that will be used to quantify intake (or dose) for each COPC and ROPC. The equations used to quantify exposures to COPCs and ROPCs differ slightly. Estimates of COPC intake will be quantified as the lifetime average daily dose (*LADD*) and average daily dose (*ADD*) in units of mg/kg·day. The *LADD* defines a dose level that is distributed (averaged) over an entire lifetime. Unlike the *LADD*, the *ADD* is averaged over a specific incremental exposure period rather than an entire lifetime. Estimates of ROPC intake will be quantified as a total intake in units of picocuries (pCi) over the entire exposure duration. This is explained in greater detail below.

The EPCs for COPCs have units of mass per mass of media (mg/kg for soil, sediment, and food) and mass per volume of media (mg/L for surface water and $\mu g/m^3$ for air). The corresponding daily intake (or dose) units for COPCs are in mass per mass per time (mg/kg·day), that is, the mass concentration of the exposure media per unit time. The EPCs for ROPCs have units of activity per mass of media (pCi/g for soil and food) and activity per volume of media (pCi/L for surface water and $pCi/m^3$ for air). The corresponding intake (or dose) units for ROPCs are activity (pCi), that is, total radioactivity received over the entire exposure period. This is because the dose due to uptake of a COPC is averaged over the exposure period while the dose from ROPC exposure is cumulative. Accordingly, cancer slope factors for COPCs are in units of time per mass concentration (day·kg/mg), reference doses for COPCs are in units of mass concentration per unit time (mg/kg·day), while slope factors for ROPCs are in units of per radioactivity ($pCi^{-1}$).

The intake equations presented in the HHRAP (EPA 2005a) are for use with COPCs and reflect the exposure concentration (*EC*) or daily intake (*I*) for COPCs for an assumed exposure time of 24 hours/day. However, for the exposure scenarios considered in this risk assessment, the exposure time (the amount of time each day that a receptor is exposed), as well as exposure frequency (number of exposure events in days per year), and the exposure duration (the number of years of exposure) vary as a function of receptor lifestyle. The HHRAP does not provide equations for periodic exposures due to varied exposure times, however, RAGS Part F (EPA 2009) states that exposures can be weighted by the fraction of the total exposure time that each period represents. That is, the exposure (*EC*) can be weighted according to the ratio of the exposure time over the averaging time. Accordingly, daily intake of COPCs are corrected for receptor exposure time by multiplying the daily intake (as computed in accordance with the HHRAP) by the receptor exposure time (*ET*) (described in subsequent sections of this work plan) as shown in the equations for intake. In addition, since risk is to be estimated by applying the reference dose or cancer slope factor, the following equations are used to convert daily COPC intake into a *LADD* and *ADD* in units of mg/kg·day. The equations below are based on the equations that appear in HHRAP Tables C-1-7 and C-1-8, and in RAGS Part F Equation 6:

for carcinogenic COPCs and ROPCs:

$$LADD_i = I_i \cdot \frac{ED_i \cdot EF_i}{AT_C \cdot CF} \text{ and } EC_i = C_a \cdot \frac{ET_i \cdot EF_i \cdot ED_i}{AT_C \cdot CF}$$

for non-carcinogenic COPCs:

$$ADD_i = I_i \cdot \frac{ED_i \cdot EF_i}{AT_N \cdot CF} \text{ and } EC_i = C_a \cdot \frac{ET_i \cdot EF_i \cdot ED_i}{AT_N \cdot CF}$$

where:

$ADD_i$ = average daily dose due to exposure pathway *i* for the receptor's total exposure period (mg/kg·day)

$LADD_i$ = lifetime average daily dose due to exposure pathway *i* for the receptor's total exposure period (mg/kg·day or pCi)

$EC_i$ = exposure concentration due to exposure pathway *i* (μg/m$^3$ or pCi/m$^3$)

$I_i$ = daily intake due to exposure pathway *i* (mg/kg·day or pCi)

$C_a$ = contaminant concentration in air (μg/m$^3$ or pCi/m$^3$)

$ED_i$ = exposure duration at the location where the receptor is exposed to pathway *i* (number of years that the receptor is exposed to the COPC or ROPC)

$EF_i$ = exposure frequency at the location where the receptor is exposed to pathway *i* (number of days per year that the receptor is exposed to the COPC or ROPC)

$ET_i$ = exposure time at the location where the receptor is exposed to pathway *i* (number of hours per day that the receptor is exposed to the COPC or ROPC)

$AT_N$ = averaging time for noncarcinogens, typically the same as the receptor's exposure duration (yr)

$AT_C$ = averaging time for carcinogens, typically the lifetime of the receptor (yr)

$CF$ = conversion factor of 365 day/yr for $LADD_i$ and $ADD_i$, and 8760 hr/yr for $EC_i$

Cumulative *ADD*s and *LADD*s for each constituent can be computed by summing the *ADD* and *LADD* for a COPC from each applicable pathway.

The equations that will be used to quantify intake or exposure due to each of the exposure pathways are based on those presented in Appendix C of EPA 2005a. The EPCs of each exposure medium (such as air and soil) will be calculated as described in Sections 6.1.4 and 7.1.7 of this RAWP. Receptor-specific exposure parameters (such as exposure frequency and duration) are summarized in the tables listed below:

| Receptors | Exposure Parameters |
|---|---|
| Hanford Site Industrial Worker | Table 7-2 |
| Resident | Table 7-3 |
| Resident Subsistence Farmer | Table 7-4 |
| Resident Subsistence Fisher | Table 7-5 |
| Resident Subsistence American Indians | Table 7-6, Table 7-7 and Table 7-8 |

The equations provided in the following subsections, along with the source of the EPCs and exposure parameters that will be used in these equations, are summarized below:

| Exposure Medium and Pathway | Equation | Source of Exposure Point Concentrations | Applicable Receptor |
|---|---|---|---|
| **Air (Section 7.1.5.1)** | | | |
| Inhalation of emissions | modified HHRAP Table C-2-1 | Section 6.1 | All |
| External exposure to ROPCs in air | modified Eq. 5 from EPA 2000 | Section 6.1 | All |
| **Soil (Section 7.1.5.2)** | | | |
| Incidental ingestion | HHRAP Table C-1-1 and modified Eq. 1 from EPA 2000 | Section 6.2 | All |
| Inhalation of resuspended dust | modified HHRAP Table C-2-1 and modified Eq. 3 from EPA 2000 | Section 6.2 | All |
| External exposure to ROPCs in soil | modified Eq. 5 from EPA 2000 | Section 6.2 | All |
| **Foodstuffs (Section 7.1.5.3)** | | | |
| Ingestion of domestic and wild produce | HHRAP Table C-1-2 | Section 6.5 | All |
| Ingestion of domestic livestock and/or wild game | HHRAP Table C-1-3 | Section 7.1.7.4 | Farmer<br>American Indian |
| Ingestion of fish | HHRAP Table C-1-4 | Section 7.1.7.5 | Fisher<br>American Indian |
| **Surface Water (Section 7.1.5.4)** | | | |
| Drinking water ingestion | HHRAP Table C-1-5 | Section 6.3 | All |
| **Sweat Lodge (Section 7.1.5.5)** | | | |
| Inhalation in sweat lodge | modified Eqs. 7 and 15 in Appendix 4 of Harris and Harper (2004) | Section 6.3 | American Indian |
| Dermal exposure in sweat lodge | modified Eq. 18 in Appendix 4 of Harris and Harper (2004) | Section 6.3 | American Indian |

### 7.1.5.1 Direct Exposure to Air

Direct exposure to air includes inhalation of vapor and particulate emissions and external exposure to ionizing radiation in air.

**Direct Inhalation**

A modified version of the equation in Table C-2-1 of the HHRAP will be used to calculate the inhalation of vapor phase and particulate emissions. The equation is modified by the introduction of the exposure time in accordance with RAGS Part F guidance (Equation 6) for periodic or microenvironment exposures (the HHRAP equation for exposure concentration from direct inhalation assumes a 24 hr/day exposure time).

$$EC_{inh} = \frac{C_a \cdot ET \cdot EF \cdot ED}{AT \cdot CF} \qquad \text{(modified HHRAP Table C-2-1)}$$

where:

$EC_{inh}$ = exposure concentration of COPCs or ROPCs through inhalation of emissions (mg/m$^3$ or pCi/m$^3$)

$C_a$ = concentration of COPC or ROPC in air (μg/m$^3$ or pCi/m$^3$) calculated as described in Section 6.1

$ET$ = exposure time (hr/day)

$EF$ = exposure frequency (day/yr)

$ED$ = exposure duration (yr)

$AT$ = averaging time for carcinogens ($AT_C$) or noncarcinogens ($AT_N$) (yr)

$CF$ = units conversion factor of $365\frac{\text{day}}{\text{yr}} \cdot 24\frac{\text{hr}}{\text{day}}$

For ROPCs, inhalation slope factors take into account the age- and gender-dependence of radionuclide intake, metabolism, dosimetry, radiogenic risk, and competing causes of death in estimating the cancer risk from low-level exposures to radionuclides in the environment (EPA 2001). Accordingly, the exposure concentration from ROPCs is converted to intake based on the inhalation rate and averaging time of the exposure. For ROPCs the total exposure from inhalation of air is:

$$\text{ROPCs: } I_{inh} = EC_{inh} \cdot IR \cdot AT \cdot CF \qquad \text{(modified HHRAP Table C-3-1)}$$

where:

$I_{inh}$ = intake of ROPC via inhalation (pCi)

$EC_{inh}$ = exposure concentration of ROPCs through inhalation of emissions (pCi/m$^3$)

$IR$ = inhalation rate (m$^3$/hr)

$AT$ = averaging time for carcinogens ($AT_C$) (yr)

$CF$ = units conversion factor of 8760 (hr/yr)

**External Exposure in Air**

A modified version of Equation 5 from *Soil Screening Guidance for Radionuclides: Technical Background Document* (EPA 2000) will be used to calculate the external exposure to ionizing radiation in

air from ROPCs. The equation has been modified by substituting air concentration for soil screening level and dividing by the slope factor to derive a dose.

ROPCs: $I_{ira} = C_a \cdot EF \cdot ED \cdot ACF \cdot \left[ET_o + \left(ET_i \cdot S_e\right)\right] \cdot CF$ (modified Eq. 5 from EPA 2000)

where:

| | | |
|---|---|---|
| $I_{ira}$ | = | external exposure to radiation from ROPCs in air (Bq-sec/m$^3$) |
| $C_a$ | = | average air concentration of ROPC (pCi/m$^3$) calculated as described in Section 6.1 |
| $EF$ | = | exposure frequency (day/yr) |
| $ED$ | = | exposure duration (yr) |
| $ACF$ | = | area correction factor for an infinite slab, 1.0 (unitless) |
| $ET_o$ | = | exposure time fraction outdoors (unitless); receptor-specific $ET_o$ values are described below |
| $ET_i$ | = | exposure time fraction indoors (unitless); receptor-specific $ET_i$ values are described below |
| $S_e$ | = | shielding factor (unitless); $S_e$ is described below |
| $CF$ | = | units conversion factor of $0.37 \frac{\text{Bq}}{\text{pCi}} \cdot 86400 \frac{\text{sec}}{\text{day}}$ |

The exposure time fraction outdoors ($ET_o$) represents the fraction of the day that the receptor is at the exposure location and outdoors while the fraction indoors ($ET_i$) represents the fraction of the day that the receptor is at the exposure location and indoors.

For the resident scenario, it is assumed that adults spend 94 % of their time indoors and 6 % outdoors (EPA 1997a) while children spend 77 % of their time indoors and 23 % outdoors. The median percent of time spent outdoors on a farm (adults and children) is reported as 12 %, and the 90th percentile is reported as 42 % (EPA 1997a). For the resident subsistence farmer and subsistence fisher scenarios, receptors (both adults and children) are assumed to spend 42 % of their time outdoors and 58 % indoors (approximately an additional 8 hours outdoors each day). For the resident subsistence American Indian, the time spent outdoors is based on values provided in the TC&WM EIS (i.e., 66 % indoors, 12 % outdoors for both adults and children). The resident subsistence American Indian also spends 2 hours/day (8 %) in a sweat lodge and another 14 % of time at an undisclosed location. External air exposure is not assessed at these locations because they are presumed to be locations where ionizing radiation is not an issue. Adults and children in the alternate tribal scenarios are assumed to spend 50 % of their time outdoors and 50 % indoors (alternate scenario #1, based on Harper 1997), 29 % of their time outdoors and 71 % of their time indoors (alternate scenario #2, based on RUDOLFI Inc. 2007).

For the Hanford site industrial worker scenario, it is assumed that work is performed both outdoors and indoors; therefore, workers spend 50 % of their work day indoors and 50 % outdoors. Outdoor occupancy patterns of the worker after work are assumed to be the same number of hours as the resident (6 % of the 24-hour day outdoors - that is, 1.44 hours/day), leaving the remaining hours of the day for indoor activities (equating to 61 % of the 24-hour day). During the weekend and other non-work days, the worker's outdoor/indoor occupancy fraction is assumed to be the same as the resident's.

A shielding factor of 0.4 is used (EPA 2000) to account for shielding while the receptor is indoors. No shielding is assumed while the receptor is outdoors, as the gamma radiation originating in soil is not impeded by a solid obstacle prior to intercepting the receptor.

#### 7.1.5.2 Exposure to Soil

Exposure to soil includes ingestion of soil, inhalation of resuspended soil, and external exposure to ROPCs in soil.

**Ingestion of Soil**

Table C-1-1 of the HHRAP will be used to calculate the average soil concentration over exposure duration for COPCs. The equation is modified with the use of the exposure time parameter. Use of this parameter is necessary since the exposure time of receptors varies for the scenarios considered in the risk assessment. A modified version of Equation 1 from *Soil Screening Guidance for Radionuclides: Technical Background Document* (EPA 2000) will be used to calculate the ingestion of soil for ROPCs. The ROPC equation from EPA (2000) has been modified by substituting soil concentration for soil screening level and dividing by the slope factor to derive an intake.

COPCs: $$I_{soil} = \frac{Cs \cdot ET \cdot CR_{soil} \cdot F_i}{BW \cdot CF_1}$$ (HHRAP Table C-1-1)

ROPCs: $$I_{soil} = Cs \cdot CR_{soil} \cdot F_i \cdot ET \cdot EF \cdot ED \cdot CF_2$$ (modified Eq. 1 from EPA 2000)

where:

$I_{soil}$ = intake of COPC or ROPC due to soil ingestion (mg/kg·day or pCi)
$Cs$ = concentration of COPC or ROPC in soil (mg/kg or pCi/g) calculated per Section 6.2
$CR_{soil}$ = consumption rate of soil (kg/day)
$F_i$ = fraction of ingested soil that is contaminated (unitless)
$ET$ = exposure time (hr/day)
$EF$ = exposure frequency (day/yr)
$ED$ = exposure duration (yr)
$BW$ = body weight (kg)
$CF_1$ = units conversion factor of $24\frac{\text{hr}}{\text{day}}$
$CF_2$ = units conversion factor of 1000 (g/kg)

**Inhalation of Resuspended Soil**

A modified version of Equation 3 from *Soil Screening Guidance for Radionuclides: Technical Background Document* (EPA 2000) will be used to calculate exposure resulting from inhalation of resuspended soil using the particulate emission factor (*PEF*) approach from the EPA soil screening guidance (EPA 2000). The equation has been modified by substituting soil concentration for soil screening level and dividing by the slope factor to derive a dose. To derive the intake of COPCs and ROPCs from soil inhalation, the equation for exposure concentration (HHRAP Table C-2-1), is modified

by substituting the soil concentration divided by the *PEF* for the air concentration. The equation is further modified by the introduction of the exposure time in accordance with RAGS Part F guidance for periodic or microenvironment exposures. Terms for these exposure parameters are applied using the equations for hazard index (HHRAP Table C-1-7) and cancer risk (HHRAP Table C-1-8).

COPCs: $$EC_{soil} = \frac{\left(\frac{Cs}{PEF}\right) \cdot ET \cdot EF \cdot ED}{AT \cdot CF_1}$$ (modified HHRAP Table C-2-1)

ROPCs: $$EC_{soil} = \frac{\left(\frac{Cs}{PEF}\right) \cdot ET \cdot EF \cdot ED \cdot CF_2}{AT \cdot CF_1}$$ (modified HHRAP Table C-2-1)

where:

| | | |
|---|---|---|
| $EC_{soil}$ | = | intake of COPC or ROPC through inhalation of resuspended soil (mg/m$^3$ or pCi/m$^3$) |
| $Cs$ | = | soil concentration of COPC or ROPC (mg/kg or pCi/g) calculated per Section 6.2 |
| $PEF$ | = | particulate emission factor (m$^3$/kg); *PEF* is described below |
| $IR$ | = | inhalation rate (m$^3$/hr) |
| $ET$ | = | exposure time (hr/day) |
| $EF$ | = | exposure frequency (day/yr) |
| $ED$ | = | exposure duration (yr) |
| $BW$ | = | body weight (kg) |
| $CF_1$ | = | units conversion factor of $24\frac{\text{hr}}{\text{day}}$ |
| $CF_2$ | = | units conversion factor of 1000 (g/kg) |

The *PEF* relates the concentration of contaminant in soil with the concentration of dust particles in the air. The presence of vegetation, gravel, pavement, or other cover will prevent the generation of fugitive dust. EPA default *PEF* values assume 50 % vegetative cover and 50 % open soil. The EPA provides site-specific dispersion modeling and meteorological factors for 29 cities in the United States and recommends developing a site-specific *PEF* by identifying the climatic zone for the site (Figure A-1, EPA 2000) followed by selecting modeling parameters corresponding to the site's climatic zone and size. The Hanford site is located in Climatic Zone 4, so a value of 40.4 is used to describe the inverse mean concentration at center of a 30-acre-square source (average value of climatic zone 4 cities). An average wind speed of 3.23 m/s (average value from Hanford Meteorological Station measurements) and a particle size mode of 262 μm have been chosen to represent site conditions (refer to Appendix A). Using methodologies found in Streile et al. (1996) and Cowherd et al. (1985), the *PEF* is conservatively estimated at $7.06 \times 10^7$ m$^3$/kg. Refer to the discussion in Appendix A for additional details.

The intake of radionuclides due to inhalation of resuspended soil can be computed using the same conversion equations used to calculate the intake from air inhalation as previously discussed.

ROPCs: $$I_{inhsoil} = EC_{soil} \cdot IR \cdot AT \cdot CF$$ (modified HHRAP Table C-3-1)

where:

$I_{inhsoil}$ = intake of constituent via inhalation (pCi)
$EC_{soil}$ = exposure concentration of COPCs or ROPCs through inhalation of emissions (pCi/m$^3$)
$IR$ = inhalation rate (m$^3$/hr)
$AT$ = averaging time for carcinogens ($AT_C$) (yr)
$CF$ = units conversion factor of 8760 (hr/yr)

**External Exposure to Soil**

A modified version of Equation 5 from *Soil Screening Guidance for Radionuclides: Technical Background Document* (EPA 2000) will be used to calculate the external exposure to ionizing radiation in soil from ROPCs. The equation has been modified by substituting soil concentration for soil screening level and dividing by the slope factor to derive a dose.

$$\text{ROPCs: } I_{irs} = \frac{Cs \cdot EF \cdot ED \cdot ACF \cdot \left[ET_o + ET_i \cdot (1 - S_e)\right]}{CF} \qquad \text{(modified Eq. 5 from EPA 2000)}$$

where:

$I_{irs}$ = external exposure to radiation from ROPCs in soil (pCi-yr/g)
$Cs$ = soil concentration of ROPC (pCi/g) calculated per Section 6.2
$ACF$ = area screening factor, 1.00
$EF$ = exposure frequency (day/yr)
$ED$ = exposure duration (yr)
$ET_o$ = exposure time fraction outdoors (unitless); receptor-specific $ET_o$ values are described below
$ET_i$ = exposure time fraction indoors (unitless); receptor-specific $ET_i$ values are described below
$S_e$ = shielding factor (unitless); $S_e$ is described below
$CF$ = units conversion factor 365 (day/yr)

The exposure time fraction outdoors ($ET_o$) represents the fraction of the day that the receptor is at the exposure location and outdoors, while the fraction indoors ($ET_i$) represents the fraction of the day that the receptor is at the exposure location and indoors.

For the resident scenario, it is assumed that adults spend 94 % of their time indoors and 6 % outdoors (EPA 1997a) while children spend 77 % of their time indoors and 23 % outdoors. The median percent of time spent outdoors on a farm (adults and children) is reported as 12 %, and the 90th percentile is reported as 42 % (EPA 1997a). For the resident subsistence farmer and subsistence fisher scenarios, receptors (both adults and children) are assumed to spend 42 % of their time outdoors and 58 % indoors (approximately an additional 8 hours outdoors each day). For the resident subsistence American Indian, the time spent outdoors is based on values provided in the TC&WM EIS (i.e., 66 % indoors, 12 % outdoors for both adults and children). The resident subsistence American Indian also spends 2 hours/day (8 %) in a sweat lodge and another 14 % of time at an undisclosed location. External soil exposure is not assessed at these locations because they are presumed to be locations where soil contamination is not an

issue. Adults and children in the alternate American Indian scenarios are assumed to spend 50 % of their time outdoors (alternate scenario #1, based on Harper 1997) and 29 % of their time outdoors (alternate scenario #2, based on RUDOLFI Inc. 2007).

For the Hanford site industrial worker scenario, it is assumed that work is performed both outdoors and indoors; therefore, workers spend 50 % of their work day indoors and 50 % outdoors. Outdoor occupancy patterns of the worker after work are assumed to be the same as the resident (6 % of the 24-hour day outdoors - that is, 1.44 hours/day), leaving the remaining hours of the day for indoor activities (equating to 61 % of the 24-hour day). During the weekend and other non-work days, the worker's outdoor/indoor occupancy fraction is assumed to be the same as the resident's.

A shielding factor of 0.4 is used (EPA 2000) to account for shielding while the receptor is indoors. No shielding is assumed while the receptor is outdoors, as the gamma radiation originating in soil is not impeded by a solid obstacle prior to intercepting the receptor.

#### 7.1.5.3 Exposure to Foodstuffs

Exposure to foodstuffs includes ingestion of domestic produce by the resident; ingestion of domestic produce, beef, pork, milk, chicken, and eggs by the resident subsistence farmer; ingestion of domestic produce and fish by the resident subsistence fisher; and ingestion of wild plants, wild game, wildfowl, wildfowl eggs, and fish by the resident subsistence American Indian and the alternate resident subsistence American Indian #1. The alternate tribal resident #2 consumes domestic foods and supplements his/her diet with wild foods.

**Ingestion of Produce**

Table C-1-2 of the HHRAP will be used to calculate the ingestion of COPCs in domestic produce and wild plants. A modified version of the equation in Table C-1-1 of the HHRAP will be used to calculate the ingestion of ROPCs in domestic produce and wild plants. The equation is modified by multiplying by the receptor body weight since the consumption rate is in terms of unit body weight.

COPCs: (HHRAP Table C-1-2)

$$I_{ag} = \left[ (Pd + Pv + Pr_{ag}) \cdot CR_{ag} + Pr_{ag} \cdot CR_{pp} + Pr_{bg} \cdot CR_{bg} \right] \cdot F_{ag}$$

ROPCs: (modified HHRAP Table C-1-2)

$$I_{ag} = \left[ (Pd + Pv + Pr_{ag}) \cdot CR_{ag} + Pr_{ag} \cdot CR_{pp} + Pr_{bg} \cdot CR_{bg} \right] \cdot F_{ag} \cdot EF \cdot ED \cdot BW \cdot CF$$

for carbon-14 and tritium:

$$I_{ag} = \left[ C_V \cdot \left( CR_{ag} + CR_{pp} + CR_{bg} \right) \right] \cdot F_{ag} \cdot EF \cdot ED \cdot BW \cdot CF$$

where:

$I_{ag}$ = intake of COPC or ROPC through ingestion of produce (mg/kg·day or pCi)

$Pd$ = COPC or ROPC concentration in aboveground produce due to direct deposition onto plant surfaces (mg/kg or pCi/g) calculated per Section 6.6. *Pd* is zero in future exposure scenarios since they occur after cessation of emissions.

$Pv$ = COPC or ROPC concentration in aboveground produce due to air-to-plant transfer (mg/kg or pCi/g) calculated per Section 6.6. $Pv$ is zero in future exposure scenarios since they occur after cessation of emissions.

$Pr_{ag}$ = COPC or ROPC concentration in aboveground produce due to root uptake (mg/kg or pCi/g) calculated per Section 6.6

$Pr_{bg}$ = COPC or ROPC concentration in belowground produce due to root uptake (mg/kg or pCi/g) calculated per Section 6.6

$C_V$ = plant concentration (carbon-14, $C_{V(C\text{-}14)}$, and tritium, $C_{V(H\text{-}3)}$) as discussed in Section 6.6.2 (pCi/g)

$CR_{ag}$ = consumption rate of aboveground unprotected produce (kg/kg·day)

$CR_{pp}$ = consumption rate of aboveground protected produce (kg/kg·day)

$CR_{bg}$ = consumption rate of belowground produce (kg/kg·day)

$F_{ag}$ = fraction of ingested produce that is contaminated (unitless)

$EF$ = exposure frequency (day/yr)

$ED$ = exposure duration (yr)

$BW$ = body weight (kg)

$CF$ = units conversion factor of 1000 (g/kg)

Consumption rates for produce (kg/kg·day) are found in Table 7-2, Table 7-3, Table 7-4, Table 7-5, Table 7-6, Table 7-7, and Table 7-8.

**Ingestion of Animal Products**

Table C-1-3 of the HHRAP will be used to calculate the ingestion of COPCs and ROPCs in homegrown beef, milk, pork, poultry, wildfowl, eggs, and wild game.

COPCs: $I_i = A_i \cdot CR_i \cdot F_i$ (HHRAP Table C-1-3)

ROPCs: $I_i = A_i \cdot CR_i \cdot F_i \cdot EF \cdot ED \cdot BW \cdot CF$ (modified HHRAP Table C-1-3)

where:

$I_i$ = intake of COPC or ROPC from animal product $i$ (such as $I_{beef}$, $I_{milk}$, $I_{pork}$, $I_{poultry}$, $I_{eggs}$ for the subsistence farmer, and $I_{wild\ fowl}$, $I_{wild\ eggs}$, $I_{game}$, $I_{game\ organs}$ for the American Indian scenarios) (mg/kg·day or pCi)

$A_i$ = concentration of COPC or ROPC in animal product $i$ (mg/kg or pCi/g) calculated per Section 7.1.7.4

$CR_i$ = consumption rate of animal product $i$ (kg/kg·day); see Table 7-4, Table 7-6, Table 7-7, and Table 7-8 for values.

$F_i$ = fraction of ingested animal tissue that is contaminated (unitless)

$EF$ = exposure frequency (day/yr)

$ED$ = exposure duration (yr)

$BW$ = body weight (kg)

$CF$ = units conversion factor of 1000 (g/kg)

**Ingestion of Fish**

A modified version of the equation in Table C-1-4 of the HHRAP will be used to calculate the ingestion of COPCs and ROPCs in fish. The modification is needed to account for consumption of fish organs by American Indian receptors.

COPCs: $I_{fish} = \left(C_{fish} \cdot CR_{fish} + C_{organs} \cdot CR_{organs}\right) \cdot F_{fish}$ (HHRAP Table C-1-4)

ROPCs: $I_{fish} = \left(C_{fish} \cdot CR_{fish} + C_{organs} \cdot CR_{organs}\right) \cdot F_{fish} \cdot EF \cdot ED \cdot BW \cdot CF$ (modified HHRAP Table C-1-4)

where:

| | | |
|---|---|---|
| $I_{fish}$ | = | intake of COPC or ROPC from fish (mg/kg·day or pCi) |
| $C_{fish}$ | = | concentration of COPC or ROPC in fish (mg/kg or pCi/g); $C_{fish}$ will be calculated from surface water or sediment concentrations as applicable, calculated per Sections 6.3 and 6.4 |
| $CR_{fish}$ | = | consumption rate of fish fillets (kg/kg-body weight/day) |
| $C_{organs}$ | = | concentration of COPC or ROPC in fish organs (mg/kg or pCi/g); $C_{organs}$ will be calculated from surface water and sediment concentrations calculated per Sections 6.3 and 6.4 |
| $CR_{organs}$ | = | consumption rate of fish organs (kg/kg-body weight/day) |
| $F_{fish}$ | = | fraction of ingested fish that is contaminated (unitless) |
| $EF$ | = | exposure frequency (day/yr) |
| $ED$ | = | exposure duration (yr) |
| $BW$ | = | body weight (kg) |
| $CF$ | = | units conversion factor of 1000 (g/kg) |

Fish consumption rates are found in Table 7-5, Table 7-6, Table 7-7, and Table 7-8.

### 7.1.5.4 Exposure to Surface Water

Exposure to surface water includes the ingestion of surface water as drinking water and sweat lodge exposures through inhalation and dermal contact (Section 7.1.5.5).

**Ingestion of Drinking Water**

Table C-1-5 of the HHRAP will be used to calculate the ingestion of COPCs and ROPCs in drinking water.

COPCs: $I_{dw} = \dfrac{C_{dw} \cdot CR_{dw} \cdot F_{dw}}{BW}$ (HHRAP Table C-1-5)

ROPCs: $I_{dw} = C_{dw} \cdot CR_{dw} \cdot F_{dw} \cdot EF \cdot ED$ (modified HHRAP Table C-1-5)

where:

| | | |
|---|---|---|
| $I_{dw}$ | = | intake of COPC or ROPC from drinking water (mg/kg·day or pCi) |
| $C_{dw}$ | = | dissolved-phase COPC or ROPC water concentration (mg/L or pCi/L) calculated per Section 6.3 |
| $CR_{dw}$ | = | consumption rate of drinking water (L/day) |
| $F_i$ | = | fraction of ingested drinking water that is contaminated (unitless) |
| $EF$ | = | exposure frequency (day/yr) |
| $ED$ | = | exposure duration (yr) |
| $BW$ | = | body weight (kg) |

Consumption rates for drinking water are found in Table 7-2, Table 7-3, Table 7-4, Table 7-5, Table 7-6, Table 7-7, and Table 7-8.

#### 7.1.5.5 Sweat Lodge Exposures

Two exposure pathways will be evaluated for the sweat lodge: inhalation and dermal absorption.

**Inhalation in Sweat Lodge**

A modified version of Equation 7 in Appendix 4 of Harris and Harper (2004) will be used to calculate inhalation exposure for receptors inside the sweat lodge. The equation was modified to reflect the exposure concentration (*EC*) to be consistent with the HHRAP (the original equation was for intake, *I*). Volatile and semivolatile organic COPCs and volatile ROPCs ($^{14}C$, $^{3}H$, and $^{129}I$) may be released as vapors from water used in the sweat lodge.

$$EC_{sl} = \frac{C_{dw} \cdot \left(\frac{V_w}{2}\right) \cdot \left(\frac{1}{\frac{2}{3} \cdot \pi \cdot r^3}\right) \cdot ET \cdot EF \cdot ED}{AT \cdot CF}$$

(modified Equation 7 in Appendix 4 of Harris and Harper [2004])

Due to the many uncertainties and the potential that aerosols may be generated by mechanical entrainment in addition to volatilization, nonvolatile inorganic COPCs and ROPCs are also evaluated for this pathway. In the sweat lodge environment, nonvolatile components become airborne as an aerosol as the water they were carried in vaporizes. Once airborne, nonvolatile compounds deposit onto solid surfaces with aqueous condensation, thus, the amount of contaminants available for inhalation is limited to that which is carried into the air phase by the volume of liquid water needed to create saturated conditions in the lodge. Harris and Harper (2004) present an equation for the intake of nonvolatile constituents utilizing the ideal gas law and the Antoine equation for the vapor pressure of water (Appendix 4, Equation 15). Modifying this equation for the exposure concentration yields the following:

$$EC_{sl} = \left(\frac{ET \cdot EF \cdot ED}{AT \cdot CF}\right) \cdot C_{dw} \cdot \left(\frac{MW_w}{R \cdot T_{sl} \cdot \rho_w}\right) \cdot e^{\left(18.3036 - \frac{3816.44}{T_{sl} - 46.13}\right)}$$

(modified Equation 15 in Appendix 4 of Harris and Harper [2004])

where:

$EC_{sl}$ = exposure concentration of COPCs or ROPCs through inhalation in the sweat lodge ($mg/m^3$ or $pCi/m^3$)

$C_{dw}$ = dissolved surface water concentration of COPCs and ROPCs (mg/L or pCi/L) calculated per Section 6.3

$V_W$ = volume of water (4 L); see the discussion of $V_W$ below

$\pi$ = the constant *pi* (unitless); $\pi \approx 3.14159265359$

$r$ = radius of sweat lodge (1 m)

$MW_w$ = molar weight of water (18.01528 g/mol)

$R$ = ideal gas constant (0.06237 $mmHg \cdot m^3/gmole \cdot K$)

$T_{sl}$ = temperature of the sweat lodge (339 K)

$\rho_w$ = density of liquid water at temperature $T_{sl}$ (980.2 g/L)

$ET$ = exposure time (hr/day)

$EF$ = exposure frequency (day/yr)

$ED$ = exposure duration (yr)

$AT$ = averaging time for carcinogens ($AT_C$) or noncarcinogens ($AT_N$) (yr)

$e$ = base of the natural logarithm (unitless). $e = \sum_{i=0}^{\infty} \frac{1}{i!} \approx 2.718282$; the units for the Antoine equation, $e^{\left(18.3036 - \frac{3816.44}{T_{sl} - 46.13}\right)}$, are mmHg

$CF$ = units conversion factor of 365 (day/yr)

Within the sweat lodge, water is splashed onto heated rocks to produce steam. It is assumed that a total of 4 L of water are used during a one-hour sweat lodge ceremony. For the HHRA, it is conservatively assumed that the entire concentration of volatile COPCs (all organics) and ROPCs ($^3H$, $^{14}C$, and $^{129}I$) in the 4 L of water may be volatilized and available for inhalation in the sweat lodge. It is possible that nonvolatile COPCs (inorganics) and ROPCs (all except $^3H$, $^{14}C$, and $^{129}I$) may become airborne as an aerosol mist. The quantity of nonvolatile constituents that may be airborne is limited by the amount of water that may be in the air at any given time[3] (CCN 064329).

Note that the daily intake of radionuclides from inhalation in the sweat lodge is:

ROPCs: $I_{inh_{sl}} = EC_{sl} \cdot IR \cdot AT \cdot CF$ (modified HHRAP Table C-3-1)

where:

$I_{inh_{sl}}$ = intake of constituent via inhalation in the sweat lodge (pCi)

$EC_{sl}$ = exposure concentration of ROPCs through inhalation in the sweat lodge ($pCi/m^3$)

[3] For nonvolatile constituents, the volume of liquid water needed to create a saturated vapor in the sweat lodge in units of liters (L) is $V_w = \left(\frac{p \cdot V_{sl}}{R \cdot T}\right)\left(\frac{MW_w}{\rho_w}\right)$ where $V_{sl}$ is volume of air space in sweat lodge occupied by water vapor (2.094 $m^3$), and other variables are as defined above (refer to Harris and Harper [2004]).

*IR* = inhalation rate ($m^3$/hr)
*AT* = averaging time for carcinogens ($AT_C$) (yr)
*CF* = units conversion factor of 8760 (hr/yr)

**Dermal Exposure in Sweat Lodge**

Equation 18 in Harris and Harper (2004) will be used to calculate the dermal absorption of volatile and semivolatile compounds (e.g. $F_v \neq 0$) from water vapor in the sweat lodge.

$$\text{COPCs: } I_d = \frac{C_{dw} \cdot \left(\frac{V_w}{2}\right) \cdot \left(\frac{1}{\frac{2}{3} \cdot \pi \cdot r^3}\right) \cdot SA \cdot Kp \cdot ET}{BW} \cdot CF$$

(Equation 18 in Appendix 4 of Harris and Harper [2004])

where:

$I_d$ = intake of COPCs from adult dermal absorption within the sweat lodge (mg/kg·day)
$C_{dw}$ = dissolved surface water concentration of COPCs (mg/L) calculated per Section 6.3
$V_W$ = volume of water (4 L); see the discussion of $V_W$ above
$\pi$ = the constant *pi* (unitless); $\pi \approx 3.14159265359$
*r* = radius of sweat lodge (1 m)
*SA* = body surface area available for contact ($m^2$)
*Kp* = permeability constant (cm/hr); *Kp* is COPC-specific and provided in Supplement 4.
*ET* = exposure time (hr/day)
*EF* = exposure frequency (day/yr)
*ED* = exposure duration (yr)
*BW* = body weight (kg)
*AT* = averaging time for carcinogens ($AT_C$) or noncarcinogens ($AT_N$) (yr)
*CF* = units conversion factor of $10^{-2}$ (m/cm)

Equations 19, 20, and 21 in Harris and Harper (2004) will be used to calculate the dermal absorption of nonvolatile compounds (e.g., $F_v = 0$) from water vapor in the sweat lodge. The intake includes the contribution from condensed (liquid) water ($I_{d,l}$) and vapor ($I_{d,v}$).

$$\text{COPCs: } I_{d,l} = \frac{C_{dw} \cdot SA \cdot Kp \cdot ET}{BW} \cdot CF$$

(Equations 19 and 20 in Appendix 4 of Harris and Harper [2004])

$$I_{d,v} = \left(\frac{SA \cdot Kp \cdot ET}{BW}\right) \cdot C_{dw} \cdot \left(\frac{MW_w}{R \cdot T_{sl} \cdot \rho_w}\right) \cdot e^{\left(18.3036 - \frac{3816.44}{T_{sl} - 46.13}\right)}$$

where:

$I_{d,l}$ = intake of COPCs from adult dermal absorption of condensate within the sweat lodge (mg/kg·day)

$I_{d,v}$ = intake of COPCs from adult dermal absorption of vapors within the sweat lodge (mg/kg·day)

$I_d$ = intake of COPCs from adult dermal absorption of condensate and vapors within the sweat lodge (mg/kg·day)

$C_{dw}$ = dissolved surface water concentration of COPCs (mg/L) calculated per Section 6.3

$V_W$ = volume of water (4 L); see the discussion of $V_W$ above

$\pi$ = the constant *pi* (unitless); $\pi \approx 3.14159265359$

$r$ = radius of sweat lodge (1 m)

$SA$ = body surface area available for contact ($m^2$)

$Kp$ = permeability constant (cm/hr); *Kp* is COPC-specific and provided in Supplement 4.

$ET$ = exposure time (hr/day)

$EF$ = exposure frequency (day/yr)

$ED$ = exposure duration (yr)

$BW$ = body weight (kg)

$AT$ = averaging time for carcinogens ($AT_C$) or noncarcinogens ($AT_N$) (yr)

$MW_w$ = molar weight of water (18.01528 g/mol)

$R$ = ideal gas constant (0.06237 mmHg·$m^3$/gmole·K)

$T_{sl}$ = temperature of the sweat lodge (389 K)

$\rho_w$ = density of liquid water at temperature $T_{sl}$ (980.2 g/L)

$CF$ = units conversion factor of $10\frac{\mathrm{L}}{\mathrm{m}^3 \cdot \mathrm{cm}}$

$e$ = base of the natural logarithm (unitless). $e = \sum_{i=0}^{\infty} \frac{1}{i!} \approx 2.718282$; the units for the Antoine equation, $e^{\left(18.3036 - \frac{3816.44}{T_{sl} - 46.13}\right)}$, are mmHg

and subsequently:

$$I_d = I_{d,l} + I_{d,v}$$

where:

$I_d$ = intake of COPCs from adult dermal absorption within the sweat lodge (mg/kg·day)

$I_{d,l}$ = intake of COPCs from adult dermal absorption of condensate within the sweat lodge (mg/kg·day)

$I_{d,v}$ = intake of COPCs from adult dermal absorption of vapors within the sweat lodge (mg/kg·day)

Dermal absorption of inorganic COPCs and ROPCs is not included because this pathway is considered to be insignificant compared to inhalation for all inorganic COPCs and ROPCs except tritium (CCN 019247). Previously, the inhalation cancer slope factor (*CSF*) provided in the Health Effects Assessment Summary Tables (HEAST) [EPA 1997b] for tritium included a 50 % contribution from

dermal absorption. The new inhalation *CSF* for tritium provided in the updated the HEAST (EPA 2001) does not include the contribution from dermal absorption; therefore, dermal absorption of tritium from water vapor in the sweat lodge is evaluated separately. The internal dose from immersion in a plume of tritiated water vapor is approximately 50 % from inhalation and 50 % from dermal absorption (Till and Meyer 1983); therefore, the dose received from dermal absorption of tritium is accounted for by multiplying the inhalation dose for this ROPC by two.

#### 7.1.5.6 Nursing Infant Exposure

**Ingestion of Breast Milk**

Table C-3-2 of the HHRAP will be used to calculate the *ADD* of COPCs and intake of ROPCs for an infant exposed to COPCs and ROPCs in breast milk. Infant exposure from breast milk is estimated assuming that the mother's breast milk has reached a steady state and that the contaminant in breast milk fat is the same as that in maternal body fat.

COPCs: $$ADD_{infant} = \frac{C_{milkfat} \cdot f_3 \cdot f_4 \cdot IR_{infant} \cdot ED_{infant}}{BW_{infant} \cdot AT_{infant}}$$ (HHRAP Table C-3-2)

ROPCs: $$ADD_{infant} = C_{milkfat} \cdot f_3 \cdot f_4 \cdot IR_{infant} \cdot ED_{infant} \cdot EF_{infant}$$ (modified HHRAP Table C-3-2)

where:

$ADD_{infant}$ = average daily dose of COPCs or ROPCs from breast milk (pg/kg·day for chemicals or pCi for radionuclides)

$C_{milkfat}$ = concentration of COPC or ROPC in milk fat of breast milk for a specific exposure scenario of the mother, described below (pg/kg·day or pCi)

$f_3$ = fraction of breast milk that is fat (unitless)

$f_4$ = fraction of ingested COPC or ROPC that is absorbed (unitless)

$IR_{infant}$ = infant ingestion rate of breast milk by infant (kg/day)

$ED_{infant}$ = infant exposure duration (yr)

$BW_{infant}$ = infant body weight of infant (kg)

$AT_{infant}$ = infant averaging time (yr)

$EF_{infant}$ = infant exposure frequency (days/yr)

The concentration in milk fat is estimated using Table C-3-1 of the HHRAP.

COPCs: $$C_{milkfat} = \frac{m \cdot h \cdot f_1}{0.693 \cdot f_2} \cdot CF$$ (HHRAP Table C-3-1)

ROPCs: $$C_{milkfat} = \frac{m \cdot h \cdot f_1}{BW_{mother} \cdot 0.693 \cdot f_2}$$ (modified HHRAP Table C-3-1)

where:

$m$ = maternal intake of COPCs or ROPCs from all adult exposures (mg/kg·day for chemicals or pCi/day for radionuclides) calculated as:

Hanford site industrial worker[4] and resident:

$$m = I_{inh} + I_{soil} + I_{inhsoil} + I_{ag} + I_{dw}$$

resident subsistence farmer:

$$m = I_{inh} + I_{soil} + I_{inhsoil} + I_{ag} + I_{beef} + I_{milk} + I_{pork} + I_{domestic\ fowl} + I_{egg} + I_{dw}$$

resident subsistence fisher:

$$m = I_{inh} + I_{soil} + I_{inhsoil} + I_{ag} + I_{fish} + I_{dw}$$

resident subsistence American Indian[4]:

$$m = I_{inh} + I_{soil} + I_{inhsoil} + I_{ag} + I_{wild\ egg} + I_{game} + I_{fish} + I_{dw}$$

alternate resident subsistence American Indian #1[4]:

$$m = I_{inh} + I_{soil} + I_{inhsoil} + I_{ag} + I_{wild\ fowl} + I_{game} + I_{game\ organs} + I_{fish} + I_{fish\ organs} + I_{dw}$$

alternate resident subsistence American Indian #2[4]:

$$m = I_{inh} + I_{soil} + I_{inhsoil} + I_{ag} + I_{beef} + I_{milk} + I_{domestic\ fowl} + I_{wild\ fowl} + I_{game} + I_{fish} + I_{dw}$$

where the individual daily intake terms, $I$, will be calculated from equations above, with the exception of $I_{inh}$ and $I_{inhsoil}$, which as discussed in further detail below.

$h$ = biological half-life of COPC or effective half-life of ROPC (includes a radioactive decay component, see below) in the mother (days); $h$ is COPC- and ROPC-specific and provided in Supplement 4

$f_1$ = fraction of ingested COPC or ROPC that is stored in fat (unitless)

$f_2$ = fraction of mother's weight that is fat (unitless)

$BW_{mother}$ = body weight of mother (kg)

0.693 = natural logarithm of 2; the quantity $h$/ln(2) equates to $1/k_{elim}$, where $k_{elim}$ is the elimination constant in the model that HHRAP Table C-3-1 was derived from (refer to EPA 1999a, Eq. 9-2)

$CF$ = units conversion factor of $10^{-9}$ (pg/mg)

The equation above assumes that the contaminant concentration has reached steady state (hence, exposure duration and frequency, and averaging time of the mother are not incorporated), and that the concentration of contaminant is the same in milk fat as it is in maternal body fat.

The intake due to inhalation pathways, $I_{inh}$ and $I_{inhsoil}$, is average daily COPC intake via inhalation (ADI) computed according to HHRAP Table C-3-1 (the variables $I_{inh}$ and $I_{inhsoil}$ are used instead of *ADI* for consistency with the other intakes shown in this RAWP). Recalling the equation for exposure concentration shown in Sections 7.1.5.1 and 7.1.5.2, $I_{inh}$ and $I_{inhsoil}$ can be computed using a modified version of HHRAP Table C-3-1:

[4] The Hanford site worker maternal exposure does not include exposure during retirement. Maternal exposure in tribal scenarios does not include sweat lodge exposures (presumably, mothers suspend sweat lodge participation during pregnancy and the subsequent the breast-feeding period).

$$I_{inh\ or\ inhsoil} = EC_{inh\ or\ inhsoil} \cdot \frac{IR}{BW_{mother}} \cdot CF$$ (HHRAP Table C-3-1 modified for exposure concentration)

where:

| | | |
|---|---|---|
| $I_{inh}$ | = | intake of COPCs via inhalation of emissions (mg/kg·day) |
| $I_{inhsoil}$ | = | intake of COPCs via inhalation of resuspended soil (mg/kg·day) |
| $EC_{inh}$ | = | exposure concentration of COPCs through inhalation of emissions (mg/m$^3$) |
| $EC_{inhsoil}$ | = | exposure concentration of COPCs through inhalation of resuspended soil (mg/m$^3$) |
| $IR$ | = | inhalation rate of mother (m$^3$/hr) |
| $BW_{mother}$ | = | body weight of mother (kg) |
| $CF$ | = | units conversion factor of $24\frac{\text{hr}}{\text{day}}$ |

For ROPCs, equations for $I_{inh}$ and $I_{inhsoil}$ are presented in Sections 7.1.5.1 and 7.1.5.2, respectively.

The effective half-life for ROPCs includes a component for biological half-life, and a component for loss due to radioactive decay (isotope nuclear half-life).

ROPCs: $$h = \frac{h_b \cdot h_r}{h_b + h_r}$$ (Equation 7-1)

where:

| | | |
|---|---|---|
| $h$ | = | effective half-life of ROPC |
| $h_b$ | = | biological half-life of ROPC |
| $h_r$ | = | radiological half-life of ROPC |

Using this equation, the effective half-lives for the radionuclides of interest are given below:

| ROPC | Biological Half-life (days) | Radiological Half-life (days) | Effective Half-life (days) |
|---|---|---|---|
| Cesium-134 | $1.14 \times 10^2$ | $7.67 \times 10^2$ | $9.92 \times 10$ |
| Cesium-137 | $1.35 \times 10^2$ | $1.10 \times 10^4$ | $1.33 \times 10^2$ |
| Iodine-129 | $1.38 \times 10^2$ | $5.84 \times 10^9$ | $1.38 \times 10^2$ |
| Strontium-90 | $2.91 \times 10^2$ | $1.05 \times 10^4$ | $2.28 \times 10^3$ |

### 7.1.6 Exposure Parameters

The equations presented above are the basis for quantifying the exposure to COPCs and ROPCs experienced by a potential receptor. The values that will be used for each parameter identified in the equations are provided in Table 7-2 through Table 7-8 and described below. These parameters are

conservative to ensure that the exposures calculated in the SLRA overestimate, rather than underestimate, risk.

#### 7.1.6.1 Hanford Site Industrial Worker

For the Hanford site industrial worker scenario, exposure values are presented in Table 7-2.

The Hanford site industrial worker is assumed to work both indoors (4 hours per day) and outdoors (4 hours per day) and to consume 200 mg soil per work-day rather than the default 50 mg per day because of this outdoor activity.

The Hanford site industrial worker is assumed to live within the offsite receptor exposure grid in addition to working at the onsite ground maximum. Exposure assumptions for the time spent at home are the same as those for a resident (Section 7.1.6.2) corrected for time spent at work. For example:

- This receptor is assumed to spend 8 hours per day at work and 16 hours per day at home for the 250 days per year he or she is also at work, and 24 hours per day at home for the 100 days per year he or she is not at work. The receptor is assumed to spend 24 hours per day at home, 350 days per year, during retirement. The remaining 15 days of the year are spent on vacation, at a location presumably beyond the assessment area.
- This receptor consumes a total of 2 L to 3 L per day of drinking water from the Columbia River maximum (i.e., on workdays the receptor consumes 2 L at work and 1 L at home, on nonwork days the receptor consumes 2 L at home).
- This receptor is assumed to spend 20 years working at the onsite ground maximum and living within the Hanford offsite grid, and another 10 years as a retiree within the Hanford offsite grid (for a total residential exposure duration of 30 years). As was the case during the 20-year career of the worker, the retired worker still spends 15 days of the year on vacation, at a location beyond the assessment area.

Soil ingestion rates are assumed to be independent of exposure time and, therefore, are not corrected for time spent at work and at home (i.e., the worker consumes 200 mg soil per day at work and 100 mg soil per day at home for a total of 300 mg soil per day, 250 days per year and 100 mg soil per day, 100 days per year).

#### 7.1.6.2 Residential Scenarios

For residential scenarios (resident, resident subsistence farmer, resident subsistence fisher), exposure values are presented in Table 7-3, Table 7-4, and Table 7-5 and are taken primarily from the HHRAP (EPA 2005a). Several exposure parameters (inhalation rate, soil ingestion rate, drinking water ingestion rate) differ from the HHRAP default values in order to be consistent with other EPA Region 10 assessments (CCN 063805, CCN 063806, CCN 063807). The source of each exposure parameter is provided, along with the value used, in Table 7-3, Table 7-4, and Table 7-5.

The adult resident and resident subsistence fisher are assumed to live within the Hanford offsite grid for 30 years. The resident subsistence farmer is assumed to live within the Hanford offsite grid for 40 years. The child is assumed to be exposed for 6 years for all three residential scenarios.

Consumption rates of food are for contaminated food grown at the receptor's home (or, for fish, from the Columbia River maximum) and do not include food purchased from uncontaminated sources. Food consumption rates are presented in units of kg dry weight (DW) produce per kg body weight per day and kg fresh weight (FW) animal product per kg body weight per day. Consumption rates for an adult and child are summarized below (refer to HHRAP Table C-1-2 and C-1-3).

| | Consumption Rate [a] (kg/kg·day) | |
|---|---|---|
| **Receptor and Food Product** | **Adult** | **Child** |
| **Resident** | | |
| Exposed aboveground produce | 0.00032 | 0.00077 |
| Protected aboveground produce | 0.00061 | 0.0015 |
| Belowground produce | 0.00014 | 0.00023 |
| Total produce | 0.0011 | 0.0025 |
| **Resident Subsistence Fisher** | | |
| Exposed aboveground produce | 0.00032 | 0.00077 |
| Protected aboveground produce | 0.00061 | 0.0015 |
| Belowground produce | 0.00014 | 0.00023 |
| Total produce | 0.0011 | 0.0025 |
| Fish | 0.00125 | 0.00088 |
| **Resident Subsistence Farmer** | | |
| Exposed aboveground produce | 0.00047 | 0.00113 |
| Protected aboveground produce | 0.00064 | 0.00157 |
| Belowground produce | 0.00017 | 0.00028 |
| Total produce | 0.0013 | 0.0030 |
| Beef | 0.00122 | 0.00075 |
| Pork | 0.00055 | 0.00042 |
| Poultry | 0.00066 | 0.00045 |
| Eggs | 0.00075 | 0.00054 |
| Total meat and eggs | 0.0032 | 0.0022 |
| Dairy | 0.01367 | 0.02268 |

[a] For the metals mercury, selenium, and cadmium, the concentration in beef, milk, and pork, and the consumption rate are in kilograms dry weight per day (EPA 2005a). Consumption rates include food-preparation loss (refer to discussion below).

The estimated consumption rates are based on food products as purchased or harvested and include food products lost during preparation. Loss during preparation is dependent on the type of food. Preparation loss for produce results from peeling, trimming, washing, and cooking. Losses from these activities are dependent on the type of produce; for example, some produce (e.g., potatoes) may be routinely washed, peeled, and cooked, while other produce (e.g., grapes) may be eaten whole and raw, and still others may be prepared and cooked or eaten whole and raw (e.g., carrots).

Preparation loss from meat (e.g., beef, pork, chicken) results from cutting, shrinkage, excess fat, bones, scraps, and juices, as well as dripping and volatile losses during cooking. Preparation losses for beef are estimated as approximately 27 % from cooking and 24 % from cutting, shrinkage, bones, etc. (EPA 1997a). These losses result in a preparation loss factor (*PL*) of 0.55 [(1 - 0.27) × (1 - 0.24)]. Preparation losses for chicken are estimated as approximately 32 % from cooking and 31 % from cutting, shrinkage, bones, etc. (EPA 1997a). These losses result in a *PL* of 0.47 [(1 - 0.32) × (1 - 0.31)]. Preparation factors for pork are estimated as approximately 28 % from cooking and 36 % from cutting, shrinkage, bones, etc. (EPA 1997a), for a net *PL* of 46%.

Preparation loss does not apply to milk and eggs.

Exposure parameters for the nursing infant are for an infant from ages 0 to 12 months. Exposure parameters for the mother of the nursing infant are the same as those presented for the adult resident and resident subsistence farmer.

#### 7.1.6.3 Resident Subsistence American Indians

For the resident subsistence American Indians , exposure values are presented in Table 7-6 and are taken primarily from the TC&WM EIS (DOE 2009). Other parameters were taken from the EFH (EPA 1997a). Children's exposure parameters were developed by proportioning the child caloric intake reported in EPA guidance (CSEFH, EPA 2008), according to the various proportions of meat, vegetable, roots, etc. in the diet of the adult American Indian hunter-gatherer as reported in the TC&WM EIS.

There are two alternate resident subsistence American Indian scenarios are included in this SLRA that are not necessarily endorsed by the DOE-ORP. These receptor scenarios were developed to more closely represent the lifestyle described by guidance documents issued by local tribes and to address uncertainties associated with the tribal lifestyle derived from the TC&WM EIS. The lifestyle and exposure parameters of the first alternate resident subsistence American Indian are primarily based on data from *Exposure Scenario for CTUIR Traditional Subsistence Lifeways* (Harris and Harper 2004) and *Application of the CTUIR Traditional Lifeways Exposure Scenario in Hanford Risk Assessments* (Harris 2008). The lifestyle and exposure parameters of the second alternate resident subsistence American Indian are primarily based on data from *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment* (RUDOLFI Inc. 2007). Other parameters were taken from the "A Native American Exposure Scenario" (Harris and Harper 1997) or from the EFH. Children's exposure parameters were developed by proportioning the child caloric intake reported in the CSEFH according to the various proportions of meat, vegetable, roots, etc. in the diet of the adult American Indian as reported in the guidance documents cited above.

The resident subsistence American Indian resident is assumed to live within the Hanford offsite receptor exposure grid for 70 years. The child is assumed to be exposed for 6 years. Each receptor is presumed to live at the offsite location 365 days/year with the exception of the first alternate resident subsistence

American Indian, who spends 1 day/month at a ceremonial location (assumed to occur at Gable Mountain[5]).

Three separate exposure duration values will be used for this scenario:

- The recommended exposure duration of 70 years assumes that this receptor is exposed during his or her entire lifetime. This exposure duration will be used for adult exposures through all pathways except those noted below. In the current scenario, the receptor is assumed to be exposed during the entire 40-year operating period of the WTP, and the next 30 years after WTP shutdown, at the offsite location. Likewise, while the WTP is in operation, it assumed that the hunting and gathering area is limited, as shown by Area I in Figure 7-2. Once WTP shutdown occurs, the hunting and gathering area is expanded to encompass the region shown as Area II in Figure 7-2. For the future scenario, the receptor is assumed to arrive at the offsite location at the same time as WTP shutdown, and is assumed to spend the entire exposure duration at that location (refer to Section 7.1.3.9). Food of the future receptor is assumed to come from the hunting and gathering Area II, as shown in Figure 7-2.
- An adult exposure duration of 40 years (the operating lifetime of the WTP) will be used for direct exposure to contaminants in air (inhalation and external radiation in air) because these exposures will last only as long as emissions from the WTP are occurring. A 40-year exposure duration will also be used for ingestion of carbon-14 and tritium in plants because these ROPCs are transferred directly to plant tissue from air, rather than being transferred from soil (see Section 6.6), and will only accumulate these ROPCs as long as emissions from the melter are occurring.
- An exposure duration of 6 years will be used for the resident subsistence American Indian child.

The subsistence tribal resident is assumed to obtain wild food gathered from the Hanford site, and in the case of the second subsistence tribal resident, is assumed to complement this diet with homegrown domestic foods. Applicable consumption rates of food presented in below do not include food purchased or collected from uncontaminated sources. Food consumption rates are presented in units of kg dry weight produce per kg body weight per day and kg fresh weight animal product per kg body weight per day. To put these values into perspective, consumption rates for an adult and child are summarized below.

| | Consumption Rate (kg/kg·day) | |
|---|---|---|
| **Food Product** | **Adult** | **Child** |
| **Subsistence Tribal Resident** | | |
| Exposed aboveground produce | 0.0025 | 0.0038 |
| Protected aboveground produce | 0.013 | 0.027 |
| Total produce | 0.016 | 0.031 |
| Wild game | 0.0060 | 0.013 |

[5] The actual location of tribal ceremonial activities varies with the nature of the activity, and is considered confidential. Gable Mountain is chosen as the location for ceremonial activities in the WTP risk assessment because it is a place of significance and is in close proximity to WTP, making related exposures at this location conservative and bounding.

| Food Product | Consumption Rate (kg/kg·day) Adult | Child |
|---|---|---|
| Wild fowl eggs | 0.00074 | 0.0022 |
| Fish | 0.0088 | 0.0101 |
| Total meat and eggs | 0.016 | 0.025 |
| **Alternate Subsistence Tribal Resident #1** | | |
| Exposed aboveground produce | 0.0048 | 0.016 |
| Belowground produce | 0.0063 | 0.021 |
| Total produce | 0.011 | 0.037 |
| Wild game | 0.0016 | 0.0050 |
| Wild game organs | 0.00018 | 0.00056 |
| Wild fowl | 0.00089 | 0.0029 |
| Fish | 0.0080 | 0.025 |
| Fish organs | 0.00089 | 0.0027 |
| Total meat | 0.012 | 0.036 |
| **Alternate Subsistence Tribal Resident #2** | | |
| Exposed aboveground domestic produce | 0.0072 | 0.0070 |
| Exposed aboveground wild produce | 0.0069 | 0.0067 |
| Belowground wild produce | 0.0062 | 0.0060 |
| Total produce | 0.020 | 0.0197 |
| Beef | 0.0040 | 0.0053 |
| Domestic Poultry | 0.0020 | 0.0026 |
| Wild game | 0.0027 | 0.0036 |
| Wild fowl | 0.0013 | 0.0017 |
| Fish | 0.0074 | 0.023 |
| Total meat and eggs | 0.017 | 0.036 |

Values have been converted to a per unit weight basis. Consumption rates do not include food preparation loss (refer to discussion below).

Quantitative preparation loss factors are not available for produce, wild game, or wildfowl. Preparation loss for produce will be assumed to be zero. Use of beef/chicken *PL*s could over-estimate losses for wild game and wildfowl because American Indian receptors may utilize more of the animal than other populations, and the ratio of lean meat to fat is typically higher in wild game and wildfowl, potentially resulting in lower preparation losses.

Conversely, the ratio of meat to skin and bone is lower in wild game and wildfowl than domestic beef and chicken, which could result in higher preparation losses. Based on these factors, preparation losses for wild game/wildfowl are estimated to be half that of domestic beef and chicken; thus *PL* factors of 0.76 [(1 – 0.135) × (1 – 0.12)] for wild game and 0.72 [(1 – 0.15) × (1 – 0.15)] for wildfowl will be used (EPA 1997a).

An inhalation rate of 8400 $m^3$ per year (0.959 $m^3$/hr) will be used for the resident subsistence American Indian adult per the TC&WM EIS. DOE 1996 reports a child inhalation rate of 15 $m^3$/day (0.625 $m^3$/hr). Guidance provided by the CTUIR was used to establish an adult and child inhalation rate of 25 $m^3$/day (1.04 $m^3$/hr) and 15 $m^3$/day (0.625 $m^3$/hr), respectively, for the alternate resident subsistence American Indian #1. Guidance provided by the Yakama Nation was used to establish an adult and child inhalation rate of 26 $m^3$/day (1.08 $m^3$/hr) and 16 $m^3$/day (0.667$m^3$/hr), respectively, for the alternate resident subsistence American Indian #2. Exposure parameters for the nursing infant are for an infant ages 0 to 12 months. Exposure parameters for the mother of the nursing infant are the same as those presented for the adult resident subsistence American Indian with the exception of maternal exposure duration which assumed equal to 25 years.

### 7.1.7 Exposure Point Concentrations

The EPCs used for estimating intakes/doses of both COPCs and ROPCs are dependent on the location of the receptor. The location of the various receptor populations identified for the quantitative risk assessment will correspond to the receptor grid nodes defined during air dispersion modeling (Section 6.1). In keeping with the protective approach for the SLRA, the EPCs used to determine receptor intakes/dose will be location and constituent (carcinogen or noncarcinogen) specific, and incorporate maximum concentration and deposition results as discussed in Section 6.1.4.3.

Air dispersion modeling will be used to identify points of maximum emission concentrations and deposition at three locations of interest[6]: at the onsite location of maximum concentration (i.e., the onsite ground maximum), at Gable Mountain, and at the Columbia River. To simplify the risk assessments, it will be assumed that receptor populations are present at these exposure locations. For example, while offsite residential receptor populations are present (e.g., in Richland), residents may not be onsite, at Gable Mountain, or at the Columbia River maximum. However, for the risk assessment, it is assumed that a variety of residential receptors are present at this location.

The four exposure locations are described in Section 7.1.1 and again, briefly, below (see Figure 7-1):

- Onsite ground maximum
- Hanford offsite
- Gable Mountain maximum
- Columbia River maximum

[6] Exposures in the offsite grid will utilize 90th percentiles to represent exposures that might otherwise be over-predicted by maximum air concentration and deposition values. Due to the migratory nature of game animals, and the diversity of vegetation across the site, the 95% UCL of the median from air dispersion modeling will be used to compute EPCs in the hunting/gathering area(s).

Because the point of maximum concentration may be different for airborne COPCs and ROPCs, and COPCs and ROPCs deposited through wet and dry deposition mechanisms, EPA (2005a) recommends selecting the point of maximum concentration. The EPA also notes that only 1 to 3 receptor grid nodes were typically selected per land use area. For the WTP, emissions will be modeled separately for three stacks (PT, LAW, and HLW Facilities) with nine points of maximum concentration possible from each stack:

- Maximum vapor-phase air concentration
- Maximum particle-phase air concentration (1 micron diameter particles)
- Maximum particle-bound-phase air concentration (2.5 micron diameter particles)
- Maximum vapor-phase wet deposition
- Maximum particle-phase wet deposition (1 micron diameter particles)
- Maximum particle-bound-phase wet deposition (2.5 micron diameter particles)
- Maximum vapor-phase dry deposition
- Maximum particle-phase dry deposition (1 micron diameter particles)
- Maximum particle-bound-phase dry deposition (2.5 micron diameter particles)

Thus, there are a total of 27 possible maximum concentrations (3 stacks × 9 phases) at each of the three locations of interest for each year of air modeling data. Because more than one maximum concentration often occurs at the same receptor grid node, it is more likely that a dozen or so grid nodes with maximum concentrations will be at each location of interest (rather than the 135 possible [3 stacks × 9 phases × 5 years]). Using the highest discrete values of $Cyv$, $Cyp_1$, $Cyp_{2.5}$, $Dydv$, $Dydp_1$, $Dydp_{2.5}$, $Dywv$, $Dywp_1$, and $Dywp_{2.5}$ from the onsite maximum, Columbia River, and Gable Mountain grids will result in the highest degree of conservatism. However, in large receptor exposure grids such as the offsite grid, the corresponding exposures to such extreme deposition and air concentrations are improbable and could result in risk estimates that are highly improbable. Accordingly, the 90th percentile of $Cyv$, $Cyp_1$, $Cyp_{2.5}$, $Dydv$, $Dydp_1$, $Dydp_{2.5}$, $Dywv$, $Dywp_1$, and $Dywp_{2.5}$ will serve as input to EPC computations. To reflect the migratory nature of game animals and wide dispersion of vegetation, the distribution-free 95 % upper confidence limit of the median provides a sufficiently conservative estimate of air concentration and deposition in the hunter/gatherer area(s).

In order to help quantify the degree of conservatism associated with using the 90th percentile from the offsite grid, the location and species values associated with the point of highest annual total air concentration and deposition will be determined in the uncertainty assessment. The EPCs associated with the grid nodes where these values occur will be computed for comparison to those EPCs computed using 90th percentiles. Results will be discussed in the uncertainty assessment of the PRA (refer to Section 10).

#### 7.1.7.1 Exposure Point Concentrations in Air

The EPCs will be calculated as described in Section 6.1 (air dispersion modeling). Chronic air concentrations are assumed to remain the same for the entire 40-year operating lifetime of the WTP. Acute air concentrations represent the worst-case, one-hour meteorological conditions and will be used for evaluating the acute scenario only (refer to Section 7.2.1.2).

#### 7.1.7.2 Exposure Point Concentrations in Soil, Surface Water, and Sediment

Concentrations of COPCs and ROPCs in soil, surface water, and sediment are estimated from deposition rates predicted by the air dispersion modeling as described in Sections 6.2 (soil), 6.3 (surface water), and 6.4 (sediment). Deposition is assumed to occur for the potential operating lifespan of the facility (40 years), and ceases after WTP shutdown.

Separate soil concentrations will be estimated for the current and future exposure periods for each receptor at the appropriate locations as described in Section 7.1.3. After WTP shutdown, there is no further accumulation of contaminants, so the annual average surface water concentrations are zero for future exposures to these media. Soil and sediment, however, will still contain residual contaminants and thus will contribute to risk in the future scenario.

#### 7.1.7.3 Exposure Point Concentrations in Plants

Exposure point concentrations for produce (fruits and vegetables) and wild plants will be calculated as described in Section 6.6. Current EPCs for homegrown and wild plants will include vapor-phase transfer from air to plants, deposition from air onto plants, and root uptake from soil into the aboveground and belowground portions of plants. Future EPCs for home grown and wild plants will only include root uptake from soil into the above and belowground portions of plants because airborne emissions will not be present following WTP shutdown.

#### 7.1.7.4 Exposure Point Concentrations for Animal Tissue (Domestic Livestock and Wild Game)

Exposure point concentrations in animal products (such as beef, milk, and wild game) will be modeled as described here. As noted in Section 6.7, this modeling effort is slightly different for the human health and ecological risk assessments. See Section 8 for the modeling required for the ecological risk assessment. This section describes the modeling for use in the HHRA and includes modeling to determine EPCs for the following animal tissue:

- Beef
- Milk
- Pork
- Chicken
- Wildfowl
- Chicken eggs
- Wildfowl eggs
- Wild game (e.g., deer)

Edible tissue concentrations will be calculated for the HHRA using feed concentrations, ingestion rates, bioaccumulation factors, and other parameters in model equations from EPA 2005a. Current and future feed concentrations (such as soil, forage, silage, and grain concentrations) will be determined as described in Section 6.6. Ingestion rates and other parameters are generally from the HHRAP (EPA 2005a) and can be found in Table 7-9. Bioaccumulation factors are COPC- and ROPC-specific and can be found in Supplement 4. As with the plant modeling (see Section 6.6.3), the bioaccumulation factors used to model

animal tissue and animal products must be corrected to account for mass balance. The mass balance correction for animal tissue is presented at the end of this section.

**Exposure Point Concentrations in Beef**

Beef cattle are assumed to consume forage, silage, and grain, as well as surface soil (i.e., 2 cm untilled soil). The equation to determine concentrations in beef tissue (EPA 2005a) for all constituents is:

$$A_{beef} = \left[\left(\sum_{i=1}^{3} F_i \cdot Qp_{i(beef)} \cdot P_i\right) + Qs_{soil(beef)} \cdot Cs_2 \cdot Bs\right] \cdot Ba_{beef} \cdot MF \qquad \text{(HHRAP Table B-3-10)}$$

where:

$A_{beef}$ = concentration of COPC or ROPC in beef (mg/kg for COPCs and pCi/g for ROPCs)

$F_i$ = fraction of plant-type *i* grown on contaminated soil and ingested by the beef cattle (unitless). The three plant types consumed by the beef cattle are forage, silage, and grain. The recommended default value of 1.0 (EPA 2005a) shown in Table 7-9 as $F_{plant}$ is used for all plant types.

$Qp_{i(beef)}$ = quantity of plant type *i* eaten by the beef cattle per day (kg/day). $Qp_{i(beef)}$ is shown in Table 7-9. The recommended values (EPA 2005a) for beef cattle raised by subsistence farmers are used: $Qp_{forage(beef)}$ = 8.8 kg/day is the amount of forage eaten by the beef cow, $Qp_{silage(beef)}$ = 2.5 kg/day for is the amount of silage eaten by the beef cow, and $Qp_{grain(beef)}$ = 0.47 kg/day is the amount of grain eaten by the beef cow.

$P_i$ = concentration of COPC or ROPC in plant type *i* that is ingested by the beef cattle (mg/kg for COPCs and pCi/g for ROPCs). $P_i$ is COPC- and ROPC-specific and calculated as follows:

$$P_{forage} = Pd_{forage} + Pv_{forage} + Pr_{ag(forage)}$$
$$P_{silage} = Pd_{silage} + Pv_{silage} + Pr_{ag(silage)}$$
$$P_{domestic\ grain} = Pr_{ag(domestic\ grain)}$$

where, for all constituents except carbon-14 and tritium:

$Pd_{forage}$ and $Pd_{silage}$ are calculated in Eq. 5-14 in the HHRAP

$Pv_{forage}$ and $Pv_{silage}$ are calculated in Eq. 5-18 in the HHRAP

$Pr_{ag(forage)}$ is calculated in Eq. 5-20A in the HHRAP using 15 cm soil (root-zone)

$Pr_{ag(silage)}$ and $Pr_{ag(domestic\ grain)}$ are calculated in Eq. 5-20A in the HHRAP using 20 cm soil (tilled)

For carbon-14, all plant concentrations (i.e., $P_{forage}$, $Pr_{silage}$, and $Pr_{grain}$) take on the plant concentration value, $C_{V(C-14)}$, calculated from air concentration as described by the Nuclear Regulatory Commission (NRC) (1977) (see Section 6.6.2). For tritium, all plant concentrations take on the plant concentration value, $C_{V(H-3)}$,

calculated from air concentration and absolute humidity as described by the NRC (1977) (see Section 6.6.2).

In the future scenario, $Pd$, $Pv$, $C_{V(C-14)}$, and $C_{V(H-3)}$ are all zero because there are no longer any emissions (no direct deposition or air-to-plant uptake).

$Qs_{soil(beef)}$ = quantity of soil ingested by the beef cattle (kg/day). The recommended default value of 0.5 kg/day (EPA 2005a) is used (see Table 7-9 of this work plan).

$Cs_2$ = soil concentration at the 2 cm soil depth (mg/kg for COPCs and pCi/g for ROPCs) calculated according to Section 6.2

$Bs$ = soil bioavailability factor (unitless). The recommended default value of 1.0 (EPA 2005a) is used (see Table 7-9).

$Ba_{beef}$ = biotransfer factor for beef (day/kg). $Ba_{beef}$ is COPC- and ROPC-specific and shown in Supplement 4. If no value is available for $Ba_{beef}$, then $A_{beef}$ cannot be calculated and the ingestion of beef pathway cannot be evaluated in the HHRA. The values for $Ba_{beef}$ in Supplement 4 will be compared against the calculated mass-limited uptake factor for beef (shown in Appendix A) and the smaller of the two values will be used in the calculation of the beef concentration ($A_{beef}$).

$MF$ = metabolism factor (unitless). $MF$ is COPC- and ROPC-specific. The recommended default $MF$ values of 0.01 for bis(2-ethylhexyl)phthalate and 1.0 for all other constituents (EPA 2005a) are used (see Table 7-9).

**Exposure Point Concentration in Milk**

Dairy cattle are assumed to consume forage, silage, and grain, as well as surface soil (i.e., 2 cm untilled soil). The equation to determine concentrations in milk (EPA 2005a) for all constituents is:

$$A_{milk} = \left[\left(\sum_{i=1}^{3} F_i \cdot Qp_{i(milk)} \cdot P_i\right) + Qs_{soil(milk)} \cdot Cs_2 \cdot Bs\right] \cdot Ba_{milk} \cdot MF \qquad \text{(HHRAP Table B-3-11)}$$

where:

$A_{milk}$ = concentration of COPC or ROPC in milk (mg/kg for COPCs and pCi/g for ROPCs).

$F_i$ = fraction of plant-type $i$ grown on contaminated soil and ingested by dairy cattle (unitless). The three plant types consumed by the dairy cattle are forage, silage, and grain. The recommended default value of 1.0 (EPA 2005a) shown in Table 7-9 as $F_{plant}$ is used for all plant types.

$Qp_{i(milk)}$ = quantity of plant type $i$ eaten by the dairy cattle per day (kg/day). $Qp_{i(milk)}$ is shown in Table 7-9. The recommended values (EPA 2005a) for dairy cattle raised by subsistence farmers are used: $Qp_{forage(milk)}$ = 13.2 kg/day is the amount of forage eaten by the dairy cow, $Qp_{silage(milk)}$ = 4.1 kg/day is the amount of silage eaten by the dairy cow, and $Qp_{grain(milk)}$ = 3.0 kg/day is the amount of grain eaten by the dairy cow.

$P_i$ = concentration of COPC or ROPC in plant type $i$ that is ingested by the dairy cattle (mg/kg for COPCs and pCi/g for ROPCs). $P_i$ is COPC- and ROPC-specific and calculated as follows:

$$P_{forage} = Pd_{forage} + Pv_{forage} + Pr_{ag(forage)}$$
$$P_{silage} = Pd_{silage} + Pv_{silage} + Pr_{ag(silage)}$$
$$P_{domestic\ grain} = Pr_{ag(domestic\ grain)}$$

where, for all constituents except carbon-14 and tritium:

$Pd_{forage}$ and $Pd_{silage}$ are calculated in Eq. 5-14 in the HHRAP

$Pv_{forage}$ and $Pv_{silage}$ are calculated in Eq. 5-18 in the HHRAP

$Pr_{ag(forage)}$ is calculated in Eq. 5-20A in the HHRAP using 15 cm soil (root-zone)

$Pr_{ag(silage)}$ and $Pr_{ag(domestic\ grain)}$ are calculated in Eq. 5-20A in the HHRAP using 20 cm soil (tilled)

For carbon-14, all plant concentrations (i.e., $P_{forage}$, $Pr_{silage}$, and $Pr_{domestic\ grain}$) take on the plant concentration, $C_{V(C-14)}$, calculated from air concentration as described by the NRC (1977) (see Section 6.6.2). For tritium, all plant concentrations take on the plant concentration value, $C_{V(H-3)}$, calculated from air concentration and absolute humidity as described by the NRC (1977) (see Section 6.6.2).

In the future scenario, $Pd$, $Pv$, $C_{V(C-14)}$, and $C_{V(H-3)}$ are all zero because there are no longer any emissions (no direct deposition or air-to-plant uptake).

$Qs_{soil(milk)}$ = quantity of soil ingested by the dairy cattle (kg/day). The recommended default value of 0.4 kg/day (EPA 2005a) is used (see Table 7-9).

$Cs_2$ = soil concentration at the 2 cm soil depth (mg/kg for COPCs and pCi/g for ROPCs) calculated according to Section 6.2

$Bs$ = soil bioavailability factor (unitless). The recommended default value of 1.0 (EPA 2005a) is used (see Table 7-9).

$Ba_{milk}$ = biotransfer factor for milk (day/kg). $Ba_{milk}$ is COPC- and ROPC-specific and shown in Supplement 4. If no value is available for $Ba_{milk}$, then $A_{milk}$ cannot be calculated, and the ingestion of milk pathway cannot be evaluated in the HHRA. The values for $Ba_{milk}$ in Supplement 4 will be compared against the calculated mass-limited uptake factor for milk (shown in Appendix A), and the smaller of the two values will be used in the calculation of the milk concentration ($A_{milk}$).

$MF$ = metabolism factor (unitless). $MF$ is COPC- and ROPC-specific. The recommended default $MF$ values of 0.01 for bis(2-ethylhexyl)phthalate and 1.0 for all other constituents (EPA 2005a) are used (see Table 7-9).

**Exposure Point Concentration in Pork**

Swine are assumed to consume silage and grain, as well as surface soil (i.e., 2 cm untilled soil). The equation to determine pork concentrations (EPA 2005a) for all constituents is:

$$A_{pork} = \left[\left(\sum_{i=2}^{3} F_i \cdot Qp_{i(pork)} \cdot P_i\right) + Qs_{soil(pork)} \cdot Cs_2 \cdot Bs\right] \cdot Ba_{pork} \cdot MF \qquad \text{(HHRAP Table B-3-12)}$$

where:

$A_{pork}$ = concentration of COPC or ROPC in pork (mg/kg for COPCs and pCi/g for ROPCs)

$F_i$ = fraction of plant-type *i* grown on contaminated soil and ingested by the swine (unitless). The two plant types consumed by the swine are silage and grain. The recommended default value of 1.0 (EPA 2005a) shown in Table 7-9 as $F_{plant}$ is used for both plant types.

$Qp_{i(pork)}$ = quantity of plant type *i* eaten by the swine per day (kg/day). $Qp_i$ is shown in Table 7-9. The recommended values (EPA 2005a) for swine raised by subsistence farmers are used: $Qp_{silage(pork)}$ = 1.4 kg/day is the amount of silage eaten by the swine, and $Qp_{grain(pork)}$ = 3.3 kg/day is the amount of grain eaten by the swine.

$P_i$ = concentration of COPC or ROPC in plant type *i* that is ingested by the swine (mg/kg for COPCs and pCi/g for ROPCs). $P_i$ is COPC- and ROPC-specific and calculated as follows:

$$P_{silage(pork)} = Pd_{silage} + Pv_{silage} + Pr_{ag(silage)}$$
$$P_{domestic\ grain} = Pr_{ag(domestic\ grain)}$$

where, for all constituents except carbon-14 and tritium:

$Pd_{silage}$ is calculated in Eq. 5-14 in the HHRAP

$Pv_{silage}$ is calculated in Eq. 5-18 in the HHRAP

$Pr_{ag(silage)}$ and $Pr_{ag(domestic\ grain)}$ are calculated in Eq. 5-20A in the HHRAP using 20 cm soil (tilled)

For carbon-14, both plant concentrations (i.e., $Pr_{silage}$ and $Pr_{domestic\ grain}$) take on the plant concentration value, $C_{V(C-14)}$, calculated from air concentration as described by the NRC (1977), see Section 6.6.2. For or tritium, both plant concentrations take on the plant concentration value, $C_{V(H-3)}$, calculated from air concentration and absolute humidity as described by the NRC (1977) (see Section 6.6.2).

In the future scenario, $Pd$, $Pv$, $C_{V(C-14)}$, and $C_{V(H-3)}$ are all zero because there are no longer any emissions (no direct deposition or air-to-plant uptake).

$Qs_{soil(pork)}$ = quantity of soil ingested by the swine (kg/day). The recommended default value of 0.37 kg/day (EPA 2005a) is used (see Table 7-9).

$Cs_2$ = soil concentration at the 2 cm soil depth (mg/kg for COPCs and pCi/g for ROPCs) calculated according to Section 6.2.

$Bs$ = soil bioavailability factor (unitless). The recommended default value of 1.0 (EPA 2005a) is used (see Table 7-9).

$Ba_{pork}$ = biotransfer factor for pork (day/kg). $Ba_{pork}$ is COPC- and ROPC-specific and shown in Supplement 4. If no value is available for $Ba_{pork}$, then $A_{pork}$ cannot be calculated, and the ingestion of pork pathway cannot be evaluated in the HHRA. The values for $Ba_{pork}$ in Supplement 4 will be compared against the calculated

mass-limited uptake factor for pork (shown in Appendix A), and the smaller of the two values will be used in the calculation of the pork concentration ($A_{pork}$).

$MF$ = metabolism factor (unitless). $MF$ is COPC- and ROPC-specific. The recommended default $MF$ values of 0.01 for bis(2-ethylhexyl)phthalate and 1.0 for all other constituents (EPA 2005a) are used (see Table 7-9).

**Exposure Point Concentration in Chicken**

Chickens are assumed to consume grain grown on a farm, as well as surface soil (i.e., 2 cm untilled soil). The grain eaten by chickens is grown in tilled (20 cm depth) soil. The equation to determine chicken concentrations (EPA 2005a) for all constituents is:

$$A_{chicken} = \left(F_{grain} \cdot Qp_{grain(chicken)} \cdot P_{domestic\ grain} + Qs_{soil(chicken)} \cdot Cs_2 \cdot Bs\right) \cdot Ba_{chicken} \qquad \text{(HHRAP Table B-3-14)}$$

where:

$A_{chicken}$ = concentration of COPC or ROPC in chicken meat (mg/kg for COPCs and pCi/g for ROPCs).

$F_{grain}$ = fraction of grain grown on contaminated soil and ingested by the chicken (unitless). The recommended default value of 1.0 (EPA 2005a) shown in Table 7-9 as $F_{plant}$ is used for grain.

$Qp_{grain(chicken)}$ = quantity of grain eaten by the chicken per day (kg/day). The recommended value of $Qp_{grain(chicken)}$ = 0.2 kg/day (EPA 2005a) shown in Table 7-9 for chickens raised by subsistence farmers is used.

$P_{domestic\ grain}$ = concentration of COPC or ROPC in grain that is ingested by the chicken (mg/kg for COPCs and pCi/g for ROPCs). $P_{domestic\ grain}$ is COPC- and ROPC-specific and calculated as follows:

$$P_{domestic\ grain} = Pr_{ag(domestic\ grain)}$$

where, for all constituents except carbon-14 and tritium:

$Pr_{ag(grain)}$ is calculated in Eq. 5-20A in the HHRAP using 20 cm soil (tilled).

For carbon-14 and tritium, $Pr_{domestic\ grain}$ takes on the plant concentration value, $C_{V(C\text{-}14)}$ or $C_{V(H\text{-}3)}$, calculated from calculated from air concentration as described by the NRC (1977), respectively (see Section 6.6.2).

$Qs_{soil(chicken)}$ = quantity of soil ingested by the chicken (kg/day); the recommended default value of 0.022 kg/day (EPA 2005a) is used (see Table 7-9).

$Cs_2$ = soil concentration at the 2 cm soil depth (mg/kg for COPCs and pCi/g for ROPCs) calculated according to Section 6.2.

$Bs$ = soil bioavailability factor (unitless); the recommended default value of 1.0 (EPA 2005a) is used (see Table 7-9).

$Ba_{chicken}$ = biotransfer factor for chicken (day/kg); $Ba_{chicken}$ is COPC- and ROPC-specific and shown in Supplement 4. If no value is available for $Ba_{chicken}$, then $A_{chicken}$

cannot be calculated, and the ingestion of chicken pathway cannot be evaluated in the HHRA. The values for $Ba_{chicken}$ in Supplement 4 will be compared against the calculated mass-limited uptake factor for poultry (shown in Appendix A), and the smaller of the two values will be used in the calculation of the chicken concentration ($A_{chicken}$).

**Exposure Point Concentration in Wildfowl**

Wildfowl are assumed to consume grain grown in the wild, as well as surface soil (i.e., 2 cm untilled soil). The grain eaten by wildfowl is grown in root-zone (15 cm depth) soil. The equation to determine wildfowl concentrations (EPA 2005a) for all constituents is:

$$A_{fowl} = \left(F_{grain} \cdot Qp_{grain(chicken)} \cdot P_{wild\ grain} + Qs_{soil(chicken)} \cdot Cs_2 \cdot Bs\right) \cdot Ba_{fchicken} \quad \text{(HHRAP Table B-3-14)}$$

where:

$A_{fowl}$ = concentration of COPC or ROPC in wildfowl (mg/kg for COPCs and pCi/g for ROPCs).

$F_{grain}$ = fraction of grain grown on contaminated soil and ingested by the wildfowl (unitless). The recommended default value of 1.0 (EPA 2005a) shown in Table 7-9 as $F_{plant}$ is used for grain.

$Qp_{grain(chicken)}$ = quantity of grain eaten by the wildfowl per day (kg/day). The recommended value of $Qp_{grain(chicken)}$ = 0.2 kg/day (EPA 2005a value for chickens) shown in Table 7-9 is used for wildfowl.

$P_{wild\ grain}$ = concentration of COPC or ROPC in grain that is ingested by the wildfowl (mg/kg for COPCs and pCi/g for ROPCs). $P_{wild\ grain}$ is COPC- and ROPC-specific and calculated as follows:

$$P_{wild\ grain} = Pr_{ag(wild\ grain)}$$

where, for all constituents except carbon-14 and tritium:

$Pr_{ag(wild\ grain)}$ is calculated in Eq. 5-20A in the HHRAP using 15 cm soil (root-zone).

For carbon-14 and tritium, $P_{wild\ grain}$ takes on the plant concentration value, $C_{V(C-14)}$ or $C_{V(H-3)}$, calculated from calculated from air concentration as described by the NRC (1977), respectively (see Section 6.6.2).

$Qs_{soil(chicken)}$ = quantity of soil ingested by the wildfowl (kg/day); the recommended default value of 0.022 kg/day (EPA 2005a value for chickens) shown in Table 7-9 is used for wildfowl.

$Cs_2$ = soil concentration at the 2 cm soil depth (mg/kg for COPCs and pCi/g for ROPCs) calculated according to Section 6.2.

$Bs$ = soil bioavailability factor (unitless). The recommended default value of 1.0 (EPA 2005a) is used (see Table 7-9).

$Ba_{chicken}$ = biotransfer factor for wild fowl (day/kg). $Ba_{chicken}$ is COPC- and ROPC-specific and shown in Supplement 4. If no value is available for $Ba_{chicken}$, then $A_{fowl}$ cannot be calculated, and the ingestion of wildfowl pathway cannot be evaluated in the HHRA. The values for $Ba_{chicken}$ in Supplement 4 will be compared against the calculated mass-limited uptake factor for poultry (shown in Appendix A), and the smaller of the two values will be used in the calculation of the wildfowl concentration ($A_{fowl}$).

**Exposure Point Concentration in Chicken Eggs**

Chicken eggs are from chickens that are assumed to consume grain grown on a farm in tilled (20 cm depth) soil as well as surface soil (i.e., 2 cm untilled soil). The equation to determine chicken egg concentrations (EPA 2005a) for all constituents is:

$$A_{egg} = \left(F_{grain} \cdot Qp_{grain(chicken)} \cdot P_{domestic\ grain} + Qs_{soil(chicken)} \cdot Cs_2 \cdot Bs\right) \cdot Ba_{egg} \qquad \text{(HHRAP Table B-3-13)}$$

where:

$A_{egg}$ = concentration of COPC or ROPC in chicken eggs (mg/kg for COPCs and pCi/g for ROPCs).

$F_{grain}$ = fraction of grain grown on contaminated soil and ingested by the chicken (unitless). The recommended default value of 1.0 (EPA 2005a) shown in Table 7-9 as $F_{plant}$ is used for grain.

$Qp_{grain(chicken)}$ = quantity of grain eaten by the chicken per day (kg/day). The recommended value of $Qp_{grain(chicken)}$ = 0.2 kg/day (EPA 2005a) shown in Table 7-9 for chickens raised by subsistence farmers is used.

$P_{domestic\ grain}$ = concentration of COPC or ROPC in grain that is ingested by the chicken (mg/kg for COPCs and pCi/g for ROPCs). $P_{domestic\ grain}$ is COPC- and ROPC-specific and calculated as follows:

$$P_{domestic\ grain} = Pr_{ag(domestic\ grain)}$$

where, for all constituents except carbon-14 and tritium:

$Pr_{ag(domestic\ grain)}$ is calculated in Eq. 5-20A in the HHRAP using 20 cm soil (tilled).

For carbon-14 and tritium, $Pr_{grain(chicken)}$ takes on the plant concentration value, $C_{V(C-14)}$ or $C_{V(H-3)}$, calculated from calculated from air concentration as described by the NRC (1977), respectively (see Section 6.6.2).

$Qs_{soil(chicken)}$ = quantity of soil ingested by the chicken (kg/day). The recommended default value of 0.022 kg/day (EPA 2005a) is used (see Table 7-9).

$Cs_2$ = soil concentration at the 2 cm soil depth (mg/kg for COPCs and pCi/g for ROPCs) calculated according to Section 6.2

$Bs$ = soil bioavailability factor (unitless). The recommended default value of 1.0 (EPA 2005a) is used (see Table 7-9).

$Ba_{egg}$ = biotransfer factor for chicken eggs (day/kg). $Ba_{egg}$ is COPC- and ROPC-specific and shown in Supplement 4. If no value is available for $Ba_{egg}$, then $A_{egg}$ cannot be calculated, and the ingestion of chicken eggs pathway cannot be evaluated in the HHRA. The values for $Ba_{egg}$ in Supplement 4 will be compared against the calculated mass-limited uptake factor for eggs (shown in Appendix A), and the smaller of the two values will be used in the calculation of the chicken egg concentration ($A_{egg}$).

**Exposure Point Concentration in Wildfowl Eggs**

Wildfowl eggs are from wildfowl, which are assumed to consume grain grown in the wild in root-zone (15 cm depth) soil, as well as surface soil (i.e., 2 cm untilled soil). The equation to determine wildfowl egg concentrations (EPA 2005a) for all constituents is:

$$A_{egg(\ fowl\ )} = \left(F_{grain} \cdot Qp_{grain(\ chicken\ )} \cdot P_{wild\ \ grain} + Qs_{soil(\ chicken\ )} \cdot Cs_2 \cdot Bs\right) \cdot Ba_{egg} \qquad \text{(HHRAP Table B-3-13)}$$

where:

$A_{egg(fowl)}$ = concentration of COPC or ROPC in wildfowl eggs (mg/kg for COPCs and pCi/g for ROPCs).

$F_{grain}$ = fraction of grain grown on contaminated soil and ingested by the wildfowl (unitless). The recommended default value of 1.0 (EPA 2005a) shown in Table 7-9 as $F_{plant}$ is used for grain.

$Qp_{grain(chicken)}$= quantity of grain eaten by the wildfowl per day (kg/day). The recommended value of $Qp_{grain(chicken)}$ = 0.2 kg/day (EPA 2005a value for chickens) shown in Table 7-9 is used for wildfowl.

$P_{wild\ grain}$ = concentration of COPC or ROPC in grain that is ingested by the wildfowl (mg/kg for COPCs and pCi/g for ROPCs). $P_{wild\ grain}$ is COPC- and ROPC-specific, site-specific, plant type-specific, and calculated as follows:

$$P_{wild\ grain} = \mathrm{Pr}_{ag(wild\ grain)}$$

where, for all constituents except carbon-14 and tritium:

$\mathrm{Pr}_{ag(wild\ grain)}$ is calculated in Eq. 5-20A in the HHRAP using 15 cm soil (root-zone).

For carbon-14 and tritium, $P_{wild\ grain}$ takes on the plant concentration value, $C_{V(C\text{-}14)}$ or $C_{V(H\text{-}3)}$, calculated from calculated from air concentration as described by the NRC (1977), respectively (see Section 6.6.2).

$Qs_{soil(chicken)}$ = quantity of soil ingested by the wildfowl (kg/day). The recommended default value of 0.022 kg/day (EPA 2005a value for chickens) shown in Table 7-9 is used for wildfowl.

$Cs_2$ = soil concentration at the 2 cm soil depth (mg/kg for COPCs and pCi/g for ROPCs). $Cs_2$ is COPC- and ROPC-specific, site-specific, and calculated according to Section 6.2.

*Bs* = soil bioavailability factor (unitless). The recommended default value of 1.0 (EPA 2005a) is used (see Table 7-9).

$Ba_{egg}$ = biotransfer factor for wildfowl eggs (day/kg). $Ba_{egg}$ is COPC- and ROPC-specific and shown in Supplement 4. If no value is available for $Ba_{egg}$, then $A_{egg(fowl)}$ cannot be calculated and the ingestion of wildfowl eggs pathway cannot be evaluated in the HHRA. The values for $Ba_{egg}$ in Supplement 4 will be compared against the calculated mass-limited uptake factor for eggs (shown in Appendix A), and the smaller of the two values will be used in the calculation of the wildfowl egg concentration ($A_{egg(fowl)}$).

**Exposure Point Concentration in Wild Game**

Wild game animals (such as deer) are assumed to consume forage grown in root-zone (15 cm) soil only. The equation used to determine the concentration in wild game is adopted from the equation used for beef, only the contribution of silage and grain is not included since those feeds are unique to domestic livestock. The equation to determine concentrations in game tissue (EPA 2005a) for all constituents is:

$$A_{game} = \left(F_{forage} \cdot Qp_{forage(deer)} \cdot P_{forage}\right) \cdot Ba_{deer} \cdot MF \qquad \text{(modified HHRAP Table B-3-13)}$$

where:

$A_{game}$ = concentration of COPC or ROPC in wild game animals (mg/kg for COPCs and pCi/g for ROPCs).

$F_{forage}$ = fraction of forage grown on contaminated soil and ingested by the wild game animals (unitless). The recommended default value of 1.0 (EPA 2005a) shown in Table 7-9 as $F_{plant}$ is used for forage.

$Qp_{forage(deer)}$ = quantity of forage eaten by the wild game animals per day (kg/day). A calculated value of $Qp_{forage(deer)}$ = 1.463 kg/day (using values from Sample et al. 1997 [66.5 kg × 0.022 kg/kg/day], refer to Section 8.1.3.3, mule deer species profile[7]) is used for wild game animals.

$P_{forage}$ = concentration of COPC or ROPC in forage that is ingested by the wild game animals (mg/kg for COPCs and pCi/g for ROPCs). $P_{forage}$ is COPC- and ROPC-specific and calculated as follows:

$$P_{forage} = Pd_{forage} + Pv_{forage} + Pr_{ag(forage)}$$

where, for all constituents except carbon-14 and tritium:

$Pd_{forage}$ is calculated in Eq. 5-14 in the HHRAP.

$Pv_{forage}$ is calculated in Eq. 5-18 in the HHRAP.

$Pr_{ag(forage)}$ is calculated in Eq. 5-20A in the HHRAP using 15 cm soil (root-zone).

[7] Note, the ecological assessment uses fresh weights while the human health assessment uses dry weights for food quantity of forage eaten.

For carbon-14 and tritium, $P_{forage}$ takes on the plant concentration value, $C_{V(C-14)}$ or $C_{V(H-3)}$, calculated from calculated from air concentration as described by the NRC (1977), respectively (see Section 6.6.2).

In the future scenario, $Pd$, $Pv$, $C_{V(C-14)}$, and $C_{V(H-3)}$ are all zero because there are no longer any emissions (no direct deposition or air-to-plant uptake).

$Ba_{deer}$ = biotransfer factor for wild game animals (day/kg). $Ba_{deer}$ is COPC- and ROPC-specific. The biotransfer factor for beef is used as a surrogate biotransfer factor for wild game animals and is shown (as $Ba_{beef}$) in Supplement 4. If no value is available for $Ba_{beef}$, then $A_{game}$ cannot be calculated, and the ingestion of game pathway cannot be evaluated in the HHRA. The values for $Ba_{beef}$ in Supplement 4 will be compared against the calculated mass-limited uptake factor for beef (shown in Appendix A), and the smaller of the two values will be used in the calculation of the wild game concentration ($A_{game}$).

$MF$ = Metabolism factor (unitless). $MF$ is COPC- and ROPC-specific. The recommended default $MF$ values of 0.01 for bis (2-ethylhexyl) phthalate and 1.0 for all other constituents (EPA 2005a) are used (See Table 7-9).

**Exposure Point Concentration in Wild Game Organs**

Guidance in *Application of the CTUIR Traditional Lifeways Exposure Scenario in Hanford Risk Assessments* (Harris 2008) recommends the inclusion of game organs in the diet described by the traditional tribal lifestyle. This reference states that animal organs are eaten, and those organs can have bioconcentrated some contaminants by as much as 10-fold. Therefore, for scenarios where the pathway applies, game organ concentration shall be taken as 10 times the equivalent game meat concentration.

$$A_{game\ organs} = 10 \times A_{game}$$

where:

$A_{game\ organs}$ = concentration of COPC or ROPC in wild game animal organs (mg/kg for COPCs and pCi/g for ROPCs).

$A_{game}$ = concentration of COPC or ROPC in wild game animals (mg/kg for COPCs and pCi/g for ROPCs).

**Feed-to-Animal Tissue Biotransfer Factors: Mass Balance Issues**

The HHRAP recommended sources for uptake factors (*Ba*) for organic chemicals sometimes result in animals predicted to take up more chemical into their tissues than is present in their food.

For example, for n-dioctyl phthalate, using the default uptake factors, more chemical is predicted to accumulate in beef cattle than is available in their feed. Using an assumed soil concentration of 1E-08 mg/kg, the total mass of n-dioctyl phthalate in soil and feed ingested by a steer is 49 mg (calculated as [the sum of concentration of n-dioctyl phthalate in soil and food, such as silage, grain, and forage] × [respective consumption rate of soil and food] × [730 days exposure duration to raise a steer to market weight]). Using the recommended default uptake factor for beef (7.77 kg/day in the HHRAP database), the predicted total mass of n-dioctyl phthalate in the beef is 296 mg (calculated as [the sum of concentration of n-dioctyl phthalate in soil and food, such as silage, grain, and forage] × [respective

consumption rate of soil and food] × [default beef uptake factor for n-dioctyl phthalate] × [567 kg, the average live weight for cattle taken to slaughter]). Thus, for a given concentration of n-dioctyl phthalate in soil and feed, cattle are predicted to take up more than 6 times the amount of n-dioctyl phthalate than is available in the soil and feed that is ingested over a two-year period (i.e., 296 mg in beef/49 mg in feed).

A conservative solution to this mass balance problem is to calculate an uptake factor that allows 100 % of the available chemical to transfer to animal tissue, but no more. This mass-limited uptake factor is not chemical-specific but rather it is a function of exposure duration and body weight. The feed-to-animal tissue mass-limited uptake factor is calculated as:

Feed-to-Animal Tissue Uptake Factor = (Exposure Duration) ÷ (Tissue Weight) (Equation 7-2)

where:

| | | |
|---|---|---|
| Uptake Factor | = | mass-limited feed-to-animal tissue uptake factor (days/kg) |
| Exposure Duration | = | duration to bring animal to market weight (days) |
| Tissue Weight | = | total mass of animal at market weight (kg) |

This mass-limited uptake factor assumes that the animals concentrate the entire mass of chemical ingested into their edible tissue, with no degradation or excretion of the chemical over the exposure duration period. This mass-limited uptake factor can be used to calculate a conservative estimate of potential dose and risk to human receptors without defying the law of conservation of mass.

The equation above is used to estimate mass-limited feed-to-animal tissue uptake factors for beef, pork, and poultry. Estimating a mass-limited feed-to-animal uptake factor for animal products (i.e., milk and eggs) is slightly different. The mass-limited feed-to-animal product uptake factor is a function of the daily product weight for the animal. The equation for the mass-limited feed-to-animal product uptake factor is:

Feed-to-Animal Product Uptake Factor = 1 ÷ (Daily Product Weight) (Equation 7-3)

where:

| | | |
|---|---|---|
| Uptake Factor | = | mass-limited feed-to-animal product uptake factor (days/kg) |
| Daily Product Weight | = | total expected weight of animal product each day (kg/day) |

This equation is used to estimate mass-limited feed-to-animal product uptake factors for milk and eggs. All calculated feed-to-animal tissue/product mass-limited uptake factors are shown in Appendix A, Section A.5.2. The final step in this mass-limited uptake factor approach is to compare the uptake factors as specified in the HHRAP (EPA 2005a) to the calculated mass-limited uptake factors, on a chemical-by-chemical basis for organic COPCs. The lesser of the two values will be used in the estimation of animal tissue/product concentrations.

#### 7.1.7.5 Exposure Point Concentrations in Fish

Exposure point concentrations in fish tissue for the human health evaluation will be modeled as described here. As noted in Section 6.7, this modeling effort is slightly different for the human health and ecological risk assessments. See Section 8 for the modeling required for the ecological risk assessment.

This section describes the models that will be used to calculate fish tissue concentrations and the uptake factors to be used in these models.

The COPCs and ROPCs in fish will be estimated using the equations presented below as recommended by EPA (2005a). The ROPCs will be evaluated using equations similar to those presented for COPCs in EPA (2005a). Values for the chemical-specific parameters are presented in Supplement 4; other parameter values are presented in Table 7-9. It should be noted that the Hanford Surface Environmental Surveillance Program collects and analyzes fish tissues from the Hanford Reach of the Columbia River. However, since the SLRA will be conducted prior to release of emissions from the WTP, the fish data collected does not represent contamination contributed by the WTP and thus cannot be used to calibrate the fish model.

For organic COPCs other than dioxins, furans, and PCBs, where log $K_{ow}$ is less than 4, and all inorganic COPCs and ROPCs with values for BAF, fish concentrations will be estimated as:

COPCs: $C_{fish} = C_{dw} \cdot BCF_{fish}$ (HHRAP Table B-4-26)

$C_{organs} = C_{dw} \cdot BCF_{organs}$

ROPCs: $C_{fish} = CF \cdot C_{dw} \cdot BCF_{fish}$ (HHRAP Table B-4-26)

$C_{organs} = CF \cdot C_{dw} \cdot BCF_{organs}$

where:

$C_{fish}$ = concentration of COPC or ROPC in fish muscle tissue (mg/kg for COPCs and pCi/g for ROPCs).

$C_{organs}$ = concentration of COPC or ROPC in fish organs (mg/kg for COPCs and pCi/g for ROPCs) $C_{dw}$ = dissolved-phase water concentration (mg/L for COPCs or pCi/L for ROPCs) calculated in Table B-4-24 of the HHRAP.

$BCF_{fish}$ = bioconcentration factor for COPCs and ROPCs in fish (L/kg). $BCF_{fish}$ is COPC- and ROPC-specific and is shown in Supplement 4.

$BCF_{organs}$ = organ-specific bioconcentration factor for COPCs and ROPCs in fish (L/kg), if available. $BCF_{organs}$ is COPC- and ROPC-specific and is shown in Supplement 4.

$CF$ = units conversion factor of $10^{-3}$ (kg/g), used for ROPCs only.

For organic COPCs other than dioxins, furans, and PCBs, where log $K_{ow}$ is greater than 4, and all inorganic COPCs and ROPCs with values for BAF, fish concentrations will be estimated as:

COPCs: $C_{fish} = C_{dw} \cdot BAF_{fish}$ (HHRAP Table B-4-27)

$C_{organs} = C_{dw} \cdot BAF_{organs}$

ROPCs: $C_{fish} = CF \cdot C_{dw} \cdot BAF_{fish}$ (HHRAP Table B-4-27)

$C_{organs} = CF \cdot C_{dw} \cdot BAF_{organs}$

where:

$C_{fish}$ = concentration of COPC or ROPC in fish muscle tissue (mg/kg for COPCs and pCi/g for ROPCs).

$C_{organs}$ = concentration of COPC or ROPC in fish organs (mg/kg for COPCs and pCi/g for ROPCs).

$C_{dw}$ = dissolved-phase water concentration (mg/L for COPCs or pCi/L for ROPCs) calculated in Table B-4-24 of the HHRAP.

$BAF_{fish}$ = bioaccumulation factor for COPCs and ROPCs in fish (L/kg). $BAF_{fish}$ is COPC- and ROPC-specific and is shown in Supplement 4.

$BAF_{organs}$ = organ-specific bioaccumulation factor for COPCs and ROPCs in fish (L/kg), if available. $BAF_{organs}$ is COPC- and ROPC-specific and is shown in Supplement 4.

$CF$ = units conversion factor of $10^{-3}$ (kg/g), used for ROPCs only.

Divalent mercury in the fish is assumed to exist or be converted to the methyl mercury (organic) form after uptake into the fish tissue (EPA 2005a). Therefore, the fish concentration of mercury will be calculated using the equation in Table B-4-27 of the HHRAP:

$$C_{fish_{MHg}} = C_{organs_{MHg}} = C_{dw_{MHg}} \cdot BAF_{fish_{MHg}}$$

$$C_{fish_{Hg^{2+}}} = C_{organs_{Hg^{2+}}} = C_{dw_{Hg^{2+}}} \cdot BAF_{fish_{Hg^{2+}}}$$

From HHRAP Table B-4-27, all divalent mercury in fish exists or is converted to the methyl mercury (organic) form after uptake into the fish tissue.

$$Hg^{2+} + CH_3^- \rightarrow CH_3Hg^+$$

$$C_{fish_{Total\ MHg}} = C_{fish_{MHg}} + \frac{MW_{MHg}}{MW_{Hg^{2+}}} \cdot C_{fish_{Hg^{2+}}}$$

$$C_{organs_{Total\ MHg}} = C_{organs_{MHg}} + \frac{MW_{MHg}}{MW_{Hg^{2+}}} \cdot C_{organs_{Hg^{2+}}}$$

Where

$C_{fish_{Total\ MHg}}$ = the total concentration of mercury in fish meat (in the form of methyl mercury) (mg/kg)

$C_{organs_{Total\ MHg}}$ = the total concentration of mercury in fish organs (in the form of methyl mercury) (mg/kg)

$MW_{MHg}$ = molecular weight of methyl mercury (215.62 g/mol)

$MW_{Hg2+}$ = molecular weight of divalent mercury (200.59 g/mol)

Other variables ($C_{dw}$, $BAF$) are as defined above, but are specific to methyl mercury (MHg) and divalent mercury ($Hg^{2+}$).

For dioxins, furans, and PCBs, fish concentrations will be estimated from sediment concentrations and *BSAF* values using the following equation:

$$C_{fish} = \frac{C_{sed} \cdot f_{lipid} \cdot BSAF_{fish}}{OC_{sed}} \qquad \text{(HHRAP Table B-4-28)}$$

where:

| | | |
|---|---|---|
| $C_{fish}$ | = | concentration of COPC in fish (mg/kg). |
| $C_{sed}$ | = | COPC concentration in bed sediment (mg/kg) calculated in Table B-4-25 of the HHRAP. |
| $f_{lipid}$ | = | fish lipid content (unitless). The recommended default value of 0.07 (EPA 2005a) is used for $f_{lipid}$ (see Table 7-9). |
| $BSAF_{fish}$ | = | biota-to-sediment accumulation factor (unitless) for fish. $BSAF_{fish}$ is COPC-specific and is shown in Supplement 4. |
| $OC_{sed}$ | = | fraction of organic carbon in bottom sediment (unitless). The recommended default value of 0.04 (EPA 2005a) is used for $OC_{sed}$ (see Table 7-9). |

**Fish Uptake Factors for Human Health Risk Assessment**

In order to estimate fish concentrations from surface water or sediment concentrations, uptake factors are needed. As discussed in the HHRAP (EPA 2005a), three types of uptake factors are used:

- Bioconcentration factors (*BCF*s)
- Bioaccumulation factors (*BAF*s)
- Biota-sediment accumulation factors (*BSAF*s)

Per the HHRAP, for compounds with log $K_{ow}$ less than 4.0, *BCF*s are used to estimate fish concentrations from surface water concentrations. For COPCs with log $K_{ow}$ greater than 4.0, except for extremely hydrophobic compounds (such as, dioxins, furans, and PCBs), *BAF*s are used to estimate fish concentrations from surface water concentrations. Since extremely hydrophobic compounds have a high tendency to bioaccumulate, they are expected to be sorbed to the bed sediments more than being associated with the water phase. Therefore, *BSAF*s are used to estimate fish concentrations from sediment concentrations for dioxins, furans, and PCBs.

The first source of values for *BCF*s, *BAF*s, and *BSAF*s is the HHRAP (EPA 2005a). For values not available in the HHRAP, a literature search (including the SLERAP [EPA 1999b]) was conducted. For values not available in literature, the approaches shown below were used to estimate fish uptake factors (*BCF*s, *BAF*s, and *BSAF*s). The final uptake factors collected or calculated from these sources are provided in Supplement 4. Where organ-specific *BCF*s, *BAF*s, and *BSAF*s were available, they were used to determine fish organ concentrations for use in the assessment of tribal exposures as applicable.

For organic COPCs where published *BCF*s are not available and where log $K_{ow}$ is less than 4.0, *BCF*s are calculated using the following equations from the HHRAP (EPA 2005a):

| Log $K_{ow}$ value | Equation | HHRAP equation number |
|---|---|---|
| Nonionic species | | |
| < 1 | log *BCF* = 0.50 | A-2-27 |
| 1 to 7 | log *BCF* = 0.77 log $K_{ow}$ - 0.70 + Σ correction factors | A-2-28 |
| 7 to 10.5 | log *BCF* = -1.37 log $K_{ow}$ + 14.4 + Σ correction factors | A-2-29 |
| > 10.5 | log *BCF* = 0.50 | A-2-30 |
| Ionic species (carboxylic acids, sulfonic acids and salts, compounds with N of +5 valence) | | |
| < 5 | log *BCF* = 0.50 | A-2-31 |
| 5 to 6 | log *BCF* = 0.75 | A-2-32 |
| 6 to 7 | log *BCF* = 1.75 | A-2-33 |
| 7 to 9 | log *BCF* = 1.00 | A-2-34 |
| > 9 | log *BCF* = 0.50 | A-2-35 |

For organic COPCs that are not dioxins, furans, or PCBs, where published *BAF*s are not available and where log $K_{ow}$ is greater than 4.0, the following approach is used to obtain *BAF*s:

1) Calculate an estimate of *BCF* according to the appropriate HHRAP Appendix A equation.
2) Obtain food chain multipliers (*FCM*s) for Trophic Level 3 and 4 fish.
3) Estimate the *BAF* using the following equation, from the SLERAP (EPA 1999b):

$$BAF_{fish} = BCF_{fish} \cdot FCM \qquad \text{(SLERAP [EPA 1999b] Eq. 5-10)}$$

where *FCM* is the largest *FCM* when considering *FCM*s for Trophic Level 3 and 4 fish.

For dioxins (PCDDs) and furans (PCDFs) where published *BSAF*s are not available, the approach shown in the HHRAP Appendix A, Section A2-2.13.4.3, will be used to obtain *BSAF*s:

- For TetraCDDs and TetraCDFs, $BSAF_{fish} = 9.0 \times 10^{-2}$
- For PentaCDDs and PentaCDFs, $BSAF_{fish} = 9.0 \times 10^{-2}$
- For HexaCDDs and HexaCDFs, $BSAF_{fish} = 4.0 \times 10^{-2}$
- For HeptaCDDs and HeptaCDFs, $BSAF_{fish} = 5.0 \times 10^{-3}$
- For OctaCDDs and OctaCDFs, $BSAF_{fish} = 1.0 \times 10^{-4}$

Empirical fish *BSAF* values are available from the US Army Corps of Engineers *BSAF* database (USACE 2010). Conservatively, the maximum reported *BSAF* will be used for the initial assessment. Should a problem be indicated, average *BSAF*s or *BSAF*s more appropriate to the fish consumed by humans will be used to assess potential risks.

## 7.2 Toxicity Assessment

The purpose of the toxicity assessment is to evaluate the potential for COPCs and ROPCs to cause adverse health effects in exposed individuals. Toxic effects have been evaluated extensively by the EPA. This section provides the results of the EPA evaluation of the COPCs and ROPCs that may be emitted by the WTP.

### 7.2.1 General Toxicity Information and EPA Guidance for COPCs

This section provides the toxicity values that will be used for evaluating COPCs in the PRA and the source/rationale for these values.

#### 7.2.1.1 Chronic Toxicity of COPCs

Chronic toxicity data have generally been obtained from the *Regional Screening Levels (RSL) for Chemical Contaminants at Superfund Sites* (EPA 2013, as amended). The RSL table is a living document that reflects the current state of the science of toxicology and risk assessment, with case-by-case exceptions as approved by Ecology.

When toxicity values for a chemical are not available from the RSLs, the use of a surrogate value may be necessary. This process involves applying a toxicity value established for one chemical to another chemical for which no value has been established. The application of surrogate values is based on similarities in structure, mechanism of action, and toxicity. Surrogate values for the SLRA are identified by Ecology and EPA Region 10, in consultation with National Center for Environmental Assessment (NCEA) (CCN 064330, CCN 063814, CCN 063802, CCN 063817, CCN 063818, CCN 063812, and CCN 063803).

Chronic toxicity values from these sources are provided in Supplement 4 and described below. Supplement 4 provides the toxicity value, its source, and whether the value has been extrapolated from another exposure pathway (i.e., oral to dermal). The same approach will be used for the toxicity assessment in both the PRA and FRA. Any new toxicity values that become available prior to development of the FRA will be incorporated in the final assessment.

**Chronic Noncarcinogenic Toxicity of COPCs**

Oral noncarcinogenic effects of COPCs will be evaluated by comparing a calculated intake or dose with an acceptable daily intake criterion (referred to as the *reference dose* [*RfD*]) established by EPA (1997b, 2004). The effects due to inhalation of noncarcinogenic of COPCs will be evaluated by comparing a calculated exposure concentration with an inhalation reference value (referred to as the reference concentration [*RfC*]).

It is widely accepted that most biological effects of chemicals occur only after a threshold dose is exceeded (Klaassen et al. 1996). For purposes of establishing noncarcinogenic health criteria, this threshold dose is usually estimated from the no observed adverse effect level (*NOAEL*) or lowest observed adverse effect level (*LOAEL*) determined from animal or human studies. The *NOAEL* is defined as the exposure level at which no statistically or biologically significant increases are present in the frequency or severity of adverse effects (EPA 1989). The *LOAEL* is the lowest exposure level at which there are statistically or biologically significant increases in frequency or severity of adverse effects (EPA 1989). The *LOAEL* or *NOAEL* from the most sensitive animal or human study is used by the EPA

to establish long-term health criteria. An *RfD* is defined as an estimate (with uncertainty spanning perhaps an order of magnitude) of the dose of a chemical (expressed in mg/kg·day) that is likely to be without an appreciable risk of deleterious effects during a lifetime (EPA 1989). Similarly, a reference concentration (*RfC*) represents the concentration of a chemical in air (expressed as $mg/m^3$) that is likely to be without an appreciable risk of deleterious effects during a lifetime (EPA 1989). When deriving *RfD*s or *RfC*s, a *NOAEL* value is used preferentially over a *LOAEL* value if both are available from the key study. EPA derives *RfD*s and *RfC*s by applying uncertainty factors to the *NOAEL* or *LOAEL* value to provide a margin of safety. The equation for deriving an *RfD* or *RfC* is shown below:

$$RfD \text{ or } RfC = (NOAEL \text{ or } LOAEL)/(UF \times MF) \qquad \text{(EPA 1989, Sect. 7.7.2, and EPA 2009, Eq. 5)}$$

where:

| | | |
|---|---|---|
| *RfD* | = | reference dose (mg/kg·day) |
| *RfC* | = | reference concentration ($mg/m^3$) |
| *NOAEL* | = | no observed adverse effect level (mg/kg·day or $mg/m^3$) |
| *LOAEL* | = | lowest observed adverse effect level (mg/kg·day or $mg/m^3$) |
| *UF* | = | uncertainty factor (unitless) |
| *MF* | = | modifying factor (unitless) |

Uncertainty factors can range from 1 to 10,000 and may include a factor of up to 10 to account for each of the following:

- Variation in sensitivity within human populations
- Extrapolation of effects observed in animals to humans
- Extrapolation from less-than-lifetime exposures in the critical study to lifetime exposures
- Extrapolation from a *LOAEL* to a *NOAEL*, if necessary

In some cases a modifying factor, usually ranging from 1 to 10 (or <1 for most essential nutrients [EPA 1989]), also is applied to the *NOAEL* or *LOAEL*. This value reflects a qualitative professional assessment of additional uncertainties in the critical study and in the entire database for the chemical not explicitly addressed by the above uncertainty factors (EPA 1989). The EPA establishes *RfD*s and *RfC*s for evaluating both subchronic (less than 7 years) and chronic (7 years or more) exposures. Chronic *RfD*s will be used to evaluate all exposure scenarios, except the acute scenario, and are presented in Supplement 4.

**Carcinogenic Toxicity of COPCs**

The health risk from exposure to a carcinogen is defined in terms of probability. This probability is defined as the likelihood of a carcinogenic response in an individual that receives a given dose of a particular compound. Oral cancer risks are estimated using chemical-specific cancer slope factors (*CSF*s). For chemicals, the *CSF* is defined as a plausible upper-bound estimate of the probability of a response (e.g., cancer) per unit intake of a chemical over a lifetime (EPA 1989). A *CSF* is provided for potentially carcinogenic COPCs in Supplement 4.

In addition to the quantitative *CSF*, a qualitative weight-of-evidence classification is assigned to characterize the quality and quantity of data used to evaluate the potential carcinogenicity of chemicals.

These classifications are provided in the Integrated Risk Information System (IRIS). As defined by EPA (1989), chemicals used to be assigned to any of six weight-of-evidence groups:

- Group A - Human carcinogen (sufficient evidence of carcinogenicity in humans)
- Group B1 - Probable human carcinogen (limited evidence of carcinogenicity in humans)
- Group B2 - Probable human carcinogen (sufficient evidence of carcinogenicity in animals, with inadequate or lack of evidence in humans)
- Group C - Possible human carcinogen (limited evidence of carcinogenicity in animals, or lack of human data)
- Group D - Not classifiable as to human carcinogenicity (inadequate or no evidence)
- Group E - Evidence of noncarcinogenicity for humans (no evidence of carcinogenicity in adequate studies)

Consistent with EPA guidance (EPA 1989), chemicals assigned a weight-of-evidence classification of A, B1, or B2 are quantitatively evaluated for carcinogenic dose-response. All Group C carcinogens are also quantitatively evaluated for carcinogenic effects.

As indicated in the *Guidelines for Carcinogen Risk Assessment* (2005c), EPA has moved away from this approach for carcinogen assessments. Early-life (childhood) exposure to carcinogens has also been evaluated by EPA as reflected in *Supplemental Guidance for Assessing Cancer Susceptibility from Early-Life Exposure* (2005d). In addition to the weight-of-evidence classifications published in IRIS, more recent EPA guidance (2005c) recommends the use of qualitative standard descriptors as part of the narrative to express conclusions about the weight of evidence for human carcinogenic potential. The EPA (2005c) defines five descriptors, which are roughly equivalent to the weight-of-evidence classifications provided by IRIS. More than one descriptor may be applicable for a single chemical (e.g., if it is likely to be carcinogenic by one route of exposure but not by others). The five descriptors are:

1. Carcinogenic to humans
2. Likely to be carcinogenic to humans
3. Suggestive evidence of carcinogenicity, but not sufficient to assess human carcinogenic potential
4. Data are inadequate for an assessment of human carcinogenic potential
5. Not likely to be carcinogenic to humans

The descriptors will be used in the PRA and the FRA as part of the risk characterization presentations for specific chemicals that may be risk drivers.

The EPA sometimes reports cancer potency as a unit risk factor (*URF*) based on chemical concentration in air or drinking water. In general, the drinking-water unit risk is derived by converting a slope factor from units of mg/kg·day to units of g/L, whereas an inhalation unit risk is developed directly from a dose response analysis using equivalent human concentrations already expressed in units of g/m$^3$ (EPA 2005c, Section 3.3.3). Oral *CSF*s are calculated from the corresponding *URF* values, when necessary, using the following equation:

$$CSF_{oral} = (URF_{oral} \times BW \times CF) / CR_{dw}$$ (EPA 1989, Sect. 7.3.3, modified)

where:

$CSF_{oral}$ = chemical-specific oral *CSF* $(mg/kg \cdot day)^{-1}$
$URF_{oral}$ = chemical-specific drinking water unit risk factor (*URF*) $(\mu g/L)^{-1}$
*BW* = default body weight (70 kg)
*CF* = conversion factor (1000 µg/mg)
$CR_{dw}$ = default drinking water consumption rate (2 L/day)

Expression of the drinking water *URF* in terms of dose is necessary to evaluate cancer risk associated with exposure media other than drinking water (such as soil). The EPA recognizes the need for expressing oral toxicity values in terms of dose (mg/kg·day) for risk assessment purposes and acknowledges that, in many cases, this conversion does not add significant uncertainty to the risk assessment process (EPA 1997b).

The interaction of the inhaled contaminant with the respiratory tract is affected by factors such as species-specific relationships of exposure concentrations (*EC*s) to deposited/delivered doses and physiochemical characteristics of the inhaled contaminant. The EPA (2009) therefore recommends that when estimating risk via inhalation, risk assessors should use the concentration of the chemical in air as the exposure metric (e.g., $mg/m^3$), rather than inhalation intake of a contaminant in air based on inhalation rate and body weight (e.g., mg/kg·day). Consequently, the equivalent derivation of an inhalation *CSF* from the *URF* is not generally performed unless the respiratory deposition and absorption characteristics of the constituent are known (refer to Section 7.5.2).

Chemicals that have been determined to cause cancer by a mutagenic mode of action are thought to pose a higher risk during early life and it is possible that exposures to such chemicals in early-life may result in higher lifetime cancer risks than a comparable duration adult exposure. If a mutagenic mode of action for carcinogenicity of a constituent has been determined by EPA, and a linear low-dose extrapolation performed, one of the following generally pertains:

1. If chemical-specific data on susceptibility from early-life exposures were available for derivation of *CSF*s, those slope factors are used for risk characterization, and age dependent adjustment factors (*ADAF*s) are not applied.
2. If chemical-specific data on susceptibility from early-life exposures were not available, the *ADAF*s are applied in calculating or estimating risks associated with early-life exposures (EPA 2005d).

If the latter case applies, the *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (EPA 2005d) recommends default *ADAF*s be applied in risk assessments for the assessment of chemicals that cause cancer by a mutagenic mode of action as detailed in Section 7.2.1.4.

**Chronic Dermal Toxicity of COPCs**

Oral *RfD*s and *CSF*s are currently available for many of the COPCs. Dermal *RfD*s and *CSF*s are estimated for COPCs from oral toxicity values using chemical-specific gastrointestinal absorption factors (*GAF*s) to calculate total absorbed dose. This conversion is necessary because most oral *RfD*s and *CSF*s are expressed as the amount of chemical *administered* per time and body weight; however, dermal exposure is expressed as an *absorbed* dose. Dermal toxicity factors are calculated from oral toxicity factors as shown below (EPA 2004):

$$RfD_{dermal} = RfD_{oral} \times GAF \qquad \text{(EPA 2004, Eq. 4.3)}$$

$$CSF_{dermal} = CSF_{oral} \div GAF \qquad \text{(EPA 2004, Eq. 4.2)}$$

Chemical-specific *GAF* values are used when available. Not all COPCs have chemical-specific *GAF* values. When quantitative data were not available, default *GAF* values of 0.8 for VOCs, 0.5 for SVOCs, and 0.2 for inorganics are used (Ecology 2002). The *GAF* values are provided in Supplement 4 along with the resulting dermal *RfD* and *CSF*s.

#### 7.2.1.2 Acute Toxicity of COPCs

Acute effects from direct inhalation of airborne COPCs (vapor and particulate) are evaluated by comparison of modeled one-hour maximum air concentrations to AIEC. The AIEC values for COPCs were selected based on the following hierarchy:

1. **Values from the NCEA** (as provided by EPA Region 10).
2. **Acute reference exposure levels (ARELs) from California EPA**. The AREL is an exposure that is not likely to cause adverse effects in a human population, including sensitive subgroups, exposed to that concentration for one hour on an intermittent basis. The ARELs are based on the most sensitive, relevant, adverse health effect reported in the medical and toxicological literature. The ARELs are designed to protect the most sensitive individuals in the population by the inclusion of margins of safety. Since margins of safety are incorporated to address data gaps and uncertainties, exceeding the AREL does not automatically indicate an adverse health impact.
3. **Acute exposure guideline levels (AEGL-1)**. If an AEGL-1 value is not available but an AEGL-2 value is available, the AEGL-2 value will be used unless a more conservative value is available from one of the other sources in the hierarchy. The one-hour AEGLs are used. The AEGL-1 is the airborne concentration of a substance above which it is predicted that the general population, including susceptible individuals, could experience notable discomfort, irritation, or certain asymptomatic, non-sensory effects. However, the effects are not disabling and are transient and reversible upon cessation of exposure. The AEGL-2 is the airborne concentration of a substance above which it is predicted that the general population, including susceptible individuals, could experience irreversible or other serious, long-lasting adverse health effects, or an impaired ability to escape.

   Airborne concentrations below the AEGL-1 represent exposure levels that could produce mild and progressively increasing odor, taste, and sensory irritation, or certain non-symptomatic, non-sensory effects. With increasing airborne concentrations above each AEGL level, there is a progressive increase in the likelihood of occurrence and the severity of effects described for each corresponding AEGL level. Although the AEGL values represent threshold levels for the general public, including sensitive subpopulations, it is recognized that certain individuals, subject to unique or idiosyncratic responses, could experience the effects described at concentrations below the corresponding AEGL level. **Note**: This description is from "National Advisory Committee for Acute Exposure Guideline Levels (AEGLs) for Hazardous Substances; Proposed AEGL Values," *Federal Register*, 18 July 2003 (Volume 68, Number 138), pages 42710-42726.
4. **Emergency response planning guidelines (ERPG-1)**. The ERPG-1 are the maximum concentration in air below which it is believed nearly all individuals could be exposed for up to one hour without experiencing other than mild transient adverse health effects or perceiving a clearly defined objectionable odor. Safety factors are not included.

5 **Temporary emergency exposure limits (TEEL-1)**. The TEELs are temporary levels of concern similar to ERPGs, and defined by the US Department of Energy for use when ERPGs are not available. As with ERPGs, safety factors are not included.

The AIEC values selected using this hierarchy are provided in Supplement 4 along with their basis. Only one NCEA provisional value (for PCBs) is used. The ARELs from California EPA include potential effects of intermittent acute exposures. AEGL-1, AEGL-2, and ERPG-1 values assume one-time-only acute exposures and are available in units of parts per million (ppm). Some TEELs are provided in ppm and some in $mg/m^3$. Values are provided in their original units, along with conversion factors, in Supplement 4. The use of values obtained other than NCEA values or California EPA ARELs will be discussed as a nonconservative uncertainty in the uncertainty section of the risk assessment.

#### 7.2.1.3 Toxicity of COPCs to Nursing Infant

Potential infant exposures to PCDD/PCDFs and coplanar PCBs in human breast milk will be evaluated in the SLRA. The interpretation of infant exposure is limited by the lack of infant dose-response data. The EPA (2005a) recommends evaluating infant exposures to dioxins in breast milk by comparing a site-specific calculated dose to the infant ($ADD_{inf}$) to a background dose to the infant.

A background infant *ADD* of 93 pg/kg·day of PCDD/PCDFs and co-planar, dioxin-like PCBs in breast milk as 2,3,7,8-tetrachlorodibenzo-p-dioxin toxic equivalents (TEQ), has been calculated by the EPA based on an average background 2,3,7,8-TCDD TEQ concentration of 25 parts per trillion (ppt) measured in breast milk. The 25 ppt 2,3,7,8-TCDD TEQ is the sum of the average breast milk concentration of 18 ppt TEQ from PCDD/PCDFs and 7-ppt TEQ from co-planar, dioxin-like PCBs. After normalization for infant body weight, this breast milk concentration of 25 ppt TEQ results in an average, background intake for the infant, $ADI_{b\text{-}inf}$, of 93 picograms per kilogram per day (pg/kg·day) of 2,3,7,8-TCDD TEQ (EPA 2005a, Section 2.3.10.2). Based on the national average background exposure level of 60 pg TEQ/kg/day of PCDD/PCDFs for nursing infants reported by EPA (2005a), 33 pg TEQ/kg/day is attributable to background exposure levels of co-planar, dioxin-like PCBs.

This background approach will also be used for evaluating potential risks to the nursing infant for exposure to “dioxin-like” coplanar PCBs. The estimated dose (as 2,3,7,8-TCDD equivalents) of coplanar PCBs will be compared to a background infant dose of 33 pg TEQ/kg·day. In addition to evaluating dioxin and PCB exposures separately, a total infant dose of dioxin-like compounds (PCDDs/PCDFs and coplanar PCBs expressed as 2,3,7,8-TCDD equivalents) will be calculated and compared to a total background dose of dioxin-like compounds of 93 pg TEQ/kg·day. This background dose may overestimate current exposures because dioxin exposures have been decreasing. The source of this value and potential range of background doses will be discussed further in the uncertainty assessment of the PRA report.

This approach is based on the assumption that, if the estimated dose to a nursing infant from site-related dioxins is below the nationwide background dose of dioxins to nursing infants, the site-related risk of cancer or noncancer effects is not significant.

In discussing infant exposure to background concentrations of dioxins, EPA (2003) notes that “breast-feeding infants have higher intakes of dioxin and related compounds for a short but developmentally important part of their lives. However, the benefits of breast feeding are widely recognized to outweigh the risks.”

Although background intakes of dioxins by nursing infants (60 pg TEQ/kg·day, EPA 2005a) are relatively high compared to adult intakes (1 pg TEQ/kg·day, EPA 2005a), the body burden of nursing infants is only about two times that of adults, and the contribution of infant exposure to eventual adult body burden is small. The reduced body burden in nursing infants (relative to intake) may be due to the rapid growth of the infant and a faster elimination/excretion rate in infants.

Transplacental transfer of dioxins from the mother to the fetus may also be a significant source of exposure. Dioxins may produce a broad range of effects in experimental animals exposed in-utero, and limited epidemiological studies have been conducted (EPA 2005c). Potential effects (cancer or noncancer, including developmental effects) of prenatal exposures are not included in the quantitative evaluation of risk.

There is currently no consensus regarding the most appropriate single approach to quantitatively evaluate potential risks associated with exposure to dioxin-like compounds by nursing infants. Alternative approaches to the two methods described above (i.e., comparison to background and lifetime risk) include calculating infant risks using (1) the infant ADD calculated in accordance with HHRAP, Table C-3-2, and (2) the lifetime risk calculated in accordance with *Supplemental Guidance for Assessing Cancer Susceptibility from Early-Life Exposure to Carcinogens* (EPA 2005d). These alternative methods will be presented in the uncertainty assessment of the PRA report.

#### 7.2.1.4 Toxicity of Mutagens

*Supplemental Guidance for Assessing Cancer Susceptibility from Early-Life Exposure to Carcinogens* (EPA 2005d) provides draft EPA guidance for evaluating early-life exposures to carcinogens. This guidance recommends that when developing quantitative estimates of cancer risk, age-specific values for both exposure and toxicity/potency should be integrated where such data are available and appropriate, specifically:

- Early life exposures to carcinogens may have a larger or smaller impact on lifetime cancer risk than later exposures, even if the total lifetime exposure is the same.
- Exposures near the end of life may have little effect on lifetime cancer risk.

EPA 2005d recommends calculating a combined lifetime risk rather than separate infant, child, and adult risks with specific adjustments based upon increased susceptibility of younger receptor for mutagenic compounds. If the incremental lifetime cancer risk (ILCR) associated with mutagenic COPCs[8] exceeds $10^{-7}$ then the risk due to exposure to these COPCs will be further assessed with consideration for age adjustment factors as described below. For mutagenic chemicals, early life exposures have a larger impact than later exposures on lifetime risk. This impact can be quantified using an age dependent adjustment factor (*ADAF*) to make the following adjustments (EPA 2005d):

- For exposures before 2 years of age, a 10-fold adjustment (*ADAF* = 10)
- For exposures between 2 and 16 years of age, a 3-fold adjustment (*ADAF* = 3)
- For exposures after 16 years of age, no adjustment (*ADAF* = 1)

[8] Mutagenic COPCs are those that are published in Table 1b of EPA 2005d or EPA the RSL Tables (EPA 2013, or most recent update) , and are subject to periodic updating by EPA. Such mutagenic COPCs are identified in Supplement 4.

However, for this risk assessment, the division between infant and children exposures, and children and adult exposures does not occur at ages 2 and 16. Depending on the receptor, the nursing infant is presumed to be exposed (to breast milk) from ages 0 to age 1 or 2. Childhood exposures are assumed to occur over the first 6 years of life, and adult exposures are considered to apply to all receptors after age 6. A sensitivity analysis on exposure assumptions for the *ADAF* age bins indicated that it there is little difference in the outcome if they are not changed from the standard assumptions, but they should be broken into the following: age 0 to <2 yr (*ADAF*=10), age 2 to <6 yr (*ADAF*=3), age 6 to <16 yr (*ADAF*=3), and older than age 16 (*ADAF*=1). The infant exposure takes place over 1 or 2 years (depending on the receptor), the child is exposed from age 2 (or 3) through age 6, and the adult is exposed from age 7 through age 70. The EPA addresses the discrepancy between the age division common to risk assessments and the application of *ADAF*s in the electronic *Handbook for Implementing the Supplemental Cancer Guidance at Waste and Cleanup Sites* (EPA 2012b).

The EPA *ADAF*s are prorated according the age divisions to derive an *ADAF* appropriate to the receptors in this RAWP by modifying the exposure duration (*ED*) according to the age bins. The corresponding cancer risk for each age interval "i" takes the following form.

$$Risk_i = C \frac{CR_i \cdot EF_i \cdot ED_i}{BW_i \cdot AT_C} \cdot CSF \cdot ADAF_i \qquad \text{(EPA 2012b)}$$

where:

- $C$ = concentration of the chemical in the contaminated environmental medium (soil or water) to which the person is exposed (mg/kg or mg/L).
- $CR_i$ = consumption or intake rate of the contaminated environmental medium for age bin *i* (mg/day or L/day).
- $BW_i$ = body weight of the exposed person for age bin *i* (kg).
- $EF_i$ = exposure frequency for age bin *i* (days/year)
- $ED_i$ = exposure duration for age bin *i* (years)
- $AT_C$ = averaging time (yr)
- $CSF$ = cancer slope factor $(mg/kg \cdot day)^{-1}$
- $ADAF_i$= age dependent adjustment factor for cancer slope factor for mutagenic chemicals for age bin *i* (unitless)

The risk assessment age divisions are shown with the *ADAF* age intervals below (EPA 2012b):

| Receptor | Exposure Duration (*ED*, years) | Age (years) | Exposure Factors | Exposure Duration ($ED_i$, years) | ADAF (unitless) |
|---|---|---|---|---|---|
| Child | 6 | 0 to <2 | Child | 2 | 10 |
| | | 2 to <6 | Child | 4 | 3 |
| Adult | varies, 30 to 70 | 6 to <16 | Adult | 10 | 3 |
| | | ≥16 | Adult | *ED* - 16 | 1 |

Total risk to the individual is the sum of the risks across all four age intervals. If exposure occurs across multiple pathways (e.g., ingestion, dermal absorption and inhalation), risks are also summed across pathways.

$$Risk_{0\,to<2} = C\frac{CR_{child}\cdot EF_{child}\cdot 2yr}{BW_{child}\cdot AT_C}\cdot CSF\cdot 10 \qquad Risk_{2\,to<6} = C\frac{CR_{child}\cdot EF_{child}\cdot 4yr}{BW_{child}\cdot AT_C}\cdot CSF\cdot 3$$

$$Risk_{6\,to<16} = C\frac{CR_{adult}\cdot EF_{adult}\cdot 10yr}{BW_{adult}\cdot AT_C}\cdot CSF\cdot 3 \qquad Risk_{\geq 16} = C\frac{CR_{adult}\cdot EF_{adult}\cdot (ED-16)yr}{BW_{adult}\cdot AT_C}\cdot CSF\cdot 1$$

and

$$Risk_{lifetime} = Risk_{0\,to<2} + Risk_{2\,to<6} + Risk_{6\,to<16} + Risk_{\geq 16}$$

The equations above are generalized for intake and the quantity $(C\times CR_i\times EF_i\times ED_i)/(BW_i\times AT_C)$ differs from the *LADD* shown in Section 7.1.5 by the value used for exposure duration, $ED_i$. Dermal absorption (which contributes to the *LADD*) and inhalation (which contributes to the *EC*) are also valid pathways. Algebraically, the term $(C\times CR_i\times EF_i\times ED_i)/(BW_i\times AT_C)$ equates to the *LADD* times the ratio of the age interval exposure duration, $ED_i$, and the receptor's exposure duration, *ED*. The same is true with regard to applying the *ADAF* to inhalation exposures. Accordingly, the equations for exposures can be written as:

ingestion and dermal absorption exposures:

$$Risk_i = LADD\cdot\frac{ED_i}{ED}\cdot CSF\cdot ADAF_i$$

and for inhalation exposures:

$$Risk_i = EC\cdot\frac{ED_i}{ED}\cdot URF\cdot ADAF_i$$

where:

- *LADD* = lifetime average daily for the receptor's total exposure period (mg/kg·day)
- *EC* = exposure concentration (µg/m$^3$)
- $ED_i$ = exposure duration for age bin *i* (years)
- *ED* = exposure duration for the receptor (number of years that the receptor is exposed to the COPC)
- *CSF* = oral or dermal cancer slope factor, as appropriate (kg·day/mg)
- *URF* = inhalation unit risk factor (m$^3$/µg)
- $ADAF_i$ = age dependent adjustment factor for cancer slope factor for mutagenic chemicals for age bin *i* (unitless)

Benzo[a]pyrene is often used as an index chemical when assessing other carcinogenic PAHs as described in the "*Provisional Guidance for Quantitative Risk Assessment of Polycyclic Aromatic Hydrocarbons*" (EPA 1993b). The EPA recommends that when assessing early-life exposure for PAHs using such an approach, the ADAF(s) should be applied to the benzo[a]pyrene slope factor before using relative potency factors to estimate risk from exposure to other PAHs (EPA 2006).

For carcinogens that act by mechanisms other than mutagenicity, early life exposure may have a larger, smaller, or no impact on lifetime cancer risk. This impact would be chemical- or mechanism-specific and cannot be quantified at this time; therefore, no adjustment factor is recommended. The potential impact of exposures near the end of life also cannot be quantified.

Radionuclides are mutagens; however, slope factors for radionuclides sufficiently consider age factors. These adjustment factors will not be used in calculating lifetime risks for nursing infants exposed to ROPCs. The PCDDs/PCDFs and PCBs are not mutagens; therefore, lifetime risk for these compounds will be calculated with no adjustment to the *CSF*.

### 7.2.2 Toxicity Information and EPA Guidance for Specific COPCs

The toxicity assessments for several COPCs and classes of COPCs with unique toxicity characteristics or methods for assessment are described below.

#### 7.2.2.1 Chromium

Hexavalent chromium ($Cr^{+6}$) is the most toxic valence state of chromium and has been shown to be a human carcinogen through inhalation. Trivalent chromium ($Cr^{+3}$) has not been shown to be carcinogenic in either humans or laboratory animals; however, the mechanism of $Cr^{+6}$ carcinogenicity in the lung is believed to be its reduction to $Cr^{+3}$ and its generation of reactive intermediates (Klaassen et al. 1996). While chromium emitted from the melter is not likely to be in the hexavalent form, the PRA will conservatively assume that 100 % of the facility emissions are hexavalent chromium for the carcinogen assessment and 100 % trivalent chromium for the noncarcinogen assessment. For the FRA, the same assumptions will be made unless WTP performance demonstration test data for this compound is available to provide more realistic estimates.

#### 7.2.2.2 Nickel

The EPA (2005a) recommends that nickel be evaluated as an inhalation carcinogen because some forms of nickel, including nickel carbonyl, nickel subsulfide, and nickel refinery dust, are considered to be carcinogens. Nickel emissions from hazardous waste combustion units are emitted as nickel oxide which, by itself, is not considered to be a carcinogen; however, nickel oxides can be reduced to nickel sulfates (some of which are carcinogenic) in the presence of sulfuric acid (EPA 2005a). In addition, nickel oxide is a major component of nickel refinery dust (other major components include nickel subsulfide and nickel sulfide), which is identified as a potential human inhalation carcinogen. The components responsible for the carcinogenicity of nickel refinery dust have not been conclusively established. Therefore, nickel emissions are evaluated as a potential carcinogen through the inhalation pathway using the inhalation *URF* for nickel refinery dust. For exposure pathways other than inhalation, nickel has not been shown to be carcinogenic and will be evaluated as a noncarcinogen using the oral *RfD* for nickel-soluble salts.

#### 7.2.2.3 Particulates

Toxicity values (i.e., *RfC*s and *URF*s) are not available to quantitatively evaluate potential adverse health effects associated with inhaling particulates. Therefore, modeled annual average concentrations of respirable particulates will be compared with the following National Ambient Air Quality Standard (NAAQS) values:

| Particle Diameter | NAAQs value[a] |
|---|---|
| < 10 μm ($PM_{10}$) | 50 μg/m$^3$ |
| < 2.5 μm ($PM_{2.5}$) | 15 μg/m$^3$ |

[a]Values are for annual average concentrations.

For air modeling purposes, it is assumed that all particulates released from the facility will have a diameter of 1 μm or 2.5 μm; therefore, the $PM_{2.5}$ standard will be used for comparison to predicted air concentrations.

#### 7.2.2.4 Trichloroethylene

Independently of any carcinogenic and/or mutagenic effects, trichloroethylene (TCE) may cause fetal cardiac malformations when a mother is exposed to TCE during a 21-day early gestation window. Region 10 human health toxicologists have determined that, to protect against potential noncancer fetal malformation outcomes, that average exposures over any 21-day period of time not exceed the concentrations in air or other media that are calculated to be protective for this exposure using the *RfD* and *RfC* provided in IRIS (EPA 2012c). If TCE exposures approach levels that would cause concern as discussed in the EPA 2012 memorandum, TCE will be further evaluated to assess its potential to present a risk of fetal cardiac malformations during a short window during early pregnancy as described in the memorandum.

#### 7.2.2.5 Polycyclic Aromatic Hydrocarbons

Potential cancer risks associated with the seven PAHs considered to be carcinogenic by EPA (benzo[a]pyrene, benzo[a]anthracene, benzo[b]fluoranthene, benzo[k]fluoranthene, chrysene, dibenz[a,h]anthracene, and indeno[1,2,3-cd]pyrene) will be evaluated using a toxicity equivalency approach. This toxicity equivalency approach is based on the CALEPA/CARB 1994 approach cited below, which is endorsed by Ecology in Model Toxics Control Act, WAC 173-340-708(8). Adequate toxicity data are available to determine a *CSF* only for benzo[a]pyrene. A relative potency factor (RPF) is assigned to each of the other six carcinogenic PAHs as compared to benzo[a]pyrene (refer to EPA 2005a, Table 2-8). Using this method, exposure concentrations are converted to equivalent concentrations of benzo[a]pyrene by multiplying the concentration by the appropriate toxicity equivalency factor (*TEF*). This approach results in toxicity equivalent (TEQ) concentrations of each carcinogenic PAH. The *CSF* for benzo[a]pyrene will then be used to evaluate risk from the equivalent concentration of each PAH. The *TEF*s, available from EPA (2005a) and Ecology (WAC 173-340-900), are presented in Supplement 4. This method will be applied to oral, dermal, and inhalation exposure pathways as shown in the generic equations below.

$$LADD_{TEQ} = I_{oral\ or\ dermal} \times TEF_{PAH}$$

$$ILCR_{oral\ or\ dermal} = LADD_{TEQ} \times CSF_{BaP}$$

$$ILCR_{inh} = EC \times TEF_{PAH} \times URF_{BaP}$$

where:

$LADD_{TEQ}$ = toxicity equivalent lifetime average daily dose due to oral (ingestion) or dermal (skin absorption) exposure pathways (mg/kg·day)

$I_{oral\ or\ dermal}$ = intake of the PAH of interest due to oral or dermal exposure pathways (mg/kg·day)

$TEF_{PAH}$ = toxicity equivalency factor associated with the PAH of interest (unitless)

$ILCR_{oral\ or\ dermal}$ = incremental lifetime cancer risk from oral or dermal exposure pathways (unitless)

$CSF_{BaP}$ = Oral or dermal *CSF* for benzo[a]pyrene (kg·day/mg)

$ILCR_{inh}$ = incremental lifetime cancer risk from inhalation pathways (unitless)

*EC* = exposure concentration of the PAH of interest through inhalation (mg/m$^3$).

*URF* = inhalation unit risk factor for benzo[a]pyrene (m$^3$/µg)

One limitation to this approach is that it does not measure point-of-action effects, such as skin cancer.

The *TEF*s are available from *Model Toxics Control Act* (MTCA) Part IX (WAC 173-340-900) for additional potentially carcinogenic PAH COPCs not included in EPA guidance (refer to MTCA Tables 708-2 and 708-3). No *RfD* values are available for evaluating noncancer effects for these PAHs. Noncancer-only effects are evaluated for acenaphthene, anthracene, fluorene, naphthalene, and pyrene. These are included with their *RfD*s in Supplement 4 (organic COPCs). If PAHs are predicted to be important emissions from the facility based on their estimated cancer risks, surrogate toxicity values may be considered. Any selection of surrogates would be conducted by Ecology and EPA toxicologists. The WTP will provide Ecology and EPA with a list of PAHs for which surrogate values are needed. The PAHs with Ecology/EPA-provided surrogates will then be included in the quantitative evaluation. The PAHs lacking Ecology/EPA-approved surrogates will be evaluated qualitatively in the uncertainty assessment in the PRA.

#### 7.2.2.6 Polychlorinated Dibenzo-p-dioxins, Polychlorinated Dibenzofurans, and Polychlorinated Biphenyls

The PCDDs, PCDFs, and PCBs are thought to act through a common mechanism of toxicity by binding to a protein known as the arylhydrocarbon receptor (AR) (for review, see Agency for Toxic Substance and Disease Registry [ATSDR] 1997 or World Health Organization [WHO] 1998). The AR-ligand complex is responsible for the activation of genes that have a deleterious effect when they are not under proper regulation by the receptor's hormones. Interaction of dioxins and similar compounds with AR, therefore, can cause immunological, neurological, endocrine, embryotoxic, and many other effects.

The similarity in action of these compounds is thought to result from their structural similarity. Dioxin is composed of two benzene rings joined by two carbon-oxygen-carbon bonds on two adjacent carbons of each benzene ring. Dibenzofurans have two benzene rings joined by a carbon-oxygen-carbon bond and a carbon-carbon bond on two adjacent carbons of each benzene ring. Biphenyls consist of two benzene rings joined by a single carbon-carbon bond. To form the polychlorinated derivatives, chloro groups are attached at various locations, as designated in the names of the compounds. Benzene rings are planar (i.e., flat) in conformation. Because two adjacent carbons on each benzene ring are joined in dioxins and dibenzofurans, both benzene rings are held in the same plane, and the chloro groups are also in that plane. Therefore, these molecules are said to be coplanar. The coplanar structure appears to be essential for interaction with AR. The benzene rings in biphenyl can rotate relative to each other, unless there are added groups that interfere with rotation (such as 2,2′,6,6′-chloro groups, which occupy the carbons immediately on both sides of the carbon-carbon bond joining the rings). The PCB congeners that are able to form a coplanar molecule (and are called *coplanar PCBs*) can interact with AR when they are in that

configuration. Therefore, coplanar PCBs are included among the COPCs with similar action to dioxins and dibenzofurans.

Potential cancer risks associated with PCDDs/PCDFs and coplanar PCBs will be evaluated using the cancer *CSF* for 2,3,7,8-TCDD of 1.0E+06 $(mg/kg \cdot day)^{-1}$ proposed in the *Exposure and Human Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds* (EPA 2003) at the direction of Ecology and EPA Region 10 (CCN 063809). While the proposed *CSF* has not yet been approved by EPA, it is more conservative than the current *CSF* published in HEAST (1997) and in the HHRAP database, and is widely considered to represent the best available science.

A discussion of risk results using both the 1997 (HEAST) and 2003 (dioxin reassessment) *CSF*s will appear in the uncertainty section of the PRA.

Because these contaminants have a common mechanism of action, it is assumed that their toxicity to biota is additive (WHO 1998, EPA 2005a). That is, the risks from all dioxins, dibenzofurans, and coplanar PCBs will be added.

**Polychlorinated Dibenzo-p-dioxins and Polychlorinated Dibenzofurans**

The EPA (2005a) recommends evaluating all PCDD/PCDF congeners with chlorine molecules substituted in the 2, 3, 7, and 8 positions as carcinogens. Potential cancer risks associated with these PCDD/PCDFs will be evaluated using a toxicity equivalency approach. This approach assigns a relative toxicity of each of the seventeen 2,3,7,8-substituted PCDDs/PCDFs as compared to 2,3,7,8-TCDD. Using the method, exposure concentrations are converted to equivalent concentrations of 2,3,7,8-TCDD by multiplying the concentration by the appropriate TEF. This conversion results in TEQ concentrations of each congener. The *CSF* for 2,3,7,8-TCDD is then used to evaluate risk from the total TEQ concentration. The most recent TEFs, available from MTCA Part IX (WAC 173-340-900) and EPA (2005a) and provided in Supplement 4, will be used. Equations used to incorporate TEFs are the same as those shown for PAHs, with the exception that 2,3,7,8-TCDD is the toxicity surrogate instead of benzo[a]pyrene.

**Polychlorinated Biphenyls**

Coplanar PCBs and PCDDs/PCDFs are similar structurally and may act through common mechanisms of toxicity. The EPA (2005a) and Ecology have implemented the use of dioxin TEFs for coplanar, dioxin-like PCBs. Using this approach, exposure concentrations of coplanar PCBs are converted to equivalent concentrations of 2,3,7,8-TCDD by multiplying the concentration by the appropriate TEF. The *CSF* for 2,3,7,8-TCDD is used to evaluate risk from the total TEQ concentration. Potential cancer risks associated with coplanar PCB emissions will be estimated using TEFs available from the HHRAP and listed in Supplement 4. Note that TEFs are available for 12 coplanar PCBs (HHRAP Table 2-5 and MTCA Table 708-4).

The estimated dose of coplanar PCBs, expressed as 2,3,7,8-TCDD equivalents, will be added to the total estimated dose of dioxins and used to estimate total risk from “dioxin-like” compounds in addition to evaluating coplanar PCB dose separately.

Other (noncoplanar) PCBs will be evaluated using the *CSF* for PCBs shown in Supplement 4. EPA (2005a) recommends different *CSF*s for different exposure routes and chlorine contents. The most conservative *CSF* (i.e., *CSF* from the high-risk persistence tier) is presented in Supplement 4 and will be used for the PRA.

Noncancer effects of PCBs will be evaluated using the *RfD* for Aroclor-1254.

### 7.2.3 Surrogate Values

When chemical-specific toxicity values for a chemical are not available, the use of a surrogate value may be necessary. This process involves applying a toxicity value established for one chemical to another chemical for which no value has been established. The application of surrogate values is based on similarities in structure, mechanism of action, and toxicity. The following surrogate values for the SLRA have been identified by Ecology and EPA Region 10 (Table 7-10).

The use of these surrogates is reflected in the toxicity values and physical/chemical property values presented in this RAWP. In the absence of toxicity data for both the original COPC and the surrogate chemical, physical/chemical data is provided. A periodic review of available information will be performed and surrogate values will be abandoned in favor of actual peer-reviewed constituent values as they are made available (e.g., toxicity values will be updated as they become available in as described in Section 7.2.1.1).

### 7.2.4 Toxicity Information and EPA Guidance for ROPCs

This section provides the toxicity values that will be used for evaluating ROPCs and the source/rationale for these values.

#### 7.2.4.1 Chronic Noncarcinogenic Toxicity of ROPCs

The ROPCs are not evaluated for noncarcinogenic effects; however, the stable form of ROPCs with noncarcinogenic effects is evaluated as COPCs. For example, the potential cancer effect of Sr-90 is evaluated as an ROPC while the potential noncancer effects of stable strontium are evaluated as a COPC. The list of inorganic COPCs includes the stable form of 11 ROPCs (antimony, barium, cadmium, cobalt, nickel, selenium, strontium, tin, uranium, yttrium, and zirconium).

#### 7.2.4.2 Carcinogenic Toxicity of ROPCs

Ionizing radiation, and therefore all ROPCs, is considered to be a Group A carcinogen. Cancer risk from exposure to ROPCs through ingestion, inhalation, and external exposure to radionuclides in soil is estimated using a *CSF*. Ingestion and inhalation *CSF*s are central estimates from a linear model of the age-averaged, lifetime radiation cancer incidence risk per unit of activity inhaled or ingested, and are expressed in units of risk/pCi (i.e., $pCi^{-1}$). Ingestion *CSF*s are taken from the *Health Effects Assessment Summary Tables (HEAST) 2001 Update* (EPA 2001) and are tabulated separately for ingestion of tap water, dietary intakes, and incidental soil ingestion. Inhalation *CSF*s (EPA 2001) are provided separately for inhalation of particulates and vapors or gas.

For external exposure to radionuclides in soil, *CSF*s are central estimates of lifetime radiation cancer risk for each year of exposure to external radiation from photon-emitting radionuclides distributed uniformly in a thick layer of soil. These *CSF*s are expressed as risk/yr per pCi/gram soil (i.e., $[pCi\text{-}yr/g]^{-1}$). The *CSF*s provided for external exposure in HEAST (EPA 2001) are derived from risk coefficients listed in *Federal Guidance Report No. 13* (FGR No. 13) (EPA 1999c) that assume an infinite depth of contaminated soil. For the WTP, however, it is expected that ROPCs will be deposited on the surface and will be uniformly distributed over the top 2 cm of the soil and not to an infinite depth (EPA 2005a). FGR No. 12 (EPA 1993) also provides dose coefficients for a soil depth of 1 cm and 5 cm. The ROPC

contamination level in untilled soil is based on an assumed contamination depth of 2 cm (refer to Section 6.2). For conservatism, dose coefficients for a soil depth of 5 cm are preferred over using 1 cm dose coefficients or straight line extrapolation between the 1 and 5 cm dose coefficients. The HEAST *CSF*s are, therefore, adjusted using dose coefficients provided in FGR No. 12 (EPA 1993), assuming that risk coefficients (and *CSF*s) scale proportionally with dose coefficients and depth. Using this approach, adjustments to HEAST factors are made using the following equation (CCN 064328):

$$CSF_{adj} = CSF_{HEAST} \times (DC_5 \div DC_{inf}) \qquad \text{(Equation 7-9)}$$

where:

| | | |
|---|---|---|
| $CSF_{adj}$ | = | adjusted cancer slope factor for external exposure to radionuclides in soil |
| $CSF_{HEAST}$ | = | HEAST factor for an infinite depth |
| $DC_5$ | = | FGR No. 12 (Table III.5) dose coefficient for 5 cm depth |
| $DC_{inf}$ | = | FGR No. 12 (Table III.7) dose coefficient for infinite depth |

The resulting depth-corrected *CSF*s are provided in Supplement 4.

Cancer risk (morbidity) from external exposure to ionizing radiation in air is evaluated using a cancer risk factor (*RF*) expressed in units of $(\text{Bq-secs/m}^3)^{-1}$. The *RF*s are obtained from FGR No. 13 (EPA 1999c) and are provided in Supplement 4.

Some ROPCs are given the suffix "+D" to indicate that cancer risk estimates using these *CSF*s include contributions to toxicity from short-lived decay products. For example, the +D slope factor for Sb-125 includes the contribution of Te-125m, which is assumed to be in equilibrium with the parent. Risks are calculated using these +D *CSF*s. Because the +D *CSF*s for Sr-90 and Cs-137 include the contributions from their short-lived decay products (Y-90 and Ba-137m), separate risks are not calculated for these decay products (Y-90 and Ba-137m). Quantifying separate cancer risks for Y-90 and Ba-137m, in addition to using +D slope factors for Sr-90 and Cs-137, would result in double counting the toxicity of these two ROPCs.

#### 7.2.4.3 Chronic Dermal Toxicity of ROPCs

Dermal absorption of ROPCs will be evaluated for tritium. The internal dose from immersion in a plume of tritiated water vapor is approximately 50 % from inhalation and 50 % from dermal absorption (Till and Meyer, 1983); therefore, for all receptors, the dermal absorption of tritium will be accounted for in the exposure assessment by multiplying the inhalation dose for this ROPC by 2. Dermal absorption of other ROPCs will not be evaluated because this pathway is considered to be insignificant compared to inhalation for all ROPCs except tritium (See Appendix B for further discussion).

#### 7.2.4.4 Acute Toxicity of ROPCs

Acute effects from a one-hour exposure to ROPCs will be estimated based on a total acute dose limit of 0.1 rem. Appendix B provides a review of the literature that establishes the basis for defining a LOAEL for radionuclides. Based on this literature review, the lowest dose where clinically significant nonstochastic effects (i.e., the acute effects of radiation) have been observed is approximately 10 rem. Applying the California EPA methodology from *The Determination of Acute Reference Exposure Levels for Airborne Toxicants* (CalEPA 1999), a default uncertainty factor of 10 is applied to convert this

LOAEL to a NOAEL of 1 rem. The acute dose limit is then estimated by applying a second default uncertainty factor of 10 to account for intraspecies variability to provide protection to sensitive subpopulations. For radiation effects, children represent a sensitive subpopulation. This acute dose limit applies to a single exposure and does not account for intermittent exposures. This approach is very conservative. Unless 5 rem to 25 rem are delivered in a very acute exposure, there would be no adverse effect; by using 0.1 rem, there would not be any anticipated effects at this level. It must be noted that the one-hour radionuclide exposure is not comparable to the one-hour chemical exposures, and 0.1 rem is not an acute criterion.

For each of the ROPCs, acute radionuclide exposure criteria (*AREC*) corresponding to an acute dose of 0.1 rem were calculated as described below. The calculated *AREC*s include two exposure pathways associated with submergence in a cloud of particulate and vapor phase radionuclides: external gamma exposure and inhalation. The following equations were used to calculate *AREC*s for these two pathways:

External Gamma Exposure:

$$AREC_E = DL / (CDE \times ET \times CF_1) \qquad \text{(Equation 7-10)}$$

Inhalation:

$$AREC_I = DL / (CDE \times BR \times ET \times CF_1 \times CF_2) \qquad \text{(Equation 7-11)}$$

Total:

$$AREC_R = \frac{1}{\frac{1}{AREC_E} + \frac{1}{AREC_I}} \qquad \text{(Equation 7-12)}$$

where:

| | | |
|---|---|---|
| $AREC_E$ | = | acute radionuclide exposure criteria for external gamma (μCi/cm$^3$) |
| $AREC_I$ | = | acute radionuclide exposure criteria for inhalation (μCi/cm$^3$) |
| $AREC_R$ | = | total acute radionuclide exposure criteria (μCi/cm$^3$) |
| $DL$ | = | dose limit of 0.1 rem (100 mrem) |
| $CDE$ | = | committed dose equivalent for radionuclide i (Sv-m$^3$/Bq-s for external gamma; Sv/Bq for inhalation) |
| $CF_1$ | = | conversion factor $\left(10^5 \frac{\text{mrem}}{\text{Sv}} \cdot 0.037 \frac{\text{Bq}}{\text{Ci}} \cdot 10^6 \frac{\text{Ci}}{\mu\text{Ci}} \cdot 10^6 \frac{\text{cm}^3}{\text{m}^3}\right)$ |
| $ET$ | = | acute exposure time (1 hr) |
| $CF_2$ | = | conversion factor (3600 s/hr) |
| $BR$ | = | breathing rate of standard man (1.2 m$^3$/hr) |

The ROPC decay products are represented in the calculation based on their respective decay probabilities. Parent radionuclides are given the "+D" designation to indicate that decay products are considered. Supplement 4 lists the parent and decay products included in the calculations. The following equation

was used to calculate the committed dose equivalent *(CDE*) for the combination of a parent and decay product radionuclides:

$$CDE{+}D = \Sigma\ CDE_i \times f_i \qquad \text{(Equation 7-13)}$$

where:

| | | |
|---|---|---|
| $CDE{+}D$ | = | committed dose equivalent for radionuclide *i* and its daughter products |
| $CDE_i$ | = | committed dose equivalent for radionuclide *i* |
| $f_i$ | = | decay probability of radionuclide *i* |

The calculated ARECs shown result in a dose of 0.1 rem from each of the 46 ROPCs; therefore, when combined for all 46 ROPCs, these concentrations would result in a total dose of 4.4 rem. These concentrations are adjusted to ensure that the overall dose from all 46 ROPCs will not exceed 0.1 rem for an acute exposure of one hour, as shown below:

$$AREC_M = AREC_R \div 44$$

where:

| | | |
|---|---|---|
| $AREC_M$ | = | acute radionuclide exposure criteria for ROPC *i* corrected for the presence of multiple ROPCs (μCi/cm$^3$) |
| $AREC_R$ | = | acute radionuclide exposure criteria for ROPC *i* as calculated above (μCi/cm$^3$) |
| 44 | = | total number of individually quantified ROPCs (Ba-137m and Y-90 are included as daughter products and are not quantified separately) |

The $AREC_M$ values for each of the ROPCs are provided in Supplement 4.

#### 7.2.4.5 Toxicity of ROPCs to Nursing Infant

Nursing infant scenarios will be evaluated for exposure to $^{90}$Sr, $^{129}$I, $^{134}$Cs, and $^{137}$Cs. Background concentrations of $^{90}$Sr, $^{129}$I, $^{134}$Cs, and $^{137}$Cs in human breast milk are not available. The potential toxicity of these ROPCs to an infant will be evaluated using the ingestion *CSF* for each of the ROPCs to calculate lifetime cancer risk as described in Section 7.2.1.3.

## 7.3 Exposure Concentration, Lifetime Average Daily Dose, and Average Daily Dose

Inhalation dose is expressed as the exposure concentration (*EC*), while oral dose and dermal adsorption is expressed as intakes, or daily dose (as the sum of intakes). The exposure concentration is derived separately for carcinogens and noncarcinogens, but the term is used for both types of airborne constituents. For intake, carcinogens and noncarcinogens are further distinguished by terminology. For evaluating exposure to carcinogenic compounds, the intake is referred to as *LADD*. For evaluating exposure to noncarcinogenic compounds, the intake is referred to as *ADD*.

Cancer risk is estimated for each potentially carcinogenic COPC and ROPC as the product of the exposure concentration and unit risk factor, or the cumulative intake (*LADD*) and the slope factor. Non-cancer risk is estimated for each potentially noncarcinogenic COPC as the ratio of the exposure concentration and reference concentration, or the cumulative intake (*ADD*) and the reference dose. This

section describes the *EC*, the *LADD*, and the *ADD* as a prelude to risk characterization discussions where the quantitative assessment of risk due to exposure to carcinogens and noncarcinogens is described.

### 7.3.1 Exposure Concentration

For all inhalation pathways, the exposure concentration is used as a measure of receptor dose against which risk is evaluated. The equation in Table C-2-1 of the HHRAP serves as the basis for computing the exposure concentration due to long term (chronic) inhalation of emissions. Some receptor exposure scenarios include exposures that include time spent in what the RAGS Part F terms as "microenvironments", or exposures that last less than 24 hr/day. Per RAGS Part F Section 3.4.1, Equation 9 is applicable:

$$EC_j = \sum \left( C_a \cdot ET_j \cdot EF_j \right) \cdot \frac{ED_j}{AT_j \cdot CF} \qquad \text{(RAGS Part F Eq. 9)}$$

where:

$C_a$ = concentration of COPC or ROPC in air (μg/m$^3$ or pCi/m$^3$) calculated as described in Section 6.1

$EC_j$ = exposure concentration for microenvironment *j* (mg/m$^3$ or pCi/m$^3$)

$ET_j$ = exposure time for microenvironment *j* (hr/day)

$EF_j$ = exposure frequency microenvironment *j* (day/yr)

$ED_j$ = exposure duration for microenvironment *j* (yr)

$AT$ = averaging time for carcinogens ($AT_C$) or noncarcinogens ($AT_N$) (yr)

$CF$ = conversion factor (8760 hr/yr)

This is the same equation as presented in sections 7.1.5.1, 7.1.5.2 (substituting the concentration in airborne soil concentration for air concentration) and 7.1.5.5 (substituting airborne vapor and aerosol concentration for air concentration).

Per the RAGS Part F Section 3.4.2, to derive an average *EC* for a receptor over multiple exposure periods (e.g., exposures that occur for only a portion of the receptor's entire averaging time), the average *EC* from each period can be weighted by the fraction of the total exposure time that each period represents, using Equation 10 of the RAGS Part F.

$$EC_{LT} = \sum \left( EC_j \cdot ED_j \right) / AT \qquad \text{(RAGS Part F Eq. 10)}$$

where:

$EC_{LT}$ = long-term average exposure concentration (mg/m$^3$ or pCi/m$^3$)

$EC_j$ = exposure concentration for the period represented by $ED_j$ (mg/m$^3$ or pCi/m$^3$)

$ED_j$ = exposure duration for period *j* (yr)

$AT$ = averaging time for carcinogens ($AT_C$) or noncarcinogens ($AT_N$) (yr)

The equations below illustrate the computation of the *EC*. The equations below are based on HHRAP equations in Table C-2-1 with modifications to account for the differing receptor lifestyles. The equations which follow contain subscripted parameters (some absent in previous equations) to aid in distinguishing exposure pathways and constituent carcinogeneity.

**Hanford site industrial worker:**

for inhalation of COPCs and ROPCs via air:

$$EC_{inh\ job} = \frac{C_{a_{g\,max}} \cdot ET_{job} \cdot EF_{job} \cdot ED_{job}}{AT_{C\,worker}} \qquad EC_{inh\ job} = \frac{C_{a_{g\,max}} \cdot ET_{job} \cdot EF_{job} \cdot ED_{job}}{AT_{N\,worker}}$$

$$EC_{inh\ offs} = \frac{C_{a_{offs}} \cdot ET_{home} \cdot EF_{home} \cdot ED_{home}}{AT_{C\,worker}} \qquad EC_{inh\ offs} = \frac{C_{a_{offs}} \cdot ET_{home} \cdot EF_{home} \cdot ED_{home}}{AT_{N\,worker}}$$

$$EC_{inh\ wkend} = \frac{C_{a_{offs}} \cdot ET_{wkend} \cdot EF_{wkend} \cdot ED_{wkend}}{AT_{C\,worker}} \qquad EC_{inh\ wkend} = \frac{C_{a_{offs}} \cdot ET_{wkend} \cdot EF_{wkend} \cdot ED_{wkend}}{AT_{N\,worker}}$$

$$EC_{inh\ retire} = \frac{C_{a_{offs}} \cdot ET_{retire} \cdot EF_{retire} \cdot ED_{retire}}{AT_{C_{retire}}} \qquad EC_{inh\ retire} = \frac{C_{a_{offs}} \cdot ET_{retire} \cdot EF_{retire} \cdot ED_{retire}}{AT_{N_{retire}}}$$

$$EC_{inh} = EC_{inh\ job} \cdot \frac{ED_{job}}{AT_C} + EC_{inh\ offs} \cdot \frac{ED_{home}}{AT_C} + EC_{inh\ wkend} \cdot \frac{ED_{wkend}}{AT_C} + EC_{inh\ retire} \cdot \frac{ED_{retire}}{AT_C} \text{ or}$$

$$EC_{inh} = EC_{inh\ job} \cdot \frac{ED_{job}}{AT_N} + EC_{inh\ offs} \cdot \frac{ED_{home}}{AT_N} + EC_{inh_{wkend}} \cdot \frac{ED_{wkend}}{AT_N} + EC_{inh\ retire} \cdot \frac{ED_{retire}}{AT_N}$$

and inhalation of COPCs and ROPCs via resuspended soil:

$$EC_{soil\ job} = \frac{Cs_{g\,max}}{PEF} \cdot \frac{ET_{job} \cdot EF_{job} \cdot ED_{job}}{AT_{C\,worker}} \qquad EC_{soil\ job} = \frac{Cs_{tD_{g\,max}}}{PEF} \cdot \frac{ET_{job} \cdot EF_{job} \cdot ED_{job}}{AT_{N\,worker}}$$

$$EC_{soil\ offs} = \frac{Cs_{offs}}{PEF} \cdot \frac{ET_{home} \cdot EF_{home} \cdot ED_{home}}{AT_{C\,worker}} \qquad EC_{soil\ offs} = \frac{Cs_{tD_{offs}}}{PEF} \cdot \frac{ET_{home} \cdot EF_{home} \cdot ED_{home}}{AT_{N\,worker}}$$

$$EC_{soil\ wkend} = \frac{Cs_{off}}{PEF} \cdot \frac{ET_{wkend} \cdot EF_{wkend} \cdot ED_{wkend}}{AT_{C\,worker}} \qquad EC_{soil\ wkend} = \frac{Cs_{tD_{offs}}}{PEF} \cdot \frac{ET_{wkend} \cdot EF_{wkend} \cdot ED_{wkend}}{AT_{N\,worker}}$$

$$EC_{soil\ retire} = \frac{Cs_{offs}}{PEF} \cdot \frac{ET_{retire} \cdot EF_{retire} \cdot ED_{retire}}{AT_{C_{retire}}} \qquad EC_{soil\ retire} = \frac{Cs_{tD_{offs}}}{PEF} \cdot \frac{ET_{retire} \cdot EF_{retire} \cdot ED_{retire}}{AT_{N_{retire}}}$$

$$EC_{soil} = EC_{soil\ job} \cdot \frac{ED_{job}}{AT_C} + EC_{soil\ offs} \cdot \frac{ED_{home}}{AT_C} + EC_{soil\ wkend} \cdot \frac{ED_{wkend}}{AT_C} + EC_{soil\ retire} \cdot \frac{ED_{retire}}{AT_C}$$

$$EC_{soil} = EC_{soil\ job} \cdot \frac{ED_{job}}{AT_N} + EC_{soil\ offs} \cdot \frac{ED_{home}}{AT_N} + EC_{soil_{wkend}} \cdot \frac{ED_{wkend}}{AT_N} + EC_{soil\ retire} \cdot \frac{ED_{retire}}{AT_N}$$

where:

| | | |
|---|---|---|
| $EC_{inh_j}$ | = | worker exposure concentration of COPCs or ROPCs through inhalation of emissions while at various microenvironments, *j*, defined by the subscripts below (Section 7.1.5.1) (mg/m$^3$ or pCi/m$^3$) |
| $EC_{soil_j}$ | = | worker exposure concentration of COPCs or ROPCs through inhalation of resuspended soil while at various microenvironments, *j*, defined by the subscripts below (Section 7.1.5.2) (mg/m$^3$ or pCi/m$^3$) |
| $EC_{inh}$ | = | long-term (cumulative) worker exposure concentration of COPCs or ROPCs through inhalation of emissions during the exposure scenario (note: the subscript "LT" is omitted for consistency with other receptors with non-periodic exposure scenarios) (mg/m$^3$ or pCi/m$^3$) |
| $EC_{soil}$ | = | long-term worker exposure concentration of COPCs or ROPCs through inhalation of resuspended soil of emissions during the exposure scenario (note: the subscript "LT" is omitted for consistency with other receptors with non-periodic exposure scenarios) (mg/m$^3$ or pCi/m$^3$) |
| $C_a$ | = | concentration of COPC or ROPC in air for each location of interest (ground maximum for on the job exposures, and offsite 90th percentile for after work, weekends, and during retirement) (Section 6.1) (μg/m$^3$ or pCi/m$^3$) |
| $Cs_j$ | = | concentration of carcinogenic COPC or ROPC in soil for each location of interest (ground maximum for on the job exposures, and offsite 90th percentile for after work, weekends, and during retirement) (Section 6.2) (mg/kg or pCi/g) |
| $Cs_{tD_j}$ | = | concentration of noncarcinogenic COPC in soil for each location of interest (ground maximum for on the job exposures, and offsite 90th percentile for after work, weekends, and during retirement) (Section 6.2) (mg/kg) |
| $ET$ | = | exposure time (hr/day) |
| $EF$ | = | exposure frequency (day/yr) |
| $ED$ | = | exposure duration (yr) |
| $AT$ | = | when shown with a receptor-specific subscript: averaging time for carcinogens ( $AT_{C_{worker}}$ or $AT_{C_{retire}}$ ) or noncarcinogens ( $AT_{N_{worker}}$ or $AT_{N_{retire}}$ ) corresponding to the worker or retiree exposure. When shown without a receptor-specific subscript: total scenario averaging time for carcinogens ($AT_C$) or noncarcinogens ($AT_N$) corresponding to the worker plus retiree averaging time ( $AT_{C_{worker}} + AT_{C_{retire}}$ or $AT_{N_{worker}} + AT_{N_{retire}}$ ) (yr) |
| subscript *gmax* | = | value associated with the ground maximum location |
| subscript *offs* | = | value associated with the offsite location |
| subscript *job* | = | value associated with worker exposures on the job |
| subscript *home* | = | value associated with worker exposures after work, while presumably at home |
| subscript *wkend* | = | value associated with worker exposures on the weekends |

subscript *retire* = value associated with exposures of the retired worker

Conversion factors are omitted for simplicity.

**Resident, subsistence farmer, and subsistence fisher:**

for inhalation of COPCs and ROPCs via air:

$$EC_{inh} = \frac{C_a \cdot ET \cdot EF \cdot ED}{AT_C} \qquad EC_{inh} = \frac{C_a \cdot ET \cdot EF \cdot ED}{AT_N}$$

and inhalation of COPCs and ROPCs via resuspended soil:

$$EC_{soil} = \frac{Cs}{PEF}\frac{ET \cdot EF \cdot ED}{AT_C} \qquad EC_{soil} = \frac{Cs_{tD}}{PEF}\frac{ET \cdot EF \cdot ED}{AT_N}$$

where:

$EC_{inh}$ = exposure concentration of COPCs or ROPCs through inhalation of emissions (Section 7.1.5.1) (mg/m$^3$ or pCi/m$^3$)

$EC_{soil}$ = exposure concentration of COPCs or ROPCs through inhalation of resuspended soil (Section 7.1.5.2) (mg/m$^3$ or pCi/m$^3$)

$C_a$ = offsite air concentration of COPC or ROPC in air (Section 6.1) (µg/m$^3$ or pCi/m$^3$)

$Cs$ = offsite soil concentration of carcinogenic COPC or ROPC (Section 6.2) (mg/kg or pCi/g)

$Cs_{tD}$ = offsite soil concentration of noncarcinogenic COPC (Section 6.2) (mg/kg)

$ET$ = exposure time (hr/day)

$EF$ = exposure frequency (day/yr)

$ED$ = exposure duration (yr)

$AT$ = averaging time for carcinogens ($AT_C$) or noncarcinogens ($AT_N$) (yr)

Conversion factors are omitted for simplicity.

**Resident Subsistence American Indian and Alternate Resident Subsistence American Indian #2:**

for inhalation of COPCs and ROPCs via air:

$$EC_{inh} = \frac{C_a \cdot ET \cdot EF \cdot ED}{AT_C} \qquad EC_{inh} = \frac{C_a \cdot ET \cdot EF \cdot ED}{AT_N}$$

and inhalation of COPCs and ROPCs via resuspended soil:

$$EC_{soil} = \frac{Cs}{PEF}\frac{ET \cdot EF \cdot ED}{AT_C} \qquad EC_{soil} = \frac{Cs_{tD}}{PEF}\frac{ET \cdot EF \cdot ED}{AT_N}$$

and inhalation of COPCs and ROPCs via vapors and aerosols released during sweat lodge events:

Volatile and semivolatile organic COPC and volatile ROPC vapors:

$$EC_{sl} = C_{dw} \cdot \left(\frac{V_w}{2}\right) \cdot \left(\frac{1}{\frac{2}{3} \cdot \pi \cdot r^3}\right) \frac{ET_{sl} \cdot EF_{sl} \cdot ED_{sl}}{AT_C} \qquad EC_{sl} = C_{dw} \cdot \left(\frac{V_w}{2}\right) \cdot \left(\frac{1}{\frac{2}{3} \cdot \pi \cdot r^3}\right) \frac{ET_{sl} \cdot EF_{sl} \cdot ED_{sl}}{AT_N}$$

Non-volatile COPC and ROPC aerosols:

$$EC_{sl} = C_{dw} \cdot \left(\frac{MW_w}{R \cdot T_{sl} \cdot \rho_w}\right) \cdot e^{\left(18.3036 - \frac{3816.44}{T_{sl} - 46.13}\right)} \frac{ET_{sl} \cdot EF_{sl} \cdot ED_{sl}}{AT_C}$$

$$EC_{sl} = C_{dw} \cdot \left(\frac{MW_w}{R \cdot T_{sl} \cdot \rho_w}\right) \cdot e^{\left(18.3036 - \frac{3816.44}{T_{sl} - 46.13}\right)} \frac{ET_{sl} \cdot EF_{sl} \cdot ED_{sl}}{AT_N}$$

where:

| | | |
|---|---|---|
| $EC_{inh}$ | = | exposure concentration of COPCs or ROPCs through inhalation of emissions (Section 7.1.5.1) (mg/m$^3$ or pCi/m$^3$) |
| $EC_{soil}$ | = | exposure concentration of COPCs or ROPCs through inhalation of resuspended soil (Section 7.1.5.2) (mg/m$^3$ or pCi/m$^3$) |
| $EC_{sl}$ | = | exposure concentration of COPCs or ROPCs through inhalation in the sweat lodge (Section 7.1.5.5) (mg/m$^3$ or pCi/m$^3$) |
| $C_a$ | = | offsite air concentration of COPC or ROPC in air (Section 6.1) (µg/m$^3$ or pCi/m$^3$) |
| $Cs$ | = | offsite soil concentration of carcinogenic COPC or ROPC (Section 6.2) (mg/kg or pCi/g) |
| $Cs_{tD}$ | = | offsite soil concentration of noncarcinogenic COPC (Section 6.2) (mg/kg) |
| $C_{dw}$ | = | dissolved surface water concentration of COPCs and ROPCs (Section 6.3) (mg/L or pCi/L) |
| $V_W$ | = | volume of water (4 L) |
| $\pi$ | = | the constant *pi* (unitless); $\pi \approx 3.14159265359$ |
| $r$ | = | radius of sweat lodge (1 m) |
| $MW_w$ | = | molar weight of water (18.01528 g/mol) |
| $R$ | = | ideal gas constant (0.06237 mmHg·m$^3$/gmole·K) |
| $T_{sl}$ | = | temperature of the sweat lodge (339 K) |
| $\rho_w$ | = | density of liquid water at temperature $T_{sl}$ (980.2 g/L) |
| $ET_{sl}$ | = | sweat lodge exposure time (hr/day) |
| $EF_{sl}$ | = | sweat lodge exposure frequency (day/yr) |
| $ED_{sl}$ | = | sweat lodge exposure duration (yr) |
| $AT$ | = | averaging time for carcinogens ($AT_C$) or noncarcinogens ($AT_N$) (yr) |
| $e$ | = | base of the natural logarithm (unitless), ≈ 2.718282; the units for the Antoine equation, $e^{\left(18.3036 - \frac{3816.44}{T_{sl} - 46.13}\right)}$, are mmHg |

Conversion factors are omitted for simplicity.

**Alternate Resident Subsistence American Indian #1:**

for inhalation of COPCs and ROPCs via air:

$$EC_{inh\ offs} = \frac{C_{a_{offs}} \cdot ET_{home} \cdot EF_{home} \cdot ED_{home}}{AT_C} \qquad EC_{inh\ offs} = \frac{C_{a_{offs}} \cdot ET_{home} \cdot EF_{home} \cdot ED_{home}}{AT_N}$$

$$EC_{inh\ gbl} = \frac{C_{a_{gbl}} \cdot ET_{gbl} \cdot EF_{gbl} \cdot ED_{gbl}}{AT_C} \qquad EC_{inh\ gbl} = \frac{C_{a_{offs}} \cdot ET_{gbl} \cdot EF_{gbl} \cdot ED_{gbl}}{AT_{N\ gbl}}$$

$$EC_{inh} = EC_{inh\ offs} \cdot \frac{ED_{home}}{AT_C} + EC_{inh\ gbl} \cdot \frac{ED_{gbl}}{AT_C} \qquad EC_{inh} = EC_{inh\ offs} \cdot \frac{ED_{home}}{AT_N} + EC_{inh_{gbl}} \cdot \frac{ED_{gbl}}{AT_N}$$

and inhalation of COPCs and ROPCs via resuspended soil:

$$EC_{soil\ offs} = \frac{Cs_{offs}}{PEF} \cdot \frac{ET_{home} \cdot EF_{home} \cdot ED_{home}}{AT_C} \qquad EC_{soil\ offs} = \frac{Cs_{tD_{offs}}}{PEF} \cdot \frac{ET_{home} \cdot EF_{home} \cdot ED_{home}}{AT_N}$$

$$EC_{soil\ gbl} = \frac{Cs_{gbl}}{PEF} \cdot \frac{ET_{gbl} \cdot EF_{gbl} \cdot ED_{gbl}}{AT_C} \qquad EC_{soil\ gbl} = \frac{Cs_{tD_{gbls}}}{PEF} \cdot \frac{ET_{gbl} \cdot EF_{gbl} \cdot ED_{gbl}}{AT_N}$$

$$EC_{soil} = EC_{soil\ offs} \cdot \frac{ED_{home}}{AT_C} + EC_{soil\ gbl} \cdot \frac{ED_{gbl}}{AT_C} \qquad EC_{soil} = EC_{soil\ offs} \cdot \frac{ED_{home}}{AT_N} + EC_{soil_{gbl}} \cdot \frac{ED_{gbl}}{AT_N}$$

and inhalation of COPCs and ROPCs via vapors and aerosols released during sweat lodge events:

Volatile and semivolatile organic COPC and volatile ROPC vapors:

$$EC_{sl} = C_{dw} \cdot \left(\frac{V_w}{2}\right) \cdot \left(\frac{1}{\frac{2}{3} \cdot \pi \cdot r^3}\right) \frac{ET_{sl} \cdot EF_{sl} \cdot ED_{sl}}{AT_C} \qquad EC_{sl} = C_{dw} \cdot \left(\frac{V_w}{2}\right) \cdot \left(\frac{1}{\frac{2}{3} \cdot \pi \cdot r^3}\right) \frac{ET_{sl} \cdot EF_{sl} \cdot ED_{sl}}{AT_N}$$

Non-volatile COPC and ROPC aerosols:

$$EC_{sw} = C_{dw} \cdot \left(\frac{MW_w}{R \cdot T_{sl} \cdot \rho_w}\right) \cdot e^{\left(18.3036 - \frac{3816.44}{T_{sl} - 46.13}\right)} \frac{ET_{sl} \cdot EF_{sl} \cdot ED_{sl}}{AT_C}$$

$$EC_{sw} = C_{dw} \cdot \left(\frac{MW_w}{R \cdot T_{sl} \cdot \rho_w}\right) \cdot e^{\left(18.3036 - \frac{3816.44}{T_{sl} - 46.13}\right)} \frac{ET_{sl} \cdot EF_{sl} \cdot ED_{sl}}{AT_N}$$

where:

| | | |
|---|---|---|
| $EC_{inh}$ | = | exposure concentration of COPCs or ROPCs through inhalation of emissions (Section 7.1.5.1) (mg/m$^3$ or pCi/m$^3$) |
| $EC_{soil}$ | = | exposure concentration of COPCs or ROPCs through inhalation of resuspended soil (Section 7.1.5.2) (mg/m$^3$ or pCi/m$^3$) |

| | | |
|---|---|---|
| $EC_{sl}$ | = | exposure concentration of COPCs or ROPCs through inhalation in the sweat lodge (Section 7.1.5.5) (mg/m$^3$ or pCi/m$^3$) |
| subscript *offs* | = | value associated with the offsite location |
| subscript *home* | = | value associated with time spent at home (offsite location) |
| subscript *gbl* | = | value associated with ceremonial activities, presumably at the Gable Mountain location |

Other variables are defined above. Conversion factors are omitted for simplicity.

### 7.3.2 Lifetime Average and Average Daily Dose

For evaluating exposure to carcinogenic compounds, the intake is referred to as *LADD*. For evaluating exposure to noncarcinogenic compounds, the intake is referred to as *ADD*. The *LADD* is equivalent to the dose to the receptor averaged over a lifetime, while the *ADD* is the average dose to the receptor over the exposure period. The *LADD* is a chronic dose used for assessing long term cancer risk while the *ADD* is the average threshold dose used for assessing the hazard. For COPCs, the daily intake, *I*, must be converted to a dose (*LADD* and *ADD*) by applying receptor specific parameters such as exposure frequency and duration, body weight, etc. For ROPCs, the equations for daily intake, *I*, generally equates to the LADD because such receptor-specific parameters are already included in the ROPC equations (see Section 7.1.5). The equations which follow contain subscripted parameters (some absent in previous equations) to aid in distinguishing exposure pathways and constituent carcinogeneity.

The *LADD* and *ADD* (due to oral intake) is calculated as the product of the daily intake and the lifetime exposure frequency (*EF*) and duration (*ED*) divided by the risk averaging time (*AT*) of the exposure pathway. Values for *EF* and *ED* are pathway specific (that is, specific to the time of exposure to soil, or time during which a specific diet is consumed or water is consumed). The application of *EF* and *ED* to the *LADD* and *ADD* computation is important because it distinguishes the lifetime dose for the various receptor/exposure pathway combinations used in the risk assessment. The value of the risk averaging time for a carcinogen ($AT_C$) is the lifetime of the receptor (generally, this is 70 years). The value of the risk averaging time for a noncarcinogen ($AT_N$) is the exposure duration (*ED*) of the receptor.

The equations below illustrate the computation of the *LADD* and *ADD*. Note that the equations below are based on HHRAP equations in tables C-1-7 and C-1-8 with modifications to account for the differing receptor lifestyles. Equations for ROPCs already include terms for an average lifetime exposure so the intake, *I*, for an ROPC is the same as the *LADD* for a ROPC. Computing the *LADD* and *ADD* based upon daily intake yields:

**Hanford site industrial worker and resident:**

for COPCs:

$$LADD_{o_{soil}} = I_{soil} \cdot \frac{EF_{soil} \cdot ED_{soil}}{AT_C} \qquad ADD_{o_{soil}} = I_{soil} \cdot \frac{EF_{soil} \cdot ED_{soil}}{AT_N}$$

$$LADD_{o_{food}} = I_{ag} \cdot \frac{EF_{food} \cdot ED_{food}}{AT_C} \qquad ADD_{o_{food}} = I_{ag} \cdot \frac{EF_{food} \cdot ED_{food}}{AT_N}$$

$$LADD_{o_{dw}} = I_{dw} \cdot \frac{EF_{dw} \cdot ED_{dw}}{AT_C} \qquad ADD_{o_{dw}} = I_{dw} \cdot \frac{EF_{dw} \cdot ED_{dw}}{AT_N}$$

for ROPCs:

$$LADD_{o_{soil}} = I_{soil}$$

$$LADD_{o_{food}} = I_{ag}$$

$$LADD_{o_{dw}} = I_{dw}$$

**Resident subsistence farmer:**

for COPCs:

$$LADD_{o_{soil}} = I_{soil} \cdot \frac{EF_{soil} \cdot ED_{soil}}{AT_C} \qquad ADD_{o_{soil}} = I_{soil} \cdot \frac{EF_{soil} \cdot ED_{soil}}{AT_N}$$

$$LADD_{o_{food}} = \begin{pmatrix} I_{ag} + I_{beef} + I_{milk} + \\ I_{pork} + I_{chicken} + I_{egg} \end{pmatrix} \cdot \frac{EF_{food} \cdot ED_{food}}{AT_C} \qquad ADD_{o_{food}} = \begin{pmatrix} I_{ag} + I_{beef} + I_{milk} + \\ I_{pork} + I_{chicken} + I_{egg} \end{pmatrix} \cdot \frac{EF_{food} \cdot ED_{food}}{AT_N}$$

$$LADD_{o_{dw}} = I_{dw} \cdot \frac{EF_{dw} \cdot ED_{dw}}{AT_C} \qquad ADD_{o_{dw}} = I_{dw} \cdot \frac{EF_{dw} \cdot ED_{dw}}{AT_N}$$

for ROPCs:

$$LADD_{o_{soil}} = I_{soil}$$

$$LADD_{o_{food}} = I_{ag} + I_{beef} + I_{milk} + I_{pork} + I_{chicken} + I_{egg}$$

$$LADD_{o_{dw}} = I_{dw}$$

**Resident subsistence fisher:**

for COPCs:

$$LADD_{o_{soil}} = I_{soil} \cdot \frac{EF_{soil} \cdot ED_{soil}}{AT_C} \qquad ADD_{o_{soil}} = I_{soil} \cdot \frac{EF_{soil} \cdot ED_{soil}}{AT_N}$$

$$LADD_{o_{food}} = (I_{ag} + I_{fish}) \cdot \frac{EF_{food} \cdot ED_{food}}{AT_C} \qquad ADD_{o_{food}} = (I_{ag} + I_{fish}) \cdot \frac{EF_{food} \cdot ED_{food}}{AT_N}$$

$$LADD_{o_{dw}} = I_{dw} \cdot \frac{EF_{dw} \cdot ED_{dw}}{AT_C} \qquad ADD_{o_{dw}} = I_{dw} \cdot \frac{EF_{dw} \cdot ED_{dw}}{AT_N}$$

for ROPCs:

$$LADD_{o_{soil}} = I_{soil}$$

$$LADD_{o_{food}} = I_{ag} + I_{fish}$$

$$LADD_{o_{dw}} = I_{dw}$$

**Resident Subsistence American Indian:**

for COPCs:

$$LADD_{o_{soil}} = I_{soil} \cdot \frac{EF_{soil} \cdot ED_{soil}}{AT_C} \qquad ADD_{o_{soil}} = I_{soil} \cdot \frac{EF_{soil} \cdot ED_{soil}}{AT_N}$$

$$LADD_{o_{food}} = \begin{pmatrix} I_{ag} + I_{game} + I_{wild\ \ fowl} + \\ I_{wild\ \ eggs} + I_{fish} \end{pmatrix} \cdot \frac{EF_{food} \cdot ED_{food}}{AT_C} \qquad ADD_{o_{food}} = \begin{pmatrix} I_{ag} + I_{game} + I_{wild\ \ fowl} + \\ I_{wild\ \ eggs} + I_{fish} \end{pmatrix} \cdot \frac{EF_{food} \cdot ED_{food}}{AT_N}$$

$$LADD_{o_{dw}} = I_{dw} \cdot \frac{EF_{dw} \cdot ED_{dw}}{AT_C} \qquad ADD_{o_{dw}} = I_{dw} \cdot \frac{EF_{dw} \cdot ED_{dw}}{AT_N}$$

for ROPCs:

$$LADD_{o_{soil}} = I_{soil}$$

$$LADD_{o_{food}} = I_{ag} + I_{wild\ \ fowl} + I_{wild\ \ egg} + I_{game} + I_{fish}$$

$$LADD_{o_{dw}} = I_{dw}$$

where:

| | | |
|---|---|---|
| $LADD_{o_{soil}}$ | = | lifetime average daily dose from soil ingestion (unitless) |
| $LADD_{o_{food}}$ | = | lifetime average daily dose from food ingestion (unitless) |
| $LADD_{o_{dw}}$ | = | lifetime average daily dose from water ingestion (unitless) |
| $ADD_{o_{soil}}$ | = | average daily dose from soil ingestion (unitless) |
| $ADD_{o_{food}}$ | = | average daily dose from food ingestion (unitless) |
| $ADD_{o_{dw}}$ | = | average daily dose from water ingestion (unitless) |
| $I_{soil}$ | = | intake of COPC or ROPC due to soil ingestion (Section 7.1.5.2) (mg/kg·day or pCi) |
| $I_{ag}$ | = | intake of COPC or ROPC through ingestion of produce (Section 7.1.5.3) (mg/kg·day or pCi) |
| $I_{beef}$ | = | intake of COPC or ROPC through ingestion of beef (Section 7.1.5.3) (mg/kg·day or pCi) |

| | | |
|---|---|---|
| $I_{milk}$ | = | intake of COPC or ROPC through ingestion of milk (Section 7.1.5.3) (mg/kg·day or pCi) |
| $I_{pork}$ | = | intake of COPC or ROPC through ingestion of pork (Section 7.1.5.3) (mg/kg·day or pCi) |
| $I_{chicken}$ | = | intake of COPC or ROPC through ingestion of chicken (Section 7.1.5.3) (mg/kg·day or pCi) |
| $I_{egg}$ | = | intake of COPC or ROPC through ingestion of eggs (Section 7.1.5.3) (mg/kg·day or pCi) |
| $I_{game}$ | = | intake of COPC or ROPC through ingestion of game (Section 7.1.5.3) (mg/kg·day or pCi) |
| $I_{wild\ fowl}$ | = | intake of COPC or ROPC through ingestion of wild fowl (Section 7.1.5.3) (mg/kg·day or pCi) |
| $I_{wild\ egg}$ | = | intake of COPC or ROPC through ingestion of wild eggs (Section 7.1.5.3) (mg/kg·day or pCi) |
| $I_{dw}$ | = | intake of COPC or ROPC from drinking water (Section 7.1.5.4) (mg/kg·day or pCi) |
| $EF_{soil}$ | = | soil exposure frequency (day/yr), the number of days per year the receptor is at a location exposed to incidental soil ingestion. |
| $ED_{soil}$ | = | soil exposure duration (yr), the number of years the receptor is at a location exposed to incidental soil ingestion. |
| $EF_{food}$ | = | food exposure frequency (day/yr), the number of days per year a receptor consumes a food type used to assess the exposure through food ingestion (e.g., produce, game, beef, fish, etc.). |
| $ED_{food}$ | = | food exposure duration (yr), the number of years a receptor consumes a food type used to assess the exposure through food ingestion (e.g. produce, game, beef, fish, etc.). |
| $EF_{dw}$ | = | drinking water exposure frequency (day/yr), the number of days per year the receptor drinks water from the Columbia River. |
| $ED_{dw}$ | = | drinking water exposure duration (yr), the number of years the receptor drinks water from the Columbia River. |
| $BW$ | = | body weight (kg) |
| $AT_C$ | = | averaging time for carcinogens (yr), generally equal to the receptor's lifespan (70 years) |
| $AT_N$ | = | averaging time for noncarcinogens (yr), generally equal to the receptor's exposure duration (*ED*) |

Conversion factors are omitted for simplicity.

### 7.3.3 External and Dermal Dose

The external dose is consider the dose adsorbed by the receptor due to external radiation (from air and soil) exposure while dermal dose is the dose absorbed through the skin by immersion to constituents. Dermal dose is only assessed in a sweat lodge exposure scenario.

For airborne ROPCs, the *LADD* is computed as:

$$LADD_{ira} = I_{ira}$$

where:

$LADD_{ira}$ = lifetime average daily dose, or intake, $I_{ira}$, from external radiation for airborne ROPCs (Bq·sec/m$^3$)

$I_{ira}$ = external exposure to gamma radiation from ROPCs in air (Section 7.1.5.1) (Bq·sec/m$^3$)

For ROPCs in surface soil, the *LADD* is computed as:

$$LADD_{irs} = I_{irs}$$

where:

$LADD_{irs}$ = lifetime average daily dose, or intake, $I_{irs}$, from external radiation for ROPCs in soil (pCi·yr/g)

$I_{irs}$ = external exposure to gamma radiation from ROPCs in soil (Section 7.1.5.2) (pCi·yr/g)

For the resident subsistence American Indian exposure scenario(s), dermal absorption occurs in the sweat lodge. The equations for computing the *LADD* and *ADD* from the dermal sweat lodge pathway are:

COPCs: $LADD_d = I_d \cdot \frac{EF_{sl} \cdot ED_{sl}}{AT_C}$ and

$$ADD_d = I_d \cdot \frac{EF_{sl} \cdot ED_{sl}}{AT_N}$$

where:

$LADD_d$ = lifetime average daily dose, or intake, $I_d$, from dermal absorption of condensate and vapors within the sweat lodge (mg/kg·day)

$I_d$ = intake of COPCs from adult dermal absorption within the sweat lodge (Section 7.1.5.5) (mg/kg·day or pCi)

$EF_{sl}$ = sweat lodge exposure frequency (day/yr)

$ED_{sl}$ = sweat lodge exposure duration (yr)

$AT_C$ = averaging time for carcinogens (yr), generally equal to the receptor's lifespan (70 years)

$AT_N$ = averaging time for noncarcinogens (yr), generally equal to the receptor's exposure duration (*ED*)

## 7.4 Risk Characterization

The purpose of the risk characterization is to evaluate the information obtained through the exposure (Section 7.1), toxicity (Section 7.2), and dose (Section 7.3) assessments to estimate the potential for receptors to experience adverse effects (cancer risks and noncancer hazards) as a result of exposure to media contaminated by emissions from the WTP. Potential health risks will be characterized separately for noncarcinogenic and carcinogenic endpoints, and chemical (i.e., nonradiological) and radiological cancer risks will be evaluated and presented separately.

### 7.4.1 Risk Characterization for Carcinogens

For carcinogens, risk is expressed as the probability that an individual will develop cancer over a lifetime as a result of exposure to a carcinogen. Cancer risk from exposure to contamination is expressed as *ILCR*, or the increased chance of cancer above the normal background rate of cancer. Cancer risk from external exposure to ionizing radiation is expressed in terms of morbidity.

The threshold for the total *ILCR* for COPCs, the total *ILCR* for ROPCs, and the constituent-specific and pathway-specific *ILCR* for COPCs and ROPCs is 1E-05, or 1 in 100,000 exposed individuals (EPA 2005a). The total *ILCR* is the sum of all the constituent-specific *ILCR*s for COPCs and ROPCs of applicable pathways (inhalation, oral, and external) for each receptor.

#### 7.4.1.1 Inhalation Risk for Carcinogens

Cancer risk is estimated for each potentially carcinogenic COPC as the product of the unit risk factor and the exposure concentration (EPA 2005a and EPA 2001):

For all inhalation pathways:

$$ILCR_{inh} = \sum EC_i \times URF \qquad \text{(HHRAP Eq. 7-1)}$$

where:

$ILCR_{inh}$ = incremental lifetime cancer risk from inhalation pathways (unitless)

$EC_i$ = exposure concentration of COPCs through inhalation from pathway *i* (mg/m$^3$). For all receptors except those in American Indian scenarios, the $EC_i$ is due to air and resuspended soil exposure (Sections 7.1.5.1 and 7.1.5.2). For tribal scenarios, $EC_i$ includes exposure to air, soil, and exposure in the sweat lodge (Section 7.1.5.5).

$URF$ = unit risk factor (m$^3$/µg)

However, for ROPCs, the cancer risk is estimated as the product of the slope factor and the intake. Recall that:

$$I_i = EC_i \cdot IR \cdot AT \cdot CF \qquad \text{(modified HHRAP Table C-3-1)}$$

where:

$I_i$ = intake of ROPC via inhalation pathway *i* (pCi)

$EC_i$ = exposure concentration of ROPCs through inhalation from pathway *i* (pCi/m$^3$). For all receptors except those in American Indian scenarios, the $EC_i$ is due to air and resuspended soil exposure (Sections 7.1.5.1 and 7.1.5.2). For American Indian scenarios, $EC_i$ includes exposure to air, soil, and exposure in the sweat lodge (Section 7.1.5.5).

$IR$ = inhalation rate (m$^3$/hr)

$AT$ = averaging time for carcinogens ($AT_C$) (yr)

$CF$ = units conversion factor of 8760 (hr/yr)

So that for ROPC exposure via inhalation pathways, the equation for risk is the product of intake and the slope factor:

$$ILCR_{inh} = \sum \left( I_i \times CSF_{inh} \right) \qquad \text{(modified HHRAP Eq. 7-2)}$$

This yields the following receptor-specific equations:

Hanford site industrial worker, resident, farmer, and fisher:

COPCs: $ILCR_{inh} = \left( EC_{inh} + EC_{soil} \right) \cdot URF$

ROPCs: $ILCR_{inh} = \left( EC_{inh} + EC_{soil} \right) \cdot IR \cdot AT \cdot CSF_{inh} \cdot CF$ or $ILCR_{inh} = \left( I_{inh} + I_{inhsoil} \right) \cdot CSF_{inh}$

Resident Subsistence American Indian(s):

COPCs: $ILCR_{inh} = \left( EC_{inh} + EC_{soil} + EC_{sl} \right) \cdot URF$

ROPCs: $ILCR_{inh} = \left( EC_{inh} + EC_{soil} + EC_{sl} \right) \cdot IR \cdot AT \cdot CSF_{inh} \cdot CF$ or
$ILCR_{inh} = \left( I_{inh} + I_{inhsoil} + I_{sl} \right) \cdot CSF_{inh}$

where:

$ILCR_{inh}$ = incremental lifetime cancer risk from inhalation pathways (unitless)

$EC_{inh}$ = exposure concentration of COPCs or ROPCs through inhalation of emissions (Section 7.1.5.1) (mg/m$^3$ or pCi/m$^3$)

$EC_{soil}$ = exposure concentration of COPCs or ROPCs through inhalation of resuspended soil (Section 7.1.5.2) (mg/m$^3$ or pCi/m$^3$)

$EC_{sl}$ = exposure concentration of COPCs or ROPCs through inhalation in the sweat lodge (Section 7.1.5.5) (mg/m$^3$ or pCi/m$^3$)

$I_{inh}$ = intake of ROPCs through inhalation of emissions (Section 7.1.5.1) (pCi)

$I_{soil}$ = intake of ROPCs through inhalation of resuspended soil (Section 7.1.5.2) (pCi)

$I_{sl}$ = intake of ROPCs through inhalation in the sweat lodge (Section 7.1.5.5) (pCi)

$URF$ = unit risk factor (m$^3$/µg)

$CSF_{inh}$ = radionuclide-specific inhalation cancer slope factor (pCi$^{-1}$)

$CF$ = units conversion factor of 8760 (hr/yr)

#### 7.4.1.2 Oral Risk for Carcinogens

For all oral (ingestion) pathways of constituents, Equation 7-2 of the HHRAP is used, with modifications for the pathway-specific *CSF*s that apply (refer to EPA 2001):

$$ILCR_o = \sum\left(LADD_{o_i} \times CSF_{o_i}\right) \qquad \text{(modified HHRAP Eq. 7-2)}$$

Such that:

for COPCs: $ILCR_o = \left(LADD_{o_{soil}} + LADD_{o_{food}} + LADD_{o_{dw}}\right) \cdot CSF_{oral}$

for ROPCs: $ILCR_o = LADD_{o_{soil}} \cdot CSF_{soil} + LADD_{o_{food}} \cdot CSF_{food} + LADD_{o_{dw}} \cdot CSF_{dw}$

where:

| | | |
|---|---|---|
| $ILCR_o$ | = | incremental lifetime cancer risk from oral pathways (ingestion) (unitless) |
| $LADD_{o_i}$ | = | lifetime average daily dose, or intake, of constituent via ingestion pathway *i* (water ingestion, food ingestion, and soil ingestion, as applicable) (pCi). |
| $CSF_{o_i}$ | = | pathway-specific ingestion *CSF* for pathway *i* (for COPCs, this is the oral *CSF*, for ROPCs, this is the water ingestion, food ingestion, and soil ingestion *CSF* as applicable) ($pCi^{-1}$). |
| $LADD_{o_{soil}}$ | = | incremental lifetime cancer risk from soil ingestion (unitless) |
| $LADD_{o_{food}}$ | = | incremental lifetime cancer risk from food ingestion (unitless) |
| $LADD_{o_{dw}}$ | = | incremental lifetime cancer risk from water ingestion (unitless) |
| $CSF_{oral}$ | = | chemical-specific oral (ingestion) *CSF* (kg·day/mg) |
| $CSF_{soil}$ | = | radionuclide-specific soil ingestion *CSF* ($pCi^{-1}$) |
| $CSF_{food}$ | = | radionuclide-specific food ingestion *CSF* ($pCi^{-1}$) |
| $CSF_{dw}$ | = | radionuclide-specific water ingestion *CSF* ($pCi^{-1}$) |

#### 7.4.1.3 External and Absorption Risk for Carcinogens

For COPC and ROPC exposure from external pathways (skin adsorption in the sweat lodge, and external ROPC exposures from air and soil), the cancer risk is estimated for each potentially carcinogenic COPC and ROPC as the product of the slope factor and the cumulative intake or external exposure (EPA 2005a and EPA 1999c):

$$ILCR_e = \sum\left(LADD_i \times CSF_i \text{ or } RF_{air}\right) \qquad \text{HHRAP, Eq. 7-2}$$

or, more specifically:

COPCs: $ILCR_e = ILCR_{e_{dermal}}$

ROPCs: $ILCR_e = ILCR_{e_{soil}} + ILCR_{e_{air}}$

where:

$ILCR_e$ = incremental lifetime cancer risk from external pathways (ingestion) (unitless)

$LADD_i$ = lifetime average daily dose, or intake, *I*, of constituent via pathway *i* (dermal absorption, external air and soil exposure) (mg/kg·day, pCi·yr/g, or Bq·sec/$m^3$)

$CSF_i$ = constituent-specific external pathway *CSF* for pathway *i* (dermal absorption, soil exposure) (kg·day/mg or $pCi^{-1}$)

$RF_{air}$ = radionuclide-specific risk coefficient for morbidity for pathway *i* (external air) ($m^3$/Bq·sec)

Other variables are defined below.

For exposure to ionizing radiation in soil, the following equation is used (EPA 1999c):

$$ILCR_{e_{soil}} = LADD_{irs} \times CSF_{adj}$$

where:

$ILCR_{e_{soil}}$ = incremental lifetime cancer risk due to external exposure to radionuclides in soil (unitless)

$LADD_{irs}$ = lifetime average daily dose to radiation from ROPCs in soil, $I_{irs}$, from ROPCs in soil (pCi·yr/g).

$CSF_{adj}$ = adjusted cancer slope factor for 5 cm depth (Section 7.2.4.2) ($pCi^{-1}$)

For exposure to ionizing radiation in air, the following equation is used (EPA 1999c):

$$ILCR_{e_{air}} = LADD_{ira} \times RF_{air}$$

where:

$ILCR_{e_{air}}$ = incremental lifetime cancer risk due to external exposure to radionuclides in air (unitless)

$LADD_{ira}$ = lifetime average daily dose, or intake, $I_{ira}$, from radiation from ROPCs in air (Bq·sec/$m^3$).

$RF_{air}$ = radionuclide-specific risk coefficient for morbidity for external air (Section 7.2.4.3) ($m^3$/Bq·sec)

For COPC exposure from dermal absorption pathways (skin adsorption in the sweat lodge), the HHRAP Equation 7-2 is used in conjunction with $CSF_{dermal}$ as presented in Section 7.2.1.1 for COPCs:

$$ILCR_{e_{dermal}} = LADD_d \times CSF_{dermal} \qquad \text{(modified HHRAP Eq. 7-2)}$$

where:

$ILCR_{e_{dermal}}$ = incremental lifetime cancer risk due to dermal absorption pathways (unitless). This quantity is only calculated for sweat lodge exposures

$LADD_d$ = lifetime average daily dose, or intake, $I_d$, from dermal absorption of condensate and vapors within the sweat lodge (mg/kg·day)

$CSF_{dermal}$ = dermal cancer slope factor (kg·day/mg)

The equations above yield the following receptor-specific equations:

Hanford site industrial worker[9], resident, farmer, and fisher:

COPCs: not determined

ROPCs: $ILCR_e = LADD_{irs} \cdot CSF_{adj} + LADD_{ira} \cdot RF_{air}$

Resident Subsistence American Indian(s):

COPCs: $ILCR_e = LADD_d \cdot CSF_{dermal}$

ROPCs: $ILCR_e = LADD_{irs} \cdot CSF_{adj} + LADD_{ira} \cdot RF_{air}$

#### 7.4.1.4 Additivity of Dioxins and PCBs

Chlorinated dioxins, chlorinated dibenzofurans, and coplanar chlorinated biphenyls are similar structurally and may act through common mechanisms of toxicity. Because they may have a common mechanism of action, it is assumed that the toxicity of these chemicals is additive (WHO 1998, EPA 2005a). This additivity is addressed in the risk characterization by presenting a total risk from PCDDs, PCDFs, and coplanar PCBs in addition to presenting individual risks from these chemicals.

#### 7.4.1.5 Additivity of Other Potential Carcinogens

The assumption of strict additivity of chemical carcinogens assumes that (1) intakes of individual chemicals are small, and (2) there is no interaction among chemicals (i.e., no synergism or antagonism). Uncertainties associated with the assumption of additivity of chemical carcinogens will be discussed in the uncertainty section of the PRA. Despite the uncertainty, a total ILCR from exposure to all carcinogenic COPCs will be calculated as the sum of the chemical-specific ILCRs.
The assumption of strict additivity of cancer risk from radionuclides is much less uncertain. A total ILCR from exposure to all ROPCs will be calculated as the sum of the radionuclide-specific ILCRs.

[9] Includes exposure during adulthood through retirement.

### 7.4.2 Risk Characterization for Noncarcinogens

Noncarcinogenic health hazards are characterized using a hazard quotient (*HQ*) and hazard index (*HI*) approach. The *HQ* is the ratio of the calculated *ADD* to the reference or "safe" dose as shown below:

$$HQ = \frac{ADD}{RfD} \quad \text{or} \quad HQ = \frac{EC}{RfC} \qquad \text{(HHRAP Eq. 7-5)}$$

where:

| | | |
|---|---|---|
| *HQ* | = | hazard quotient (unitless) |
| *ADD* | = | average daily dose (mg/kg·day) |
| *RfD* | = | reference dose (mg/kg·day) |
| *EC* | = | exposure air concentration (mg/m$^3$) |
| *RfC* | = | reference concentration (mg/m$^3$) |

The *HQ*s will be calculated for each noncarcinogenic COPC. The ROPCs having potential health effects unassociated with radioactivity (i.e., noncancer effects) will be evaluated as inorganic COPCs. An *HQ* of 1 or less indicates that the chemical-specific *ADD* is below the level associated with adverse effect. An *HQ* threshold level of 0.25 has been selected as a risk management decision by Ecology and EPA Region 10 to provide a conservative evaluation of hazard and is consistent with other EPA guidance (EPA 1998).

#### 7.4.2.1 Inhalation Risk for Noncarcinogens

The hazard index is estimated for each noncarcinogenic COPC as the quotient of the exposure concentration and reference concentration:

For all inhalation pathways:

$$HQ_{inh} = \frac{\sum EC_i}{RfC} \qquad \text{(HHRAP Eq. 7-5)}$$

where:

| | | |
|---|---|---|
| $HQ_{inh}$ | = | hazard quotient from inhalation pathways (unitless) |
| $EC_i$ | = | exposure concentration of COPCs through inhalation of emissions from pathway *i* (mg/m$^3$). For all receptors except those in American Indian scenarios, the inhalation pathways include air and resuspended soil exposure (Sections 7.1.5.1 and 7.1.5.2). For American Indian scenarios, inhalation pathways include exposure to air, soil, and exposure in the sweat lodge (Section 7.1.5.5). |
| *RfC* | = | chemical-specific inhalation reference concentration (mg/m$^3$) |

This yields the following receptor-specific equations:

Hanford site industrial worker, resident, farmer, and fisher:

$$HQ_{inh} = \left(EC_{inh} + EC_{soil}\right) \div RfC$$

Resident Subsistence American Indian(s):

$$HQ_{inh} = \left(EC_{inh} + EC_{soil} + EC_{sl}\right) \div RfC$$

#### 7.4.2.2 Oral Risk for Noncarcinogens

For all oral (ingestion) pathways of constituents, Equation 7-5 of the HHRAP is used:

$$HQ_o = \sum ADD_{o_i} \div RfD_{oral}$$

Such that:

$$HQ_o = \left(ADD_{o_{soil}} + ADD_{o_{food}} + ADD_{o_{dw}}\right) \div RfD_{oral}$$

where:

$HQ_o$ = hazard quotient from oral pathways (ingestion) (unitless)

$ADD_{o_i}$ = average daily dose, or intake, of constituent via ingestion pathway *i* (water ingestion, food ingestion, and soil ingestion, as applicable) (mg/kg·day).

$RfD_{oral}$ = ingestion *RfD* (mg/kg·day).

$ADD_{o_{soil}}$ = incremental lifetime cancer risk from soil ingestion (unitless)

$ADD_{o_{food}}$ = incremental lifetime cancer risk from food ingestion (unitless)

$ADD_{o_{dw}}$ = incremental lifetime cancer risk from water ingestion (unitless)

#### 7.4.2.3 Absorption Risk for Noncarcinogens

There is no external radiation exposure pathway associated with COPCs. For COPC exposure from dermal absorption pathways (skin adsorption in the sweat lodge), the HHRAP Equation 7-5 is used in conjunction with $RfD_{dermal}$ as presented in Section 7.2.1.1 for COPCs:

$$HQ_d = ADD_d \div RfD_{dermal} \qquad \text{(HHRAP Eq. 7-5)}$$

where:

$HQ_d$ = incremental lifetime cancer risk due to dermal absorption pathways (unitless). This quantity is only calculated for sweat lodge exposures

$ADD_d$ = average daily dose, or intake, $I_d$, from dermal absorption of condensate and vapors within the sweat lodge (mg/kg·day)

$RfD_{dermal}$ = dermal reference dose (mg/kg·day)

#### 7.4.2.4 Additivity of Noncarcinogens

Multiple chemical exposures can result in synergism, antagonism, and/or additivity of biological responses when the chemicals act on similar target organs or when they are metabolized by the same enzymatic pathways. Additivity of noncarcinogenic health effects should only be considered if the chemicals have the same toxicological endpoint (e.g., organ or enzyme system), which implies the same mechanism of action. Additivity for all chemicals will initially be assumed for the SLRA regardless of toxicological mechanism or endpoint. This approach is likely to overestimate the true human health risks associated with exposure to the COPCs, since many chemicals may act on different target organs. If the target hazard index (*HI*) is exceeded, a segregation of the *HI* by toxicological endpoint will be considered. If segregation by toxicological endpoint is used, chemical groupings by endpoint will be assigned with approval by Ecology and EPA. In addition to multiple chemicals, receptors will be assumed to be exposed to the multiple pathways identified in Table 7-1.

The simplified equation for calculating a generic *HI* is presented below:

$$HI = \sum HQ_i \qquad \text{(HHRAP Eq. 7-6)}$$

where:

$HI$ = hazard index for a specific exposure pathway

$HQ_i$ = hazard quotient for COPC *i*

An *HI* threshold level of 0.25 will be used in the SLRA to provide a conservative evaluation of hazard.

### 7.4.3 Risk Characterization for Acute Effects

Acute health hazards are characterized using an acute hazard quotient (*AHQ*). The *AHQ* is the ratio of the one-hour acute air concentration to the appropriate acute reference value as shown below:

COPCs: $AHQ = C_{acute} \div AIEC$ (HHRAP Eq. 7-9)

ROPCs: $AHQ = C_{acute} \div AREC_M$ (modified HHRAP Eq. 7-9)

where:

$C_{acute}$ = one-hour acute air concentration (mg/m$^3$ or µCi/m$^3$)

$AHQ$ = acute hazard quotient (unitless)

$AIEC$ = acute inhalation exposure criteria (Section 7.2.1.2) (mg/m$^3$)

$AREC_M$ = acute radionuclide exposure criteria (Section 7.2.4.4) (µCi/cm$^3$)

As defined by the above equation, an *AHQ* of 1 or less indicates that the maximum one-hour air concentration is below the reference value. An *AHQ* threshold level of 1 is used to provide a conservative evaluation of hazard per EPA (CCN 063809).

## 7.5 Uncertainty in Human Health Risk Assessment

This section provides an overview of some of the primary sources of uncertainty unique to the HHRA. Uncertainties associated with the COPC and ROPC selection, emission rates, and environmental modeling, described in previous sections, also contribute to the uncertainty in the HHRA. As described in Section 10 of this RAWP, an uncertainty assessment will be included in the SLRA to evaluate the contributors to, and potential impact of, uncertainty in the risk assessment.

### 7.5.1 Uncertainty in Exposure Assessment

Sources of uncertainty in the exposure assessment include:

- Contaminant concentrations in exposure media
- Land-use assumptions
- Selection of representative human receptor populations and exposure parameter values
- The makeup of the waste stream, the formation of PICs, and the modeling of emissions

Each of these sources of uncertainty in the exposure assessment is described briefly below.

**Contaminant Concentrations in Exposure Media**

The uncertainty associated with estimating exposure concentrations in air, soil, surface water, sediment, and plants is described in Section 6.8 of this RAWP. The HHRA also includes ingestion of animal products (such as beef and eggs). The uptake models used to estimate contaminant concentrations in animal products are highly uncertain. Conservative assumptions used to compensate for this uncertainty include the assumption that animals feed exclusively on contaminated plants and the use of conservative uptake factors, including some mass-limited uptake factors.

**Land-Use Assumptions**

Land use can change at any time; therefore, even defining current land use (i.e., during WTP operations) has some uncertainty associated with it, and defining future land use (i.e., after WTP shutdown) has even greater uncertainty. To compensate for this uncertainty, receptors are assumed to be present at the locations of maximum concentration regardless of actual land use at those locations and as determined by combining air model concentration and deposition rates according to Section 6.1. For example, a current residential scenario will be evaluated at the Hanford offsite location regardless of whether or not this location is presently in residential use or coincides with an air model grid point.

**Selection of Representative Receptor Populations and Exposure Parameter Values**

Every individual is unique, with different activity patterns (e.g., amount of time spent at home or work) and different physiologic characteristics (e.g., body weight). Therefore, modeling broad categories of receptors (e.g., resident) introduces uncertainty because (1) a limited number of general receptor categories are evaluated, and (2) exposure parameters are assigned within each receptor category to represent the activity patterns and physiologic characteristics of that receptor type. To compensate for this uncertainty, receptor types representing the highest potential for exposure are evaluated in the risk assessment, and these receptors are modeled using upper-bound assumptions to describe their activity patterns. For example, evaluation of a resident who is assumed to be at home 24 hours per day, 350 days per year at the point of maximum contaminant concentration will overestimate the risk to many other

receptor types not included in the quantitative risk assessment, such as a school child at the same location who may be at school 8 hours per day, 180 days per year.

While exposure assumptions used in the HHRA are designed to overestimate risk, some assumptions could underestimate the risk because of an individual having higher exposure than accounted for in the exposure assessment, or exposures via pathways (such as dermal) that have been identified by EPA in numerous risk assessments as being insignificant contributors to risk from thermal treatment facilities.

**Makeup of the Waste, Formation of PICs and Modeling of Emissions**

The Hanford tank waste is composed of several hundred organic and inorganic compounds including radionuclides. The tank chemistry is complex and varies from tank to tank. There are a number of technical challenges that complicate or interfere with getting a representative sample and analyzing that sample accurately. As such, there are uncertainties with assumed waste composition and properties. Likewise, the thermal processes within the WTP (e.g., the LAW and HLW melters and thermal catalytic oxidation unit) that produce products of incomplete combustion are complex reactions that are influenced by such variables as temperature, catalytic effects, organic precursors, and residence time. It is difficult to predict the types and concentrations of PICs produced by the varying feed streams that will be processed by the WTP. As such, the emissions modeling that has been performed in support of risk assessment carries uncertainties with regard to feed assumptions, PIC generation, in addition to equipment performance, and contaminant behavior. The use of conservative assumptions relating to these and other emissions related parameters are intended to overestimate the likely emissions from the WTP and compensate for the uncertainties with the feed and PIC formation mechanisms.

### 7.5.2 Uncertainty in Toxicity Assessment

Sources of uncertainty in the toxicity assessment include uncertainties surrounding the following:

- Toxicity values (*RfD*s and *RfC*s, *CSF*s and *URF*s)
- Cancer weight-of-evidence classifications
- Toxicity value data gaps
- Route-to-route extrapolations

Each of these sources of uncertainty in the toxicity assessment is described briefly below.

**Toxicity Values**

Because most of the toxicity values (*RfD*s and *RfC*s, *CSF*s and *URF*s) are based on laboratory exposures of animals, actual effects of environmental exposures to humans is unknown. Therefore, EPA-derived toxicity values are designed to provide an upper-bound estimate of risk (e.g., by incorporating numerous uncertainty factors). However, previous or concurrent exposures from sources other than the WTP are not considered in the EPA toxicity values for most chemicals. For example, all humans have been exposed to dioxins and PCBs and have some body burden associated with them. The additional exposures and their potential to increase body burdens of these chemicals due to the plant emissions are evaluated independent of existing risks, or as an incremental increase. It is not known how much such incremental exposures actually affect an individual's potential to suffer adverse effects from these exposures. This is regarded as a nonconservative uncertainty in the overall risk assessment.

**Cancer Weight-Of-Evidence Classification**

Uncertainty in the cancer weight-of-evidence classification will be considered in the HHRA by evaluating all Class A (human carcinogen), Class B (probable human carcinogen), and Class C (possible human carcinogen) chemicals as carcinogens.

**Toxicity Value Data Gaps**

The lack of toxicity data for some COPCs will contribute to an underestimation of risk if these chemicals are present in the emissions and are toxic to humans at the concentration emitted.

**Route-To-Route Extrapolations**

Uncertainties are associated with the estimation of dermal toxicity values from oral values. In addition, to address the toxicity value data gap issue, further assessment of constituent inhalation risk may be performed if oral toxicity values are available. If no *URF* or *RfC* are available, they can be derived from the $CSF_{inh}$ or $RfD_{inh}$ (respectively) using the conversion in WAC 173-340-708(7)b, which directs the risk assessor to take into account, where available, the respiratory deposition and absorption characteristics of the gases and inhaled particles.

### 7.5.3 Uncertainty in Risk Characterization

The risk characterization combines the results of the exposure assessment and toxicity assessment; therefore, all of the uncertainty in these two steps, as well as the steps prior to the exposure assessment (e.g., fate and transport modeling), contributes to the uncertainty in the risk characterization. Additional uncertainty in the risk characterization step surrounds the practice of summing cancer risks and noncancer hazard results across all chemicals and exposure pathways, regardless of the mode of action, as described below.

The assumption of strict additivity of chemical carcinogens that will be used in the SLRA assumes that (1) intakes of individual chemicals are small, and (2) there is no interaction among chemicals (i.e., no synergism or antagonism). The assumption of strict additivity of cancer risk from radionuclides is much less uncertain than for chemicals because the mode of action is the same for all radionuclides.

Multiple chemical exposures to noncarcinogens can result in synergism, antagonism, and/or additivity of biological responses when the chemicals act on similar target organs, or when the chemicals are metabolized by the same enzymatic pathways. The assumption of additivity will be used in the SLRA and is likely to overestimate the true human health hazards associated with exposure to the COPCs, since many chemicals may act on different target organs.

In addition to multiple chemicals, receptors may be exposed through more than one pathway. As the EPA (1989) notes:

> There are two steps required to determine whether risks or hazard indices for two or more pathways should be combined for a single exposed individual or group of individuals. The first is to identify reasonable exposure pathway combinations. The second is to examine whether it is likely that the <u>same</u> individuals would consistently face the "reasonable maximum exposure" for more than one pathway.

To maintain the conservative bias of the risk assessment, it is assumed that each receptor is exposed to all COPCs and ROPCs by all pathways.

### 7.5.4 Summary of Uncertainty

Human health risk assessment is a multi-step process and uncertainty is introduced at all steps of the process, including COPC and ROPC selection, estimating emission rates, environmental modeling, exposure assessment, toxicity assessment, and risk characterization. Whenever possible, conservative assumptions are used to compensate for uncertainties so that the final estimate of risk represents an overestimate, rather than an underestimate, of risk to actual receptor populations.

As described in Section 10 of this RAWP, an uncertainty assessment will be included in the SLRA to evaluate the contributors to, and potential impact of, uncertainty in the risk assessment. The purpose of the uncertainty assessment is to identify and discuss areas of uncertainty associated with the quantitative estimates of risk for the WTP. This discussion serves to place the risk estimates in proper perspective to allow fully informed risk management decisions.

## 7.6 Summary for Human Health Risk Assessment

Risks and hazards to human health from the potential emission of COPCs and ROPCs result from (1) exposure to the COPC or ROPC, and (2) the toxicity of the COPC or ROPC. The screening HHRA utilizes estimated emission rates (Section 5) and results of the fate and transport modeling (Section 6) to calculate potential human exposure to COPCs and ROPCs. This exposure information is combined with toxicity data to estimate the potential for adverse effects to human populations in the vicinity of the WTP.

The PRA will use conservative exposure assumptions to compensate for the high level of uncertainty associated with conducting a risk assessment for a facility that is still in the design phase. The PRA will include a qualitative uncertainty analysis.

The COPCs or ROPCs that exceed risk goals in the PRA will be revisited to determine whether unrealistic parameters were assigned to them in the PRA. If the analysis conducted in the PRA is considered reasonable, it may be necessary to alter operational characteristics of the WTP in order to reduce emissions to be within acceptable risk limits.

The FRA will include estimated emissions based on engineering calculations (e.g., PT Facility system emissions and vapor-phase organic emissions from WTP process cells) and environmental performance demonstration tests for the LAW and HLW vitrification systems. Based on the results of the environmental performance demonstration tests, the FRA may involve running new models, modeling additional chemicals, or changing model parameters. Information that will require updating in the FRA, as specified in the WTP DWP (WA7890008967), will include:

- Toxicity data current at the time of the submittal
- Compounds newly identified, or updated emissions data from current waste characterization and emission testing
- Air modeling updated to include stack gas parameters based on most current emissions testing and current WTP unit design
- Physical/transport properties of constituents current at the time of the submittal
- Process description based on current WTP unit design
- Emissions data and all supporting calculations based on current WTP unit design

- Update of receptor locations based on land use or land-use zoning changes, if any

If the risk goals are exceeded in the PRA or the FRA additional site specific data will be evaluated for use in the assessments, subject to Ecology approval.

## 7.7 References

### 7.7.1 Project Documents

CCN 019247, *Washington Department of Ecology/Tetra Tech Em Inc. Input On Issues Associated with the Final Work Plan for Screening Level Risk Assessment for the RPP-WTP (RPT-W375-EN00001, Rev. 1) (Risk Assessment Work Plan),* Memorandum documenting E-mail communications from Jerry Yokel, Washington State Department of Ecology, and Tetra Tech letter to Jerry Yokel, Washington State Department of Ecology, 27 March 2001.

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**Table 7-1 Human Receptor Populations and Exposure Pathways for the PRA for the Hanford WTP**

| Receptor | Location | Exposure Pathways | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Inhalation Routes | | | External Radiation | | Ingestion | | | | | | | | | | | Absorp-tion | |
| | | Emissions[a] | Resuspended Soil | Sweat Lodge Vapors[b] | Radionuclides in air | Radionuclides in soil | Soil | Produce | Beef | Milk | Chicken & Eggs | Pork | Wild Produce | Wild Game | Wildfowl & Eggs | Fish[b] | Drinking Water[b] | Sweat Lodge Vapors[b] | Breast milk[c] |
| **Plausible Exposure Scenarios (evaluated in current and future timeframes)** | | | | | | | | | | | | | | | | | | | |
| Hanford site industrial worker (adult) | Works at onsite ground maximum | x | x | | x | x | x | | | | | | | | | | | | x |
| | Resides at Hanford offsite location | x | x | | x | x | x | | | | | | | | | | | | |
| | Consumes homegrown produce (offsite) | | | | | | | x | | | | | | | | | | | |
| | Consumes water (Columbia River max) | | | | | | | | | | | | | | | | x | | |
| Resident (adult and child) | Resides at Hanford offsite location | x | x | | x | x | x | | | | | | | | | | | | x |
| | Consumes homegrown produce (offsite) | | | | | | | x | | | | | | | | | | | |
| | Consumes water (Columbia River max) | | | | | | | | | | | | | | | | x | | |
| Resident Subsistence American Indian (adult and child) | Resides at Hanford offsite location | x | x | | x | x | x | | | | | | | | | | | | x |
| | Consumes wild produce/game (hunter/gatherer area) | | | | | | | | | | | | x | x | x | x | | | |
| | Consumes water (Columbia River max) | | | x | | | | | | | | | | | | | x | x | |
| **Worst-Case Exposure Scenario (evaluated in current and future timeframes)** | | | | | | | | | | | | | | | | | | | |
| Resident subsistence farmer (adult and child) | Resides at Hanford offsite location | x | x | | x | x | x | | | | | | | | | | | | x |
| | Consumes homegrown produce/livestock (offsite) | | | | | | | x | x | x | x | x | | | | | | | |
| | Consumes water (Columbia River max) | | | | | | | | | | | | | | | | x | | |
| Resident subsistence fisher (adult and child) | Resides at Hanford offsite location | x | x | | x | x | x | | | | | | | | | | | | x |
| | Consumes homegrown produce (offsite) | | | | | | | x | | | | | | | | | | | |
| | Consumes fish and water (Columbia River max) | | | | | | | | | | | | | | | x | x | | |
| Acute exposure | Acute maximum | x | | | x | | | | | | | | | | | | | | |

**Table 7-1 Human Receptor Populations and Exposure Pathways for the PRA for the Hanford WTP**

| Receptor | Location | Exposure Pathways | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Inhalation Routes | | | External Radiation | | Ingestion | | | | | | | | | | | Absorp-tion | |
| | | Emissions[a] | Resuspended Soil | Sweat Lodge Vapors[b] | Radionuclides in air | Radionuclides in soil | Soil | Produce | Beef | Milk | Chicken & Eggs | Pork | Wild Produce | Wild Game | Wildfowl & Eggs | Fish[b] | Drinking Water[b] | Sweat Lodge Vapors[b] | Breast milk[c] |
| **Alternate Exposure Scenarios (evaluated in current and future timeframes)** | | | | | | | | | | | | | | | | | | | |
| Alternate Resident subsistence American Indian, scenario #1 (adult and child) | Resides at Hanford offsite location | X | X | | X | X | X | | | | | | | | | | | | X |
| | Visits Gable Mountain maximum | X | X | | X | X | X | | | | | | | | | | | | |
| | Consumes wild produce/game (hunter/gatherer area) | | | | | | | | | | | | X | X | X | X | | | |
| | Consumes fish and water (Columbia River max) | | | X | | | | | | | | | | | | | X | X | |
| Alternate Resident subsistence American Indian, scenario #2 (adult and child) | Resides at Hanford offsite location | X | X | | X | X | X | | | | | | | | | | | | X |
| | Consumes homegrown produce/livestock (offsite) | | | | | | | X | X | X | X | X | | | | | | | |
| | Consumes wild produce/game (hunter/gatherer area) | | | | | | | | | | | | X | X | X | X | | | |
| | Consumes fish and water (Columbia River max) | | | X | | | | | | | | | | | | | X | X | |
| | Consumes water (Columbia River max) | | | X | | | | | | | | | | | | | X | X | |

X = complete exposure pathway for receptor.

[a] Includes direct inhalation of vapor phase and particulate emissions. Applicable to current timeframe only (during WTP emissions).

[b] Pathway attributable to exposure to water/fish from the Columbia River maximum. Applicable to current timeframe only (during WTP emissions); subsequent to WTP operation, deposited constituents are transported down river.

[c] Includes nursing infant assessment - maternal exposures indicated.

**Table 7-2 Hanford Site Industrial Worker Exposure Parameters**

| Parameter | Description | Units | Onsite Worker | Source or Reference |
|---|---|---|---|---|
| EF | Exposure frequency | days/yr | 350 | CCN 064331, EPA to WTP Regarding Human Exposure Scenarios and Exposure Parameters, Personal communication between SAIC and US Environmental Protection Agency, Region 10, at a meeting held on 8 and 9 October 2002 in Seattle, Washington, USA. |
| $EF_{work}$ | Exposure frequency at work | days/yr | 250 | CCN 063816, EPA to WTP Regarding Exposure Durations for Worker, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 19 December 2002. |
| $EF_{retire}$ | Exposure frequency during retirement | days/yr | 350 | HHRAP Table C-1-8 |
| ED | Exposure duration | yr | 20 | CCN 063816, EPA to WTP Regarding Exposure Durations for Worker, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 19 December 2002. |
| $ED_{work}$ | Exposure duration at work | yr | 20 | |
| $ED_{retire}$ | Exposure duration during retirement | yr | 10 | |
| ET | Exposure time | hr/day | 24 | CCN 063807, EPA to WTP Regarding Exposure Scenarios and Exposure, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 13 June 2002. |
| $ET_{work}$ | Exposure time at work | hr/day | 8 | |
| BW | Body weight | kg | 70 | HHRAP Appendix C |
| $AT_{N\ inhal}$ | Inhalation Averaging time for noncarcinogens | yr | 20 | CCN 063816, EPA to WTP Regarding Exposure Durations for Worker, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 19 December 2002. |
| $AT_N$ | Averaging time for noncarcinogens | yr | 20 | |
| $AT_{N\ retire}$ | Averaging time for noncarcinogens during retirement | yr | 10 | |

**Table 7-2 Hanford Site Industrial Worker Exposure Parameters**

| Parameter | Description | Units | Onsite Worker | Source or Reference |
|---|---|---|---|---|
| IR | Inhalation rate | $m^3$/hr | 0.833 | CCN 063805, EPA to WTP Regarding Exposure Parameters, E-mail communication from Cathy Massimino, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 4 September 2002. |
| $IR_{work}$ | Inhalation rate at work | $m^3$/hr | 1.5 | |
| $CR_{soil}$ | Ingestion rate for soil | kg/day | 0.0001 | CCN 064331, EPA to WTP Regarding Human Exposure Scenarios and Exposure Parameters, Personal communication between SAIC and US Environmental Protection Agency, Region 10, at a meeting held on 8 and 9 October 2002 in Seattle, Washington, USA. |
| $CR_{soil\ work}$ | Ingestion rate for soil at work | kg/day | 0.0002 | |
| $CR_{dw}$ | Ingestion rate for drinking water | L/day | 2 | CCN 063813, EPA to WTP Regarding Drinking Water Ingestion Rate for Worker, E-mail communication from Marcia Bailey, US Environmental Protection Agency EPA Region 10 to Sharon Robers, SAIC, 10 January 2003. |
| $CR_{dw\ offwork}$ | Ingestion rate for drinking water after work | L/day | 1 | |
| $CR_{ag}$ | Consumption rate: aboveground domestic produce | kg/kg-day FW | 0.00032 | HHRAP Table C-1-2 (Resident) |
| $CR_{bg}$ | Consumption rate: belowground produce | kg/kg-day FW | 0.00014 | |
| $CR_{pp}$ | Consumption rate: protected produce | kg/kg-day FW | 0.00061 | |

**Table 7-2 Hanford Site Industrial Worker Exposure Parameters**

| Parameter | Description | Units | Onsite Worker | Source or Reference |
|---|---|---|---|---|
| $ET_o$ | Exposure time factor for outdoor exposure to ROPCs in soil (non-work days) | unitless | 0.060 | EPA. 1997. Exposure Factors Handbook, EPA/600/P-95/002Fa. Office of Research and Development, US Environmental Protection Agency, Washington, DC, USA. (for after-workday exposures, the exposure time factor for indoor exposure is adjusted for the 8 hrs spent at work [0.607 = 0.94 - 8/24]) |
| $ET_i$ | Exposure time factor for indoor exposure to ROPCs in soil (non-work days) | unitless | 0.940 | |
| $ET_{o\ offwork}$ | Exposure time factor for outdoor exposure to ROPCs in soil after work | unitless | 0.060 | |
| $ET_{i\ offwork}$ | Exposure time factor for indoor exposure to ROPCs in soil after work | unitless | 0.607 | |
| $ET_{o\ work}$ | Exposure time factor for outdoor exposure to ROPCs in soil at work | unitless | 0.167 | Exposure is for 4 hr/day indoor, and 4 hr/day outdoor while at work. See RAWP Section 7.1.6.1. |
| $ET_{i\ work}$ | Exposure time factor for indoor exposure to ROPCs in soil at work | unitless | 0.167 | |
| **Nursing Infant Exposure Parameters** | | | | |
| $ED_{infant}$ | Exposure duration | yrs | 1 | HHRAP Table C-3-2 |
| $ED_{maternal}$ | Maternal exposure duration | yr | 20 | CCN 063816, EPA to WTP Regarding Exposure Durations for Worker, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 19 December 2002. |
| $IR_{milk}$ | Ingestion rate: breast milk | L/day | 0.688 | HHRAP Table C-3-2 |

**Table 7-2 Hanford Site Industrial Worker Exposure Parameters**

| Parameter | Description | Units | Onsite Worker | Source or Reference |
|---|---|---|---|---|
| $BW_{infant}$ | Body weight | kg | 7.2 | CCN 063806, EPA to WTP Regarding Infant Body Weight, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 31 October 2002 (2:26 p.m.). |
| $AT_{infant}$ | Averaging time for carcinogens | yr | 1 | HHRAP Table C-3-2 |

HHRAP: EPA. 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. (http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm).

**Table 7-3 Resident Exposure Parameters**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| EF | Exposure frequency | days/yr | 350 | 350 | HHRAP Tables C-1-7 and C-1-8 |
| ED | Exposure duration | yr | 30 | 6 | HHRAP Tables C-1-7 and C-1-8 |
| ET | Exposure time | hr/day | 24 | 24 | Assumed (conservative) value |
| BW | Body weight | kg | 70 | 15 | HHRAP Appendix C |
| $AT_C$ | Averaging time for carcinogens | yr | 70 | 70 | HHRAP Table C-1-7 |
| $AT_{N\ inhal}$ | Inhalation Averaging time for noncarcinogens | yr | 30 | 6 | HHRAP Table C-1-8 |
| $AT_N$ | Averaging time for noncarcinogens | yr | 30 | 6 | |
| IR | Inhalation rate | $m^3$/hr | 0.833 | 0.417 | CCN 063805, EPA to WTP Regarding Exposure Parameters, E-mail communication from Cathy Massimino, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 4 September 2002. |
| $CR_{soil}$ | Ingestion rate for soil | kg/day | 0.0001 | 0.0002 | HHRAP Table C-1-1 |
| $CR_{dw}$ | Ingestion rate for drinking water | L/day | 2 | 1 | CCN 063807, EPA to WTP Regarding Exposure Scenarios and Exposure, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 13 June 2002. |
| $CR_{ag}$ | Consumption rate: aboveground domestic produce | kg/kg-day FW | 0.00032 | 0.00077 | HHRAP Table C-1-2 |
| $CR_{bg}$ | Consumption rate: belowground produce | kg/kg-day FW | 0.00014 | 0.00023 | |
| $CR_{pp}$ | Consumption rate: protected produce | kg/kg-day FW | 0.00061 | 0.0015 | |
| $ET_o$ | Exposure time factor for outdoor exposure to ROPCs in soil | unitless | 0.06 | 0.23 | EPA. 1997. Exposure Factors Handbook, EPA/600/P-95/002Fa. Office of Research and Development, US Environmental Protection Agency, |

**Table 7-3 Resident Exposure Parameters**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| $ET_i$ | Exposure time factor for indoor exposure to ROPCs in soil | unitless | 0.94 | 0.77 | Washington, DC, USA.(Table 15-176; 1.5 hr/day outdoor occupancy [adult], 5.6 hr/day (wt. ave.) outdoor occupancy [child]) |
| **Nursing Infant Exposure Parameters** | | | | | |
| $ED_{infant}$ | Exposure duration | yr | na | 1 | HHRAP Table C-3-2 |
| $ED_{maternal}$ | Maternal exposure duration | yr | 30 | na | HHRAP Table C-3-1 |
| $IR_{milk}$ | Ingestion rate: breast milk | L/day | na | 0.688 | HHRAP Table C-3-2 |
| $BW_{infant}$ | Body weight | kg | na | 7.2 | CCN 063806, EPA to WTP Regarding Infant Body Weight, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 31 October 2002 (2:26 p.m.). |
| $AT_{infant}$ | Averaging time for carcinogens | yr | na | 1 | HHRAP Table C-3-2 |

na = not applicable

HHRAP: EPA. 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. (http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm).

**Table 7-4 Resident Subsistence Farmer Exposure Parameters**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| EF | Exposure frequency | days/yr | 350 | 350 | HHRAP Tables C-1-7 and C-1-8 |
| ED | Exposure duration | yr | 40 | 6 | HHRAP Tables C-1-7 and C-1-8 |
| ET | Exposure time | hr/day | 24 | 24 | Assumed (conservative) value |
| BW | Body weight | kg | 70 | 15 | HHRAP Appendix C |
| $AT_C$ | Averaging time for carcinogens | yr | 70 | 70 | HHRAP Table C-1-7 |
| $AT_{N\ inhal}$ | Inhalation Averaging time for noncarcinogens | yr | 40 | 6 | HHRAP Table C-1-8 |
| $AT_N$ | Averaging time for noncarcinogens | yr | 40 | 6 | |
| IR | Inhalation rate | $m^3$/hr | 0.833 | 0.417 | CCN 063805, EPA to WTP Regarding Exposure Parameters, E-mail communication from Cathy Massimino, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 4 September 2002. |
| $CR_{soil}$ | Ingestion rate for soil | kg/day | 0.0001 | 0.0002 | HHRAP Table C-1-1 |
| $CR_{dw}$ | Ingestion rate for drinking water | L/day | 2 | 1 | CCN 063807, EPA to WTP Regarding Exposure Scenarios and Exposure, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 13 June 2002. |
| $CR_{ag}$ | Consumption rate: aboveground domestic produce | kg/kg-day FW | 0.00047 | 0.00113 | HHRAP Table C-1-2 |
| $CR_{bg}$ | Consumption rate: belowground produce | kg/kg-day FW | 0.00017 | 0.00028 | |
| $CR_{pp}$ | Consumption rate: protected produce | kg/kg-day FW | 0.00064 | 0.00157 | |
| $CR_{domestic\ fowl}$ | Consumption rate domestic chicken | kg/kg-day FW | 0.00066 | 0.00045 | HHRAP Table C-1-3 |
| $CR_{beef}$ | Consumption rate: beef | kg/kg-day FW | 0.00122 | 0.00075 | HHRAP Table C-1-3 |
| $CR_{pork}$ | Consumption rate: pork | kg/kg-day FW | 0.00055 | 0.00042 | |
| $CR_{eggs}$ | Consumption rate: eggs | kg/kg-day FW | 0.00075 | 0.00054 | |

**Table 7-4 Resident Subsistence Farmer Exposure Parameters**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| $CR_{milk}$ | Consumption rate: milk | kg/kg-day FW | 0.01367 | 0.02268 | HHRAP Table C-1-3 |
| $ET_o$ | Exposure time factor for outdoor exposure to ROPCs in soil | unitless | 0.42 | 0.42 | EPA. 1997. Exposure Factors Handbook, EPA/600/P-95/002Fa. Office of Research and Development, US Environmental Protection Agency, Washington, DC, USA.(Table 15-112, 90th percentile for all, 600 minutes outdoor occupancy) |
| $ET_i$ | Exposure time factor for indoor exposure to ROPCs in soil | unitless | 0.58 | 0.58 | |
| **Nursing Infant Exposure Parameters** | | | | | |
| $ED_{infant}$ | Exposure duration | yr | na | 1 | HHRAP Table C-3-2 |
| $ED_{maternal}$ | Maternal exposure duration | yr | 40 | na | HHRAP Table C-3-1 |
| $IR_{milk}$ | Ingestion rate: breast milk | L/day | na | 0.688 | HHRAP Table C-3-2 |
| $BW_{infant}$ | Body weight | kg | na | 7.2 | CCN 063806, EPA to WTP Regarding Infant Body Weight, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 31 October 2002 (2:26 p.m.). |
| $AT_{infant}$ | Averaging time for carcinogens | yr | na | 1 | HHRAP Table C-3-2 |

na = not applicable

HHRAP: EPA. 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. (http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm).

**Table 7-5 Resident Subsistence Fisher Exposure Parameters**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| EF | Exposure frequency | days/yr | 350 | 350 | HHRAP Tables C-1-7 and C-1-8 |
| ED | Exposure duration | yrs | 30 | 6 | HHRAP Tables C-1-7 and C-1-8 |
| ET | Exposure time | hr/day | 24 | 24 | Assumed (conservative) value |
| BW | Body weight | kg | 70 | 15 | HHRAP Appendix C |
| $AT_C$ | Averaging time for carcinogens | yr | 70 | 70 | HHRAP Table C-1-7 |
| $AT_{N\ inhal}$ | Inhalation Averaging time for noncarcinogens | yr | 30 | 6 | HHRAP Table C-1-8 |
| $AT_N$ | Averaging time for noncarcinogens | yr | 30 | 6 | |
| IR | Inhalation rate | $m^3$/hr | 0.833 | 0.417 | CCN 063805, EPA to WTP Regarding Exposure Parameters, E-mail communication from Cathy Massimino, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 4 September 2002. |
| $CR_{soil}$ | Ingestion rate for soil | kg/day | 0.0001 | 0.0002 | HHRAP Table C-1-1 |
| $CR_{dw}$ | Ingestion rate for drinking water | L/day | 2 | 1 | CCN 063807, EPA to WTP Regarding Exposure Scenarios and Exposure, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 13 June 2002. |
| $CR_{ag}$ | Consumption rate: aboveground domestic produce | kg/kg-day FW | 0.00032 | 0.00077 | HHRAP Table C-1-2 |
| $CR_{bg}$ | Consumption rate: belowground produce | kg/kg-day FW | 0.00014 | 0.00023 | |
| $CR_{pp}$ | Consumption rate: protected produce | kg/kg-day FW | 0.00061 | 0.0015 | |
| $CR_{fish}$ | Consumption rate: fish | kg/kg-day FW | 0.00125 | 0.00088 | HHRAP Table C-1-4 |

**Table 7-5 Resident Subsistence Fisher Exposure Parameters**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| $ET_o$ | Exposure time factor for outdoor exposure to ROPCs in soil | unitless | 0.42 | 0.42 | EPA. 1997. Exposure Factors Handbook, EPA/600/P-95/002Fa. Office of Research and Development, US Environmental Protection Agency, Washington, DC, USA.(Table 15-112, 90th percentile for all, 600 minutes outdoor occupancy) |
| $ET_i$ | Exposure time factor for indoor exposure to ROPCs in soil | unitless | 0.58 | 0.58 | |
| **Nursing Infant Exposure Parameters** | | | | | |
| $ED_{infant}$ | Exposure duration | yr | na | 1 | HHRAP Table C-3-2 |
| $ED_{maternal}$ | Maternal exposure duration | yr | 30 | na | HHRAP Table C-3-1 |
| $IR_{milk}$ | Ingestion rate: breast milk | L/day | na | 0.688 | HHRAP Table C-3-2 |
| $BW_{infant}$ | Body weight | kg | na | 7.2 | CCN 063806, EPA to WTP Regarding Infant Body Weight, E-mail communication from Marcia Bailey, US Environmental Protection Agency Region 10, to Sharon Robers, SAIC, 31 October 2002 (2:26 p.m.). |
| $AT_{infant}$ | Averaging time for carcinogens | yr | na | 1 | HHRAP Table C-3-2 |

na = not applicable

HHRAP: EPA. 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. (http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm).

**Table 7-6 DOE Resident Subsistence American Indian Exposure Parameters**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| EF | Exposure frequency | days/yr | 365 | 365 | Appendix D (p. D-296) of EIS-0189 ("continuous occupancy") |
| ED | Exposure duration | yr | 70 | 6 | For adults, the equation for HQsn on p. Q-14 of EIS-0391: Averaging Time (25,550 days). Per the HHRAP, a exposure duration of 6 yrs is assumed for children. |
| ET | Exposure time | hr/day | 24 | 24 | Appendix D (p. D-296) of EIS-0189 ("continuous occupancy") |
| $ET_{sw}$ | Exposure time for sweat lodge | hr/day | 2 | 2 | Table Q-14 of EIS-0391 |
| BW | Body weight | kg | 70 | 16 | Table D.2.1.3 of EIS-0189 |
| $AT_C$ | Averaging time for carcinogens | yr | 70 | 70 | Eqn for HQsn on p. Q-14 of EIS-0391: Averaging Time |
| $AT_{N\ inhal}$ | Inhalation Averaging time for noncarcinogens | yr | 40 | 6 | Duration of WTP operation; 40 yr duration applies to adult inhalation and water exposures (including fish consumption), 70 yr duration applies to adult soil and ingestion (excluding fish) related exposures. Child averaging time is limited to 6 yrs. See exposure duration source/reference. |
| IR | Inhalation rate | $m^3$/hr | 0.959 | 0.625 | Table Q-9 of EIS-0391 (adult); Table D.2.1.3 of EIS-0189 (child) |
| $CR_{soil}$ | Ingestion rate for soil | kg/day | 0.0001 | 0.0002 | Table Q-10 of EIS-0391 (adult); Table D.2.1.3 of EIS-0189 (child). The value in Table Q-10 is a weighted average for the adult and child. Backing out the child consumption rate in EIS-0189 yields and adult consumption rate of approximately 100 mg/day. |
| $CR_{dw}$ | Ingestion rate for drinking water | L/day | 4 | 1.5 | Table J-23 of EIS-0391 (adult); Table D.2.1.3 of EIS-0189 (child) |

**Table 7-6 DOE Resident Subsistence American Indian Exposure Parameters**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| $CR_{ag\ wild}$ | Consumption rate: aboveground wild produce | kg/kg-day FW | 0.0025 | 0.0038 | Table J-23 & Q-10 of EIS-0391: Leafy vegetable consumption rate (adult). For the child, assume a daily mass (vegetable) consumption of 34% of the adult based on comparison of data in CSEFH (weighted average consumption, mean values, ages 3 through 6) and EFH (weighted average consumption, mean values, ages 7 through 70). |
| $CR_{ag}$ | Consumption rate: aboveground domestic produce | kg/kg-day FW | - | - | No domestic agriculture consumption reported/available. |
| $CR_{bg}$ | Consumption rate: belowground produce | kg/kg-day FW | - | - | No belowground agriculture consumption reported/available. |
| $CR_{pp}$ | Consumption rate: protected produce | kg/kg-day FW | 0.013 | 0.027 | Table J-23 & Q-10 of EIS-0391: Fruit, vegetable, and grain consumption rate (kg/yr) (adult). For the child, assume a daily mass (fruit) consumption of 48% of the adult based on comparison of data in CSEFH (weighted average consumption, mean values, ages 3 through 6) and EFH (weighted average consumption, mean values, ages 7 through 70). |
| $CR_{fowl}$ | Consumption rate wild fowl | kg/kg-day FW | - | - | No wild fowl consumption reported/available. |
| $CR_{domestic\ fowl}$ | Consumption rate domestic chicken | kg/kg-day FW | - | - | No domestic fowl consumption reported/available. |
| $CR_{game}$ | Consumption rate: wild game | kg/kg-day FW | 0.0060 | 0.013 | Table Q-10 of EIS-0391: Meat and poultry consumption (only a deer exposure equation is provided in App. Q so its assumed the rate provided here is just for game) (adult). For the child, assume a daily mass (beef) consumption of 48% of the adult based on comparison of data in CSEFH (weighted average consumption, mean values, ages 3 through 6) and EFH (weighted average consumption, mean values, ages 7 through 70). |
| $CR_{game\ organs}$ | Consumption rate: game organs | kg/kg-day FW | - | - | No organ consumption reported/available. |

**Table 7-6 DOE Resident Subsistence American Indian Exposure Parameters**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| $CR_{beef}$ | Consumption rate: beef | kg/kg-day FW | - | - | No domestic livestock consumption reported/available. |
| $CR_{pork}$ | Consumption rate: pork | kg/kg-day FW | - | - | No domestic livestock consumption reported/available. |
| $CR_{eggs}$ | Consumption rate: eggs | kg/kg-day FW | 0.00074 | 0.0022 | Table J-23 of EIS-0391 (adult).For the child, assume a daily mass (eggs) consumption of 67% of the adult based on comparison of data in CSEFH (weighted average consumption, mean values, ages 3 through 6) and EFH (weighted average consumption, mean values, ages 7 through 70). |
| $CR_{fish}$ | Consumption rate: fish | kg/kg-day FW | 0.0088 | 0.0101 | Table J-23 and Q.2.4.2, 3rd para. of EIS-0391 (adult).<br>For the child, assume a daily mass (total fish) consumption of 26% of the adult based on comparison of data in CSEFH (weighted average consumption, mean values, ages 3 through 6) and EFH (weighted average consumption, mean values, ages 7 through 70). |
| $CR_{fish\ organs}$ | Consumption rate: fish organs | kg/kg-day FW | - | - | No organ consumption reported/available. |
| $CR_{milk}$ | Consumption rate: milk | kg/kg-day FW | 0 | 0 | Table Q-3 of EIS-0391 |
| $ET_o$ | Exposure time factor for outdoor exposure to ROPCs in soil | unitless | 0.12 | 0.12 | Table Q-5 of EIS-0391 (note: the EIS assumes that for a portion of the time the receptor was not present at the location.) |
| $ET_i$ | Exposure time factor for indoor exposure to ROPCs in soil | unitless | 0.66 | 0.66 | Table Q-5 of EIS-0391 |
| SA | Dermal Surface Area | $m^2$ | 1.8 | 0.76 | EFH Tables 6-2 and 6-3, average of male & female 50th percentile dermal surface areas (adults).<br>CSEFH, Table 7-1, mean total body skin surface area of a 3 to <6 yr. old (for child). |
| $V_w$ | Volume of Water used in Sweat Lodge | L | 4 | 4 | Harper & Harris 1997 |

**Table 7-6 DOE Resident Subsistence American Indian Exposure Parameters**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| D | Diameter of Sweat Lodge | m | 2 | 2 | Harper & Harris 1997 |
| $T_{sl}$ | Temperature of Sweat Lodge | °F | 122 | 122 | Table Q-15 of EIS-0391 |
| **Nursing Infant Exposure Parameters** | | | | | |
| $ED_{infant}$ | Exposure duration | yrs | na | 1 | Based on CSEFH, Table 15-12, mean value rounded to the nearest whole year. |
| $ED_{maternal}$ | Maternal exposure duration | yrs | 25 | na | Assume the same as in Harris 2004. Section 2.2.3, page 15 |
| $IR_{milk}$ | Ingestion rate: breast milk | L/day | na | 0.62 | CSEFH, Table 15-1, mean for infants 6-12 mos. |
| $BW_{infant}$ | Body weight | kg | na | 9.2 | Based on CSEFH, Table 8-1, mean of 6 to <11 month old infant. |
| $AT_{infant}$ | Averaging time for carcinogens | yrs | na | 1 | Set to exposure duration. |

1 na = not applicable

CSEFH: EPA. 2008. *Child-Specific Exposure Factors Handbook*, EPA/600/R-06/096F, National Center for Environmental Assessment, Office of Research and Development, Washington DC, September 2008.

EIS-0391: US Department of Energy. 2012. *Final Tank Closure and Waste Management Environmental Impact Statement for the Hanford Site*, DOE/EIS-0391, Richland, Washington, November.

EIS-0189: US Department of Energy. 1996. *Tank Waste Remediation System, Hanford Site, Richland, Washington, Final Environmental Impact Statement*, DOE/EIS-0189, August.

EFH: EPA. 1997. *Exposure Factors Handbook, Final*, EPA/600/P-95/002F, US Environmental Protection Agency, National Center for Environmental Assessment, Office of Research and Development, Washington, DC, August 1997.

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**Table 7-7 Alternate Resident Subsistence American Indian Scenario #1**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| EF | Exposure frequency | days/yr | 365 | 365 | Harris 2004. |
| $EF_{ceremony}$ | Exposure frequency during tribal ceremonies | days/yr | 12 | 12 | Assumed value (1 day/mon. for ceremonial activities, see RAWP) |
| ED | Exposure duration | yr | 70 | 6 | Harris 2008. |
| ET | Exposure time | hr/day | 24 | 24 | Harris 2004. |
| $ET_{sw}$ | Exposure time for sweat lodge | hr/day | 1 | 1 | Harris 2004. |
| BW | Body weight | kg | 70 | 15 | Harris 2004 (adult) and HHRAP 2005 (child). |
| $AT_C$ | Averaging time for carcinogens | yr | 70 | 70 | Harris 2008. |
| $AT_{N\ inhal}$ | Inhalation Averaging time for noncarcinogens | yr | 40 | 6 | Harris 2008.<br>40 yr duration applies to inhalation and water exposures (including fish consumption), 70 yr duration applies to soil and ingestion (excluding fish) related exposures. |
| $AT_N$ | Averaging time for noncarcinogens | yr | 70 | 6 | |
| IR | Inhalation rate | $m^3$/hr | 1.04 | 0.625 | Harris 2008. |
| $CR_{soil}$ | Ingestion rate for soil | kg/day | 0.0004 | 0.0004 | Harris 2008. |
| $CR_{dw}$ | Ingestion rate for drinking water | L/day | 4 | 2 | Harris 2008 (adults only).<br>Child consuption rate assumed half of the adult's (see Rudolfi 2007). |
| $CR_{ag\ wild}$ | Consumption rate: aboveground wild produce | kg/kg-day FW | 0.0048 | 0.016 | Harris 2008 (adults only).<br>Based on 337 g/day of berries, fruits, other vegetation, greens, tea, medicines, spices, honey, sweeteners, seeds, nuts, and grains. Children's exposure factors are based on CSEFH, 1466 kcal/d diet for a 3-5 yr old, shown in Table 6-35, but scaled from the adult (see Harris 2004, Section 2.3.1). |
| $CR_{ag}$ | Consumption rate: aboveground domestic produce | kg/kg-day FW | - | - | No domestic agriculture consumption reported/available. |

**Table 7-7 Alternate Resident Subsistence American Indian Scenario #1**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| $CR_{bg}$ | Consumption rate: belowground produce | kg/kg-day FW | 0.0063 | 0.021 | Harris 2008 (adults only). Based on 440 g/day of bulbs, tubers, and roots. Children's exposure factors are based on CSEFH, 1466 kcal/d diet for a 3-5 yr old, shown in Table 6-35, but scaled from the adult (see Harris 2004, Section 2.3.1). |
| $CR_{pp}$ | Consumption rate: protected produce | kg/kg-day FW | - | - | Specific protected produce values are not reported; it is assumed aboveground consumption rates include protected produce. |
| $CR_{fowl}$ | Consumption rate wild fowl | kg/kg-day FW | 0.00089 | 0.0029 | Harris 2008 (for adults). Children's exposure factors are based on CSEFH, 1466 kcal/day diet for a 3-5 yr old, shown in Table 6-35, but scaled from the adult (see Harris 2004, Section 2.3.1). |
| $CR_{domestic\ fowl}$ | Consumption rate domestic chicken | kg/kg-day FW | - | - | No domestic fowl consumption reported/available. |
| $CR_{game}$ | Consumption rate: wild game | kg/kg-day FW | 0.0016 | 0.0050 | Harris 2008 (adults only). Per Harris 2008, organ consumption accounts for 10% of the total game consumed, thus, 90% is attributed to game meat. Children's exposure factors are based on CSEFH, 1466 kcal/d diet for a 3-5 yr old, shown in Table 6-35, but scaled from the adult (see Harris 2004, Section 2.3.1). |
| $CR_{game\ organs}$ | Consumption rate: game organs | kg/kg-day FW | 0.00018 | 0.00056 | Harris 2008 (adults only). Per Harris 2008, organ consumption accounts for 10% of the total game consumed. Children's exposure factors are based on CSEFH, 1466 kcal/d diet for a 3-5 yr old, shown in Table 6-35, but scaled from the adult (see Harris 2004, Section 2.3.1). |
| $CR_{beef}$ | Consumption rate: beef | kg/kg-day FW | - | - | No domestic livestock consumption reported/available. |
| $CR_{pork}$ | Consumption rate: pork | kg/kg-day FW | - | - | No domestic livestock consumption reported/available. |
| $CR_{eggs}$ | Consumption rate: eggs | kg/kg-day FW | - | - | Provided in the fowl consumption rate (Harris 2008). Proportion of diet is not stated. |

**Table 7-7 Alternate Resident Subsistence American Indian Scenario #1**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| $CR_{fish}$ | Consumption rate: fish | kg/kg-day FW | 0.0080 | 0.025 | Harris 2008 (adults only). Per Harris 2008, organ consumption accounts for 10% of the total fish consumed. Children's exposure factors are based on CSEFH, 1466 kcal/d diet for a 3-5 yr old, shown in Table 6-35, but scaled from the adult (see Harris 2004, Section 2.3.1). |
| $CR_{fish\ organs}$ | Consumption rate: fish organs | kg/kg-day FW | 0.00089 | 0.0027 | Harris 2008 (adults only).Per Harris 2008, organ consumption accounts for 10% of the total fish consumed. Children's exposure factors are based on CSEFH, 1466 kcal/d diet for a 3-5 yr old, shown in Table 6-35, but scaled from the adult (see Harris 2004, Section 2.3.1). |
| $CR_{milk}$ | Consumption rate: milk | kg/kg-day FW | - | - | No milk consumption reported/available. |
| $ET_o$ | Exposure time factor for outdoor exposure to ROPCs in soil | unitless | 0.5 | 0.5 | Harris 2008. |
| $ET_i$ | Exposure time factor for indoor exposure to ROPCs in soil | unitless | 0.5 | 0.5 | Harris 2008. |
| SA | Dermal Surface Area | $m^2$ | 1.8 | 0.76 | Harris 2008, Appendix A, Table 3 (adults only). CSEFH, Table 7-1, mean total body skin surface area of a 3 to <6 yr. old (for child). |
| $V_w$ | Volume of Water used in Sweat Lodge | L | 4 | 4 | Harper & Harris 1997 |
| D | Diameter of Sweat Lodge | m | 2 | 2 | Harper & Harris 1997 |
| $T_{sl}$ | Temperature of Sweat Lodge | °F | 150 | 150 | Harper & Harris 1997 |
| **Nursing Infant Exposure Parameters** | | | | | |
| $ED_{infant}$ | Exposure duration | yr | na | 2 | Harris 2004. Section 2.2.3, page 15 |
| $ED_{maternal}$ | Maternal exposure duration | yr | 25 | na | Harris 2004. Section 2.2.3, page 15 |
| $IR_{milk}$ | Ingestion rate: breast milk | L/day | na | 0.742 | Harper & Harris 1997 |

**Table 7-7 Alternate Resident Subsistence American Indian Scenario #1**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| $BW_{infant}$ | Body weight | kg | na | 9.2 | Based on CSEFH, Table 8-1, mean of 6 to <11 month old infant. |
| $AT_{infant}$ | Averaging time for carcinogens | yrs | na | 2 | Set to exposure duration. |

na = not applicable

CSEFH: EPA. 2008. *Child-Specific Exposure Factors Handbook*, EPA/600/R-06/096F, National Center for Environmental Assessment, Office of Research and Development, Washington DC, September 2008.

EFH: EPA. 1997. *Exposure Factors Handbook, Final*, EPA/600/P-95/002F, US Environmental Protection Agency, National Center for Environmental Assessment, Office of Research and Development, Washington, DC, August 1997.

HHRAP: EPA. 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, EPA/530/R-05/006, US Environmental Protection Agency, Washington, DC. (http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm).

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Harris 2004: Harris SG and Harper BL. 2004. *Exposure Scenario for CTUIR Traditional Subsistence Lifeways*. Department of Science & Engineering, Confederated Tribes of the Umatilla Indian Reservation, P.O. Box 638, Pendleton, Oregon 97801.

Harris 2008: Harris SG. 2008. *Application of the CTUIR Traditional Lifeways Exposure Scenario in Hanford Risk Assessments*, Department of Science & Engineering, Confederated Tribes of the Umatilla Indian Reservation, P.O. Box 638, Pendleton, Oregon 97801.

RIDOLFI Inc. 2007: RIDOLFI Inc. 2007. *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment*, Yakama Nation ERWM Program, September 2007.

**Table 7-8 Alternate Resident Subsistence American Indian Scenario #2**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| EF | Exposure frequency | days/yr | 365 | 365 | RIDOLFI Inc. 2007. |
| ED | Exposure duration | yr | 70 | 6 | RIDOLFI Inc. 2007. |
| ET | Exposure time | hr/day | 24 | 24 | RIDOLFI Inc. 2007. |
| $ET_{sw}$ | Exposure time for sweat lodge | hr/day | 7 | 0.71 | RIDOLFI Inc. 2007.<br>Used recommended value from report which was 7 hour/day for adults, and the average reported (5 hrs/week) for children. |
| BW | Body weight | kg | 70 | 16 | RIDOLFI Inc. 2007. |
| $AT_C$ | Averaging time for carcinogens | yr | 70 | 70 | RIDOLFI Inc. 2007. |
| $AT_{N\ inhal}$ | Inhalation Averaging time for noncarcinogens | yr | 40 | 6 | RIDOLFI Inc. 2007.<br>40 yr duration applies to inhalation and water exposures (including fish consumption), 70 yr duration applies to soil and ingestion (excluding fish) related exposures. |
| $AT_N$ | Averaging time for noncarcinogens | yr | 70 | 6 | |
| IR | Inhalation rate | $m^3$/hr | 1.08 | 0.67 | RIDOLFI Inc. 2007. |
| $CR_{soil}$ | Ingestion rate for soil | kg/day | 0.0002 | 0.0004 | RIDOLFI Inc. 2007. |
| $CR_{dw}$ | Ingestion rate for drinking water | L/day | 4 | 2 | RIDOLFI Inc. 2007. |
| $CR_{ag\ wild}$ | Consumption rate: aboveground wild produce | kg/kg-day FW | 0.0069 | 0.0067 | RIDOLFI Inc. 2007.<br>Table 7 of Ridolfi 2007 provides an adult value of 1118 g/day summed across the categories of wild roots, stalks/leaves, and vegetables plus an additional 299 g/day of fruit for a total of 1417 g/day. The total for children is 314 g/day. Based on Figure 9, the average domestic produce (assume aboveground) consumption constitutes 36% of the produce diet, while average wild aboveground (stalks, leaves, berries) and belowground (roots) produce consumption constitutes 34% and 31% of the produce diet, respectively. The same diet proportions are assumed for adults and children. |
| $CR_{ag}$ | Consumption rate: aboveground domestic produce | kg/kg-day FW | 0.0072 | 0.0070 | |
| $CR_{bg}$ | Consumption rate: belowground produce | kg/kg-day FW | 0.0062 | 0.0060 | |

**Table 7-8 Alternate Resident Subsistence American Indian Scenario #2**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| $CR_{pp}$ | Consumption rate: protected produce | kg/kg-day FW | - | - | Specific protected produce values are not reported; it is assumed aboveground consumption rates include protected produce. |
| $CR_{fowl}$ | Consumption rate wild fowl | kg/kg-day FW | 0.0013 | 0.0017 | RIDOLFI Inc. 2007. Table 7 of Ridolfi 2007 provides a value of 704 g/day for meat consumption (212 g/day for children). Page 20 indicates %60 of meat consumed is domestic. The reference does not indicate the percentage of meat from game/livestock verses wild/domestic fowl. Data from Harris (2008) indicates poultry is 33% of the game & fowl diet for adults. EPA data (EPA/600/R-06/096F, Table 6-35, 1466 kcal/day diet for a 3-5 yr. old) when proportioned between game and fowl according to Harris (2008) and scaled from the adult per Harris (2004), yields a similar value for the proportion of a child's diet that is poultry (~33%). Thus it is assumed that the receptor diet (for game & fowl only) is 33% poultry. |
| $CR_{domestic\ fowl}$ | Consumption rate domestic chicken | kg/kg-day FW | 0.0020 | 0.0026 | |
| $CR_{game}$ | Consumption rate: wild game | kg/kg-day FW | 0.0027 | 0.0036 | RIDOLFI Inc. 2007. Table 7 of Ridolfi 2007 provides a value of 704 g/day for meat consumption (212 g/day for children). Page 20 indicates %60 of meat consumed is domestic. Based on the assumptions used for poultry consumption, it is assumed that 67% of meat consumption is game/livestock (beef). |
| $CR_{game\ organs}$ | Consumption rate: game organs | kg/kg-day FW | - | - | No organ consumption reported/available. |

**Table 7-8 Alternate Resident Subsistence American Indian Scenario #2**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| $CR_{beef}$ | Consumption rate: beef | kg/kg-day FW | 0.0040 | 0.0053 | RIDOLFI Inc. 2007.Table 7 of Ridolfi 2007 provides a value of 704 g/day for meat consumption (212 g/day for children). Page 20 indicates %60 of meat consumed is domestic. Based on the assumptions used for poultry consumption, it is assumed that 67% of meat consumption is game/livestock (beef). |
| $CR_{pork}$ | Consumption rate: pork | kg/kg-day FW | - | - | No distinction made for type of livestock so beef consumption is assumed (no pork consumption assumed). |
| $CR_{eggs}$ | Consumption rate: eggs | kg/kg-day FW | - | - | Included in the fowl consumption rate |
| $CR_{fish}$ | Consumption rate: fish | kg/kg-day FW | 0.0074 | 0.023 | RIDOLFI Inc. 2007. Table 7 of Ridolfi 2007 provides a value of 519 g/day for adult and 363 g/day for child fish consumption. |
| $CR_{fish\ organs}$ | Consumption rate: fish organs | kg/kg-day FW | - | - | No organ consumption reported/available. |
| $CR_{milk}$ | Consumption rate: milk | kg/kg-day FW | 0.017 | 0.031 | RIDOLFI Inc. 2007. Assume the milk is from domestic, comercial sources (Adult: 1.2 L/day / 70 kg = 0.0171 kg/kg-day. Child: 0.5 L/day / 16 kg = 0.0313 kg/kg-day). |
| $ET_o$ | Exposure time factor for outdoor exposure to ROPCs in soil | unitless | 0.29 | 0.29 | RIDOLFI Inc. 2007. Section 3.2.5.1 (assume max of 7 hrs/day) |
| $ET_i$ | Exposure time factor for indoor exposure to ROPCs in soil | unitless | 0.71 | 0.71 | RIDOLFI Inc. 2007. Section 3.2.5.1 states a maximum of 7 hrs/day is out doors, thus the remaining time is assumed to be indoors. |
| SA | Dermal Surface Area | $m^2$ | 1.8 | 0.76 | In the absence of data in RUDOLFI Inc. 2007, use Harris 2008, Appendix A, Table 3 (adults only). CSEFH, Table 7-1, mean total body skin surface area of a 3 to <6 yr. old (for child). |

**Table 7-8 Alternate Resident Subsistence American Indian Scenario #2**

| Parameter | Description | Units | Adult | Child | Source or Reference |
|---|---|---|---|---|---|
| $V_w$ | Volume of Water used in Sweat Lodge | L | 4 | 4 | Harper & Harris 1997 |
| D | Diameter of Sweat Lodge | m | 2 | 2 | Harper & Harris 1997 |
| $T_{sl}$ | Temperature of Sweat Lodge | °F | 150 | 150 | Harper & Harris 1997 |
| **Nursing Infant Exposure Parameters** | | | | | |
| $ED_{infant}$ | Exposure duration | yr | na | 2 | Assume the same as in Harris 2004. Section 2.2.3, page 15. |
| $ED_{maternal}$ | Maternal exposure duration | yr | 25 | na | Assume the same as in Harris 2004. Section 2.2.3, page 15 |
| $IR_{milk}$ | Ingestion rate: breast milk | L/day | na | 0.742 | Harper & Harris 1997 |
| $BW_{infant}$ | Body weight | kg | na | 9.2 | Based on CSEFH, Table 8-1, mean of 6 to <11 month old infant. |
| $AT_{infant}$ | Averaging time for carcinogens | yr | na | 2 | Set to exposure duration. |

na = not applicable

CSEFH: EPA. 2008. *Child-Specific Exposure Factors Handbook*, EPA/600/R-06/096F, National Center for Environmental Assessment, Office of Research and Development, Washington DC, September 2008.

Harper & Harris 1997: "A Native American Exposure Scenario," Risk Anal., Volume 17, Issue 6, p 789-795.

RIDOLFI Inc. 2007: RIDOLFI Inc. 2007. *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment*, Yakama Nation ERWM Program, September 2007.

**Table 7-9 Modeling Parameters for Estimating Exposure Point Concentrations in Biota for Human Consumption**

| Parameter | Description | Units | Value | Reference |
|---|---|---|---|---|
| **General/Global Biota Parameters** | | | | |
| $F_{plant}$ | Fraction of plant grown on contaminated soil and ingested by the specific animal being modeled - applies to all plant types (produce, forage, silage, grain are possibilities) eaten by the animal being modeled | unitless | 1 | [1] |
| *Bs* | Soil bioavailability factor | unitless | 1 | [1] |
| *MF* | Metabolism factor for Bis(2-ethylhexyl)phthalate | unitless | 0.01 | [1] |
| | Metabolism factor for all other constituents | unitless | 1 | [1] |
| **Beef Parameters** | | | | |
| $Qp_{forage}$ | Quantity of forage eaten by beef cattle per day | kg DW plant/day | 8.8 | [1] |
| $Qp_{silage}$ | Quantity of silage eaten by beef cattle per day | kg DW plant/day | 2.5 | [1] |
| $Qp_{grain}$ | Quantity of grain eaten by beef cattle per day | kg DW plant/day | 0.47 | [1] |
| *Qs* | Quantity of soil ingested by beef cattle per day | kg/day | 0.5 | [1] |
| *Ba* | Biotransfer factor for beef | day/kg FW tissue | constituent specific | |
| **Dairy Parameters** | | | | |
| $Qp_{forage}$ | Quantity of forage eaten by dairy cattle per day | kg DW plant/day | 13.2 | [1] |
| $Qp_{silage}$ | Quantity of silage eaten by dairy cattle per day | kg DW plant/day | 4.1 | [1] |
| $Qp_{grain}$ | Quantity of grain eaten by dairy cattle per day | kg DW plant/day | 3 | [1] |
| *Qs* | Quantity of soil eaten by dairy cattle per day | kg/day | 0.4 | [1] |
| *Ba* | Biotransfer factor for dairy cattle | day/kg FW tissue | constituent specific | |
| **Pork Parameters** | | | | |
| $Qp_{forage}$ | Quantity of forage eaten by swine per day | kg DW plant/day | 0 | [1] |
| $Qp_{silage}$ | Quantity of silage eaten by swine per day | kg DW plant/day | 1.4 | [1] |
| $Qp_{grain}$ | Quantity of grain eaten by swine per day | kg DW plant/day | 3.3 | [1] |
| *Qs* | Quantity of soil eaten by swine per day | kg/day | 0.37 | [1] |
| *Ba* | Biotransfer factor for swine | day/kg FW tissue | constituent specific | |
| **Chicken (domestic and wild) Parameters** | | | | |
| $Qp_{forage}$ | Quantity of forage eaten by chicken per day | kg DW plant/day | 0 | [1] |
| $Qp_{silage}$ | Quantity of silage eaten by chicken per day | kg DW plant/day | 0 | [1] |
| $Qp_{grain}$ | Quantity of grain eaten by chicken per day | kg DW plant/day | 0.2 | [1] |
| *Qs* | Quantity of soil eaten by chicken per day | kg/day | 0.022 | [1] |
| *Ba* | Biotransfer factor for chicken | day/kg FW tissue | constituent specific | |
| **Game Parameters** | | | | |
| $Qp_{forage}$ | Quantity of forage eaten by game per day | kg DW plant/day | 1.463 | [2] |
| $Qp_{silage}$ | Quantity of silage eaten by game per day | kg DW plant/day | 0 | [3] |

**Table 7-9 Modeling Parameters for Estimating Exposure Point Concentrations in Biota for Human Consumption**

| Parameter | Description | Units | Value | Reference |
|---|---|---|---|---|
| $Qp_{grain}$ | Quantity of grain eaten by game per day | kg DW plant/day | 0 | [3] |
| $Qs$ | Quantity of soil eaten by game per day | kg/day | 0 | [3] |
| $Ba$ | Biotransfer factor for game | day/kg FW tissue | constituent specific | |
| **Fish Parameters** | | | | |
| $f_{lipid}$ | Fish lipid content | unitless | 0.07 | [1] |
| $OC_{sed}$ | Fraction of organic carbon in bottom sediment | unitless | 0.04 | [1] |

[1] EPA. 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities*, Final, EPA/530/R-05/006. September 2005. US Environmental Protection Agency, Washington DC.

[2] Higley, K. A., and R. Kuperman, 1996. "Ecotoxicological benchmarks for radionuclide contaminants at RFETS, Appendix C," EAD Argonne National Laboratory Report RF/ER-96-0039. Assumes average for mule deer.

[3] No data available - assumed value

**Table 7-10 Toxicity Surrogates**

| Constituent | Surrogate |
|---|---|
| Petroleum hydrocarbons | The Washington State *Model Toxics Control Act* (Ecology 2001) method will be used to calculate surrogate toxicity values for the inhalation pathway for hydrocarbons lacking chemical-specific values. |
| methyl isocyanate (CAS #624-83-9) | acrylonitrile (CAS #107-13-1) |
| and cis-1,3-dichloropropene (CAS #10061-01-5) | cis-1,2-dichloroethene (CAS #156-59-2) |
| trichlorofluoroethane (CAS #27154-33-2) | trichlorofluoromethane (CAS #75-69-4) |
| 5-nitroacenaphthene (CAS #602-87-9) | Acenaphthene (CAS #83-32-9) |
| sec-butyl benzene (CAS #135-98-8) | tert-butyl benzene (CAS #98-06-6) |
| dichloropentadiene (CAS #61626-71-9) | chlorocyclopentadiene (CAS #41851-50-7) |

**Figure 7-1 Exposure Assessment Grids**

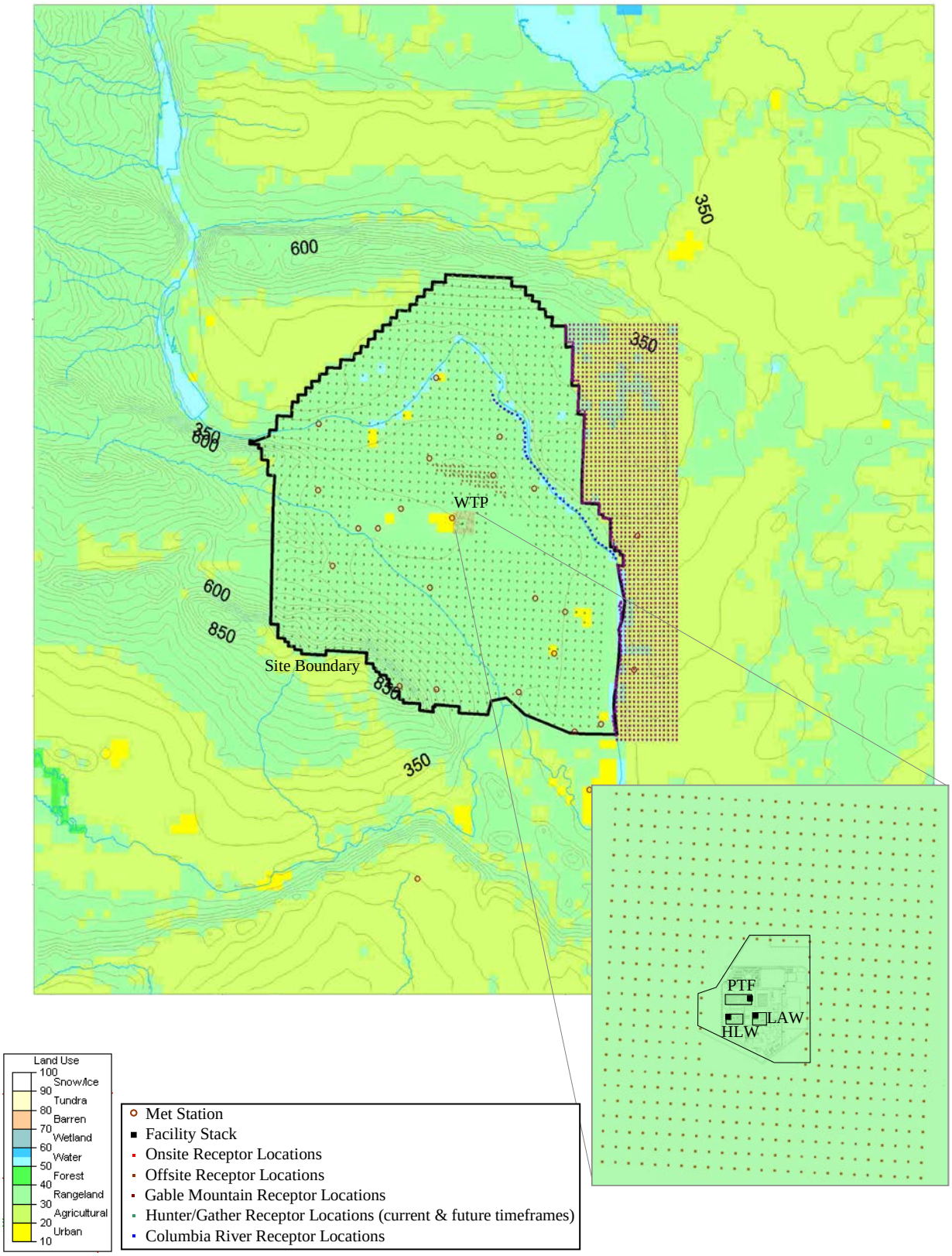


**Figure 7-2 Resident Subsistence American Indian Hunting and Gathering Areas**

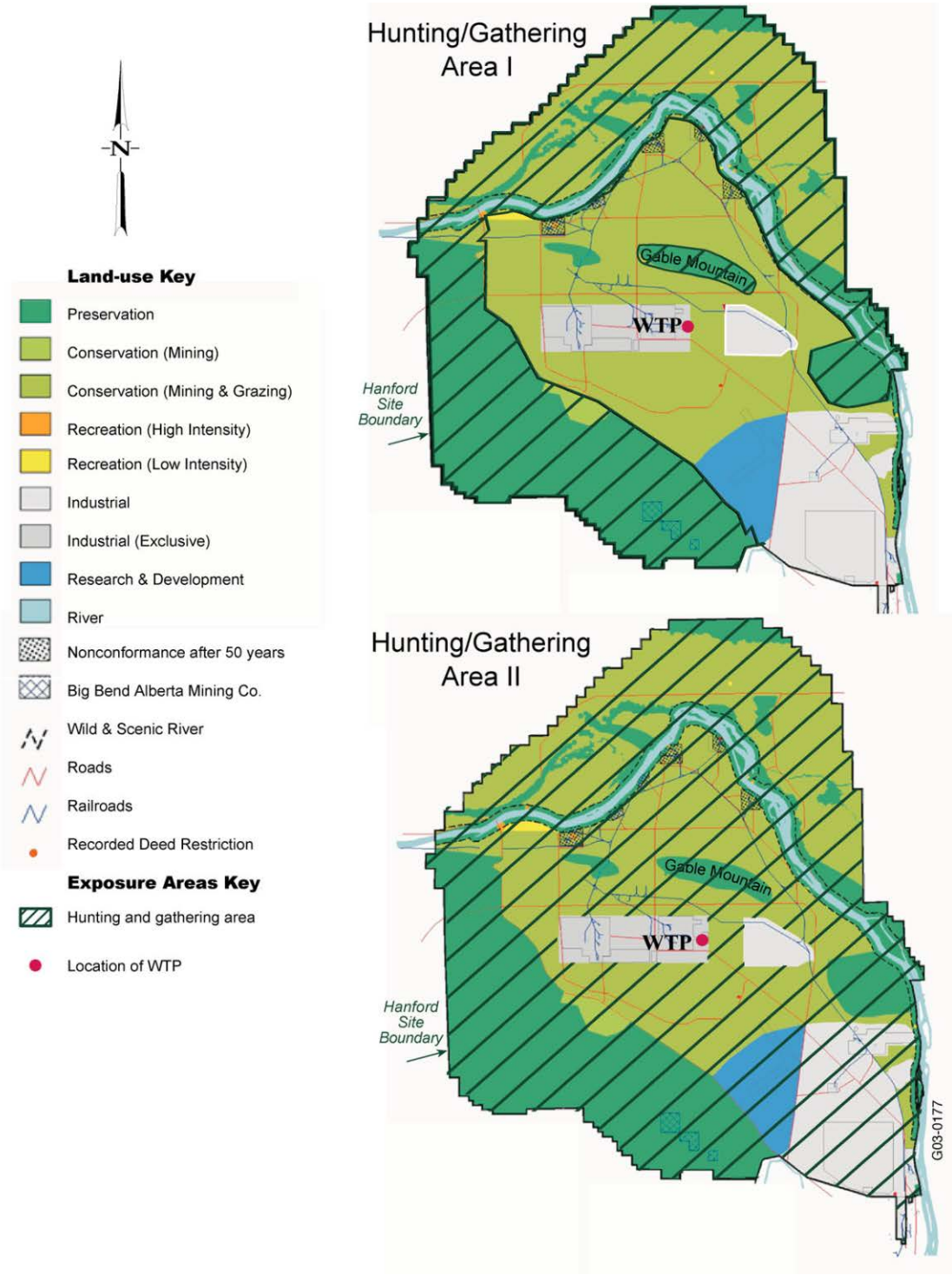


**Figure 7-3 Locations of Potential Human Receptors including Potentially Sensitive Receptors**

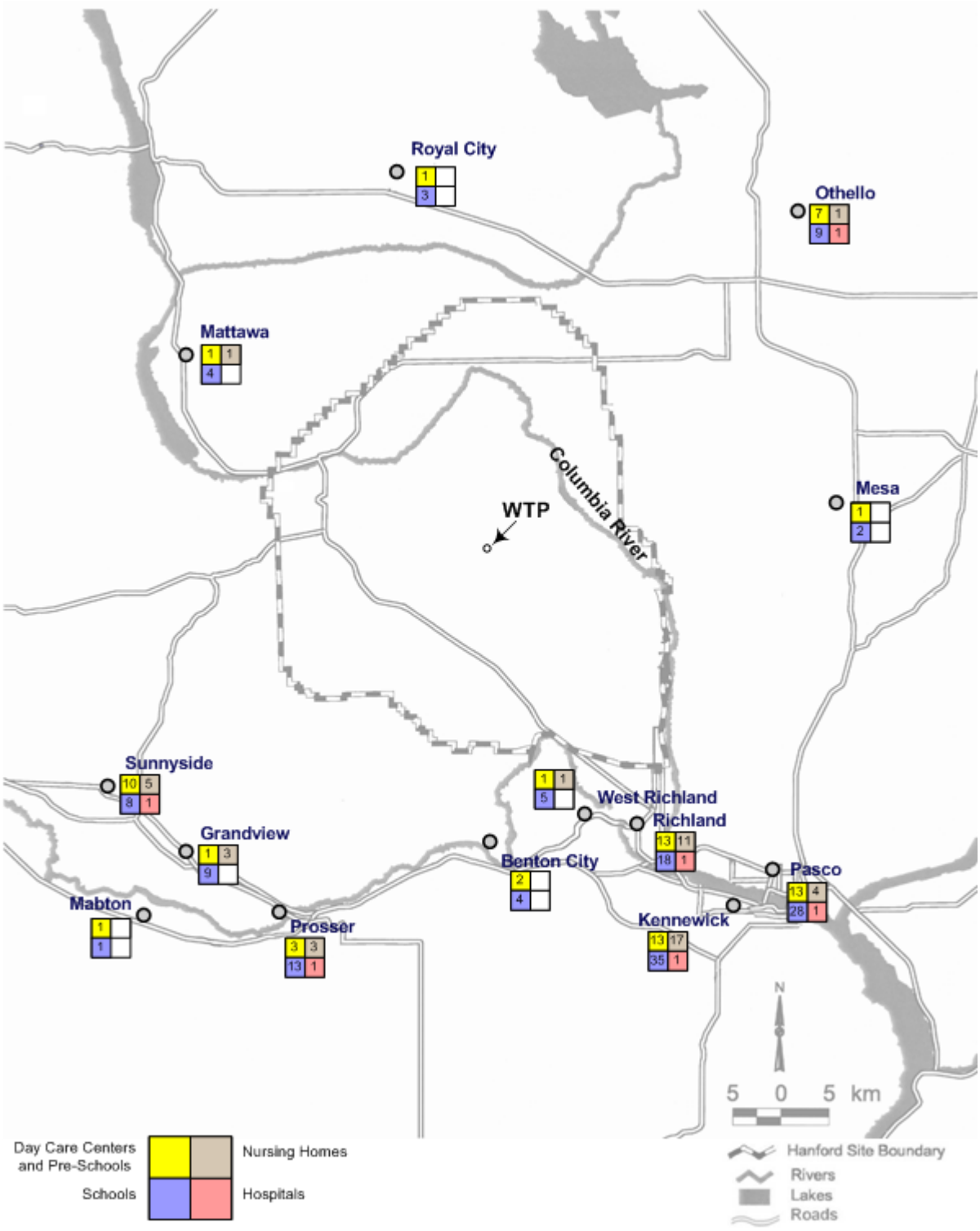
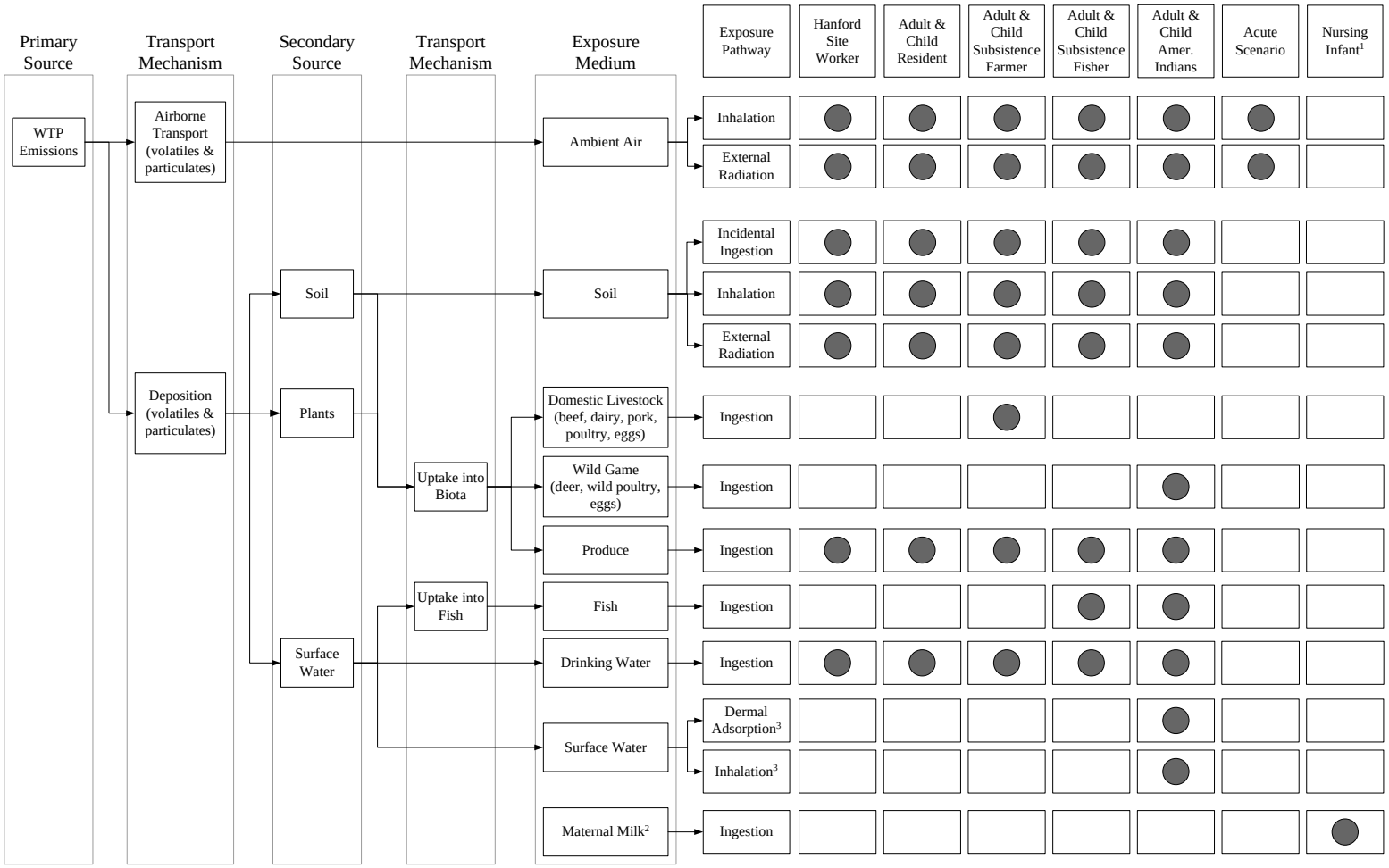


**Figure 7-4 Human Health Conceptual Exposure Model**

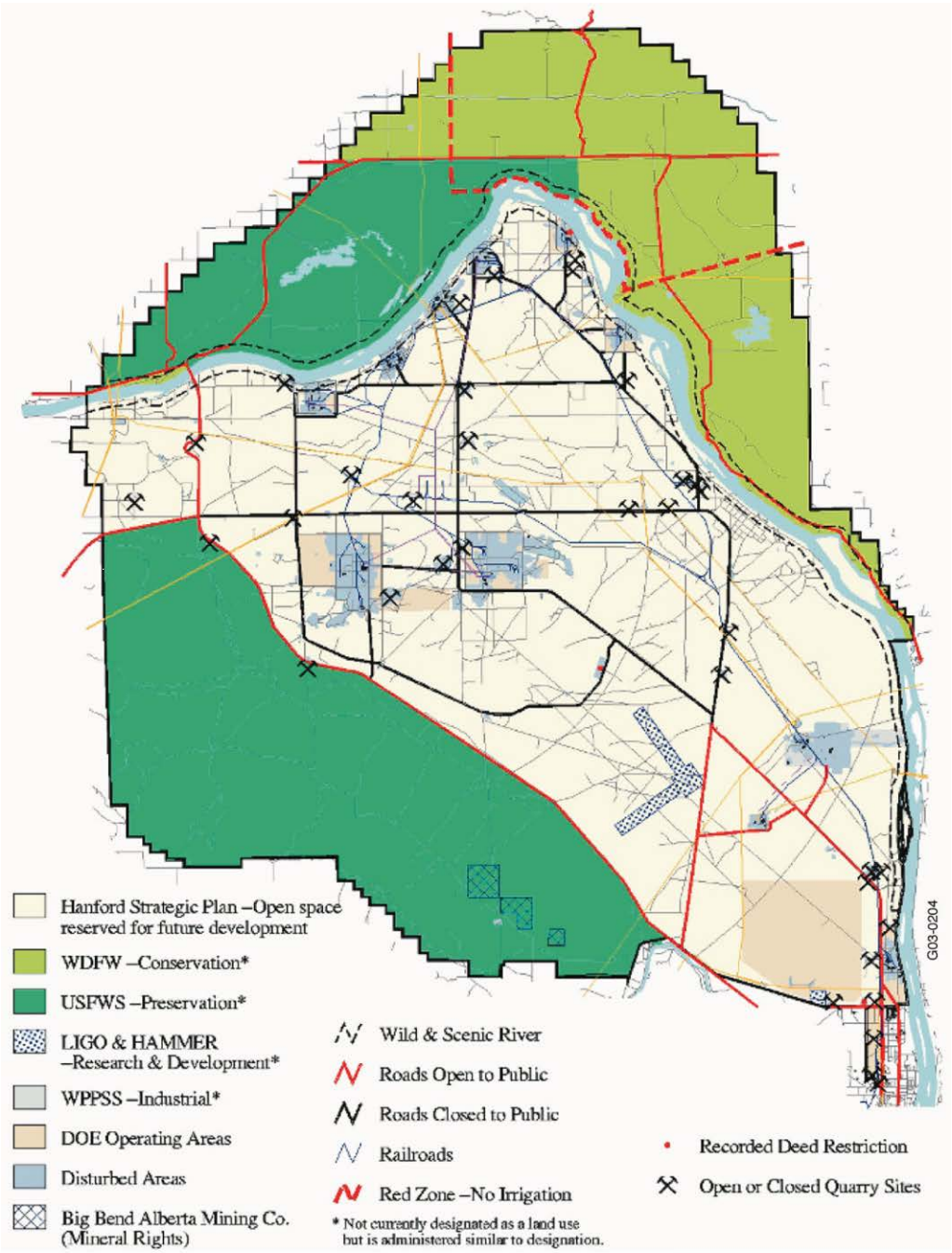



Notes:
[1] Nursing infant is evaluated for the worker, resident, farmer, fisher, and tribal exposure scenarios
[2] Maternal milk pathway assumes maternal exposure to all media for the applicable adult receptor exposure scenario
[3] Sweat lodge exposure is a pathway unique to tribal exposures and includes adsorption and inhalation of water vapor/aerosols

Pathway not complete

Pathway complete and evaluated in screening human health risk assessment

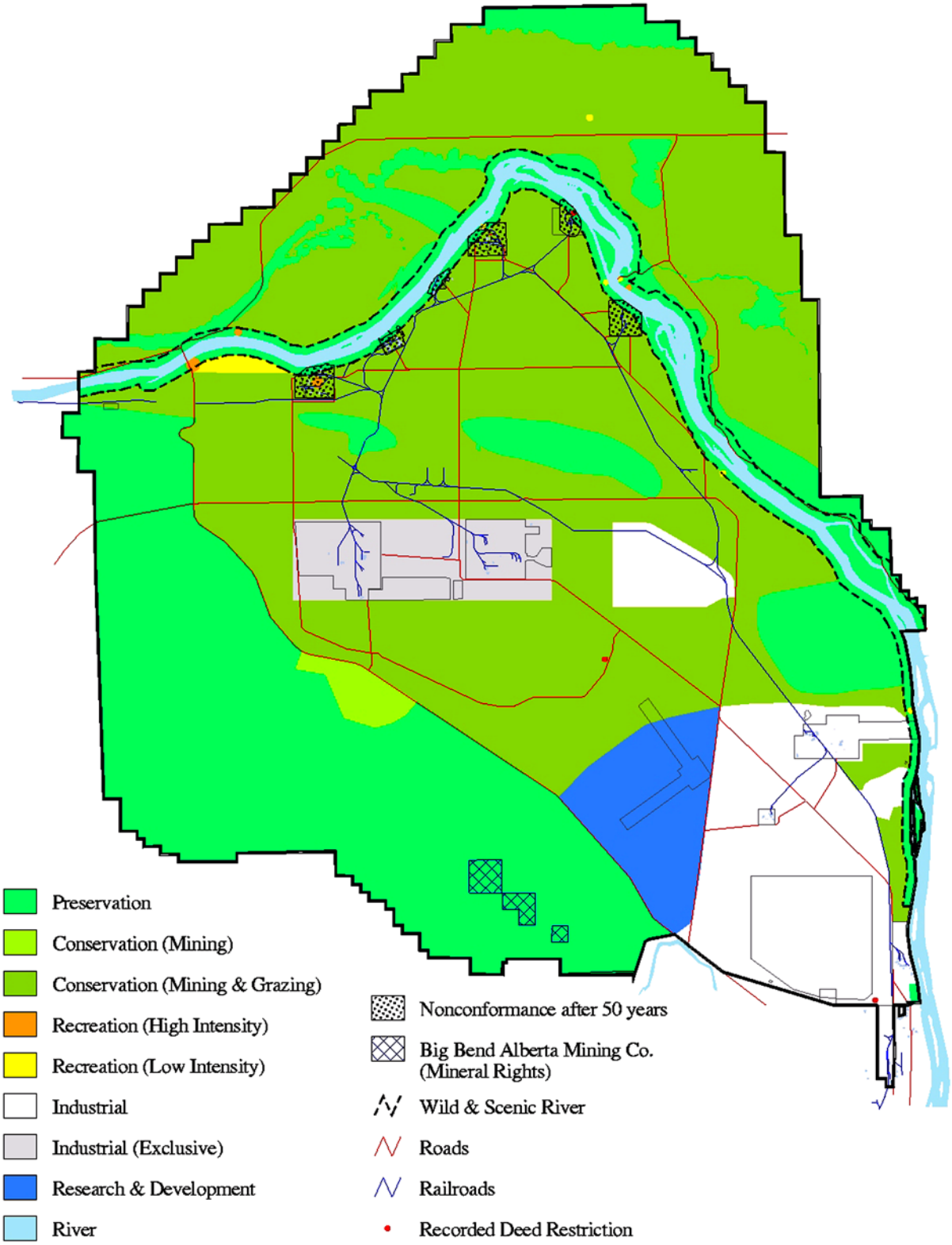
**Figure 7-5 Hanford Site Existing Land Use Map - 1996**




BHI:rpp 04/23/98 clup/noactionalt.aml Database: 07-DEC-1998

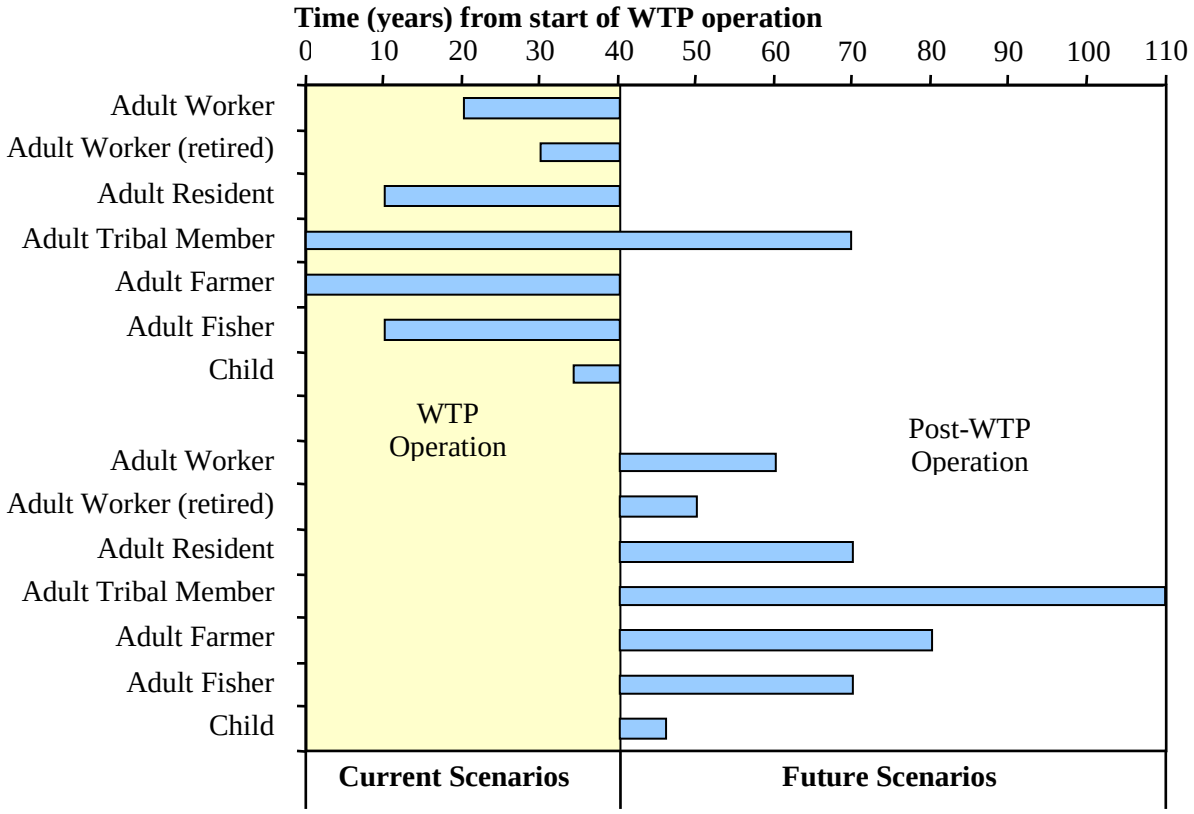
| | |
|---|---|
| WDFW | Washington Department of Fish and Wildlife |
| USFWS | U.S. Fish and Wildlife Service |
| LIGO | Laser Interferometer Gravitational Wave Observatory |
| HAMMER | Hazardous Materials Management and Emergency Response |
| WPPSS | Washington Public Power Supply System |

**Figure 7-6 Hanford Site Projected Land Use Map - 2046**




BHI:rpp 04/23/98 clup/prefalt.aml Database: 07-DEC-1998

**Figure 7-7 Receptor Exposure Timeline**



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# 8 Screening-Level Ecological Risk Assessment

The screening-level ecological risk assessment (SLERA) incorporates four fundamental components of the ERA process: (1) problem formulation, (2) exposure assessment, (3) effects assessment, and (4) risk characterization. Selection of COPCs and ROPCs (discussed in Section 4 of this work plan), quantification of emissions (discussed in Section 5), and dispersion modeling (discussed in Section 6) feed critical information to this process. The SLERA is intended to meet three goals identified in EPA draft guidance (SLERAP, EPA 1999): the SLERA (1) provides the maximum, most conservative exposure estimate, (2) "identifies which pathways are driving risk specific to a COPC and receptor," and (3) "allows risk management efforts to be prioritized." These methods will be used for both the PRA and the FRA, which will differ in that the PRA will use soil and surface water concentrations modeled from estimated stack emissions, whereas the FRA will use soil and surface water concentrations that are based on the results of a performance demonstration test using surrogate waste as well as estimated stack emissions. The WTP recognizes that there are significant limitations to using a limited performance demonstration test to predict the ability of the melter offgas systems to control emissions. However, proven thermal treatment approaches will be used to select test constituents that are representative of the worst-case constituents and operating conditions so that a conservative estimate of performance is obtained.

## 8.1 Problem Formulation

This section of the RAWP focuses on the conceptual exposure model (Section 8.1.1), ecological setting (Section 8.1.2), ecological receptor identification (Section 8.1.3), and assessment/measurement endpoints (Section 8.1.4). Each is defined below.

### 8.1.1 Ecological Conceptual Exposure Model

A conceptual exposure model has been developed that identifies ecological receptors and complete exposure pathways (i.e., exposure scenarios). The conceptual exposure model is shown as Figure 8-1. The end product of the conceptual exposure model is the identification of exposure scenarios that are defined by exposure pathways and potentially exposed populations. The conceptual model was developed from information obtained from EPA (1999) and *Screening Assessment and Requirements for a Comprehensive Assessment: Columbia River Comprehensive Impact Assessment* (DOE-RL 1998).

The conceptual model focuses on identifying complete exposure pathways for potentially exposed receptor populations. An exposure pathway is the means through which an organism comes in contact with a chemical or radionuclide in the environment. Exposure pathways are determined by environmental conditions (such as location of habitat and home ranges as well as wind speed/direction), the potential for chemical migration among media (such as air, soil, or surface water), and the behavior and diet of potentially-exposed plant and animal populations. Although several potential pathways may exist, not all pathways may be complete. For a pathway to be complete, all of the following four factors must exist:

1. a source of COPC or ROPC release into the Hanford Site environment
2. a release and transport mechanism (such as deposition to soil) that moves the COPC or ROPC from the source, such as a stack, to other locations in the environment

3. a point of contact with a contaminated medium
4. an exposure route to the receptor, such as ingesting or inhaling affected media

These four factors were considered in the conceptual model. The sources of COPC and ROPC release are the stack and process cell emissions from the WTP (Section 3). Air dispersion (Section 6.1), soil and surface water accumulation (Sections 6.2 and 6.3, respectively), potential points of contact, and complete exposure pathways are identified to formulate exposure scenarios that will be the focus of the quantitative risk assessment.

### 8.1.2 Ecological Characterization

The ecological setting and habitats at Hanford and offsite locations determine what receptors will be potentially exposed and the important complete pathways. For example, deserts and water bodies have different receptors and exposure pathways. The habitats, food webs, and receptors are the same for both the Hanford Site and offsite locations. The Hanford Site and offsite locations for approximately 100 km in any direction are located in the shrub-steppe region of the Columbia Basin (Daubenmire 1970). The shrub-steppe vegetation zone historically included a shrub overstory and an understory of grasses. The typical plant and animal communities at the Hanford Site and adjacent offsite areas are qualitatively similar. Populations of disturbance-intolerant native species are likely smaller and populations of invasive and native species more tolerant of disturbance are likely larger in offsite areas disturbed by agriculture, grazing, and urbanization. Ecological resources at the Hanford Site are extensive, diverse, and important, as explained by Neitzel et al. (2005). The Hanford Site, unlike adjacent areas, has not been farmed or grazed for over 50 years. It has become a refuge for a variety of plant and animal species (Gray and Rickard 1989), containing one of the largest remaining undisturbed shrub-steppe ecosystems in Washington State (see Appendix C for a listing of plants and animals observed on the site). About 665 $km^2$ (257 $mi^2$) of undeveloped lands located on site (almost half of the total area of the Hanford Site) have been designated as ecological study areas or refuges (Figure 8-2).

#### 8.1.2.1 Physiographic Setting

The Hanford Site and adjacent region lie within the Intermountain Semidesert Province (USFS 1994). This province includes the plains and plateaus of the Columbia-Snake River Plateau and the Wyoming Basin. The climate is cool, the average temperature being about 50 °F, and semi-arid, with the average annual precipitation ranging from approximately 6 inches to 20 inches across the province from west to east. At the Hanford Site, the average annual precipitation totals about 6 inches. This precipitation is evenly distributed throughout the fall, winter, and spring months, with little precipitation during the summer months.

The Hanford Site and adjacent region lie within the semi-arid Pasco Basin of the Columbia Plateau in southeastern Washington State. The Hanford Site occupies an area of approximately 1450 $km^2$ (560 $mi^2$) north of the confluence of the Yakima River with the Columbia River. The Pasco Basin lies within the southwest corner of the larger Columbia Basin. The Hanford Site occupies approximately one-third of the land area within the Pasco Basin. The Columbia River flows through the northern part of the Hanford Site and forms part of the Hanford Site's eastern boundary after turning south. The Yakima River runs near the southern boundary. Rattlesnake Mountain, Yakima Ridge, and Umtanum Ridge form the southwestern and western boundaries of the Hanford Site. The Saddle Mountains form the northern boundary. Adjoining lands to the west, north, and east are principally range and agricultural land. The

Hanford Site exhibits low relief, ranging from 120 m above mean sea level (MSL) at the Columbia River to 230 m MSL in the vicinity of the WTP sites.

The 200 Area and WTP site are located on the Central Plateau. The Central Plateau is characterized by generally low-relief hills with deeply incised river drainages. Gable Butte and Gable Mountain (small east to west ridges), located north of the Central Plateau, are characterized by folded layers of rock that are the high points along the Umtanum anticlinal ridge (Neitzel et al. 2005).

#### 8.1.2.2 Regional Ecology

The region comprising the Hanford Site and offsite locations has been characterized as shrub-steppe. The National Biological Service has identified native shrub and grassland steppes in Washington and Oregon as endangered ecosystems (DOE 1999).

Biodiversity in the region is enhanced by the large, relatively undisturbed tract of native shrub-steppe habitat on the Hanford Site and by the Hanford Reach, a stretch of the Columbia River below the Priest Rapids Dam (DOE 1999). Additional factors influencing biodiversity include topographic features such as Rattlesnake Mountain, Gable Butte, Gable Mountain, and the presence of a variety of soils ranging from sand to silty and sandy loam. Unique terrestrial habitats include basalt outcrops, scarps (cliffs), scree slopes, and sand dunes. Offsite areas likely have similar unique habitats. Aquatic habitats are mostly associated with the Columbia River and include open water habitat, wetlands, and riparian areas (Figure 8-4).

Cold Creek and a tributary, Dry Creek, are ephemeral streams within the Yakima River drainage system that roughly parallel State Route 240 through the Hanford Site. Both streams drain areas to the west of Hanford Site. Surface flow, when it occurs, infiltrates and disappears into the surface sediments in the western portion of the Hanford Site. Rattlesnake Springs, located on the western portion of the Hanford Site, forms a small surface stream that flows for about 3 km (1.8 mi) before disappearing into the ground (Figure 8-3).

West Lake is a small saline pond located north of the 200 East Area (Figure 8-2) and is recharged from groundwater (Neitzel et al. 2005). West Lake has not received direct effluent discharges from any Hanford Site facilities. This water body is created by an elevated water table within a low surface area south of Gable Mountain. This artificially elevated water table occurs under much of the Hanford Site, reflecting the augmented recharge from Hanford Site operations. The water level and size of the lake has been decreasing over the past several years because of reduced wastewater discharge (Neitzel et al. 2005).

Gable Mountain Pond (also to the north of the 200 East Area but south of West Lake) and the B Pond System (immediately east of the 200 East Area) received cooling water discharges from several facilities at the Hanford Site (Rogers and Rickard 1977). These artificial water bodies, formed by the wastewater discharges from the operation of the separation facilities, were decommissioned and covered with soil.

The 200 Area Treated Effluent Disposal Facility (TEDF) disposal ponds (east of the 200 East Area) consists of two disposal ponds that receive industrial wastewater permitted in accordance with Ecology's State Waste Discharge Permit Program (WAC 173-216). The wastewater evaporates into the air or percolates into the ground from the disposal ponds (Neitzel et al. 2005).

There are several naturally occurring vernal ponds near Gable Mountain and Gable Butte. These ponds appear to occur where a depression is present in a relatively shallow buried basalt surface. Water collects

within the depression over the winter, resulting in a shallow pond that dries during the summer months (Neitzel et al. 2005).

**Vegetation**

The Columbia Basin (Daubenmire 1970) is a shrub-steppe ecosystem characterized by bunchgrasses and sagebrushes (Figure 8-4). This ecosystem is also referred to as high desert, northern desert shrub, or desert scrub (Franklin and Dyrness 1973). Prior to settlement by western Europeans, the dominant plant in the area was big sagebrush (*Artemisia tridentata*) with an understory of perennial bunchgrasses, especially Sandberg's bluegrass (*Poa sandbergii*) and bluebunch wheatgrass (*Agropyron spicatum*). Following settlement in the early 1800s, grazing and agriculture disrupted the native vegetation and opened the way for invasive species such as Russian thistle (*Salsola kali*) and cheatgrass (*Bromus tectorum*). Cheatgrass is now dominant in fields that were cultivated prior to the establishment of the Hanford Site. Cheatgrass also is well established on rangelands at elevations less than 244 m (800 ft) (DOE 1999). Establishment of the Hanford Site as a nuclear complex in 1943 resulted in the creation of a secured area of mostly undeveloped land with scattered, small industrial facilities. Consequently, the Hanford Site is one of a small number of remaining shrub-steppe tracts in Washington State that is relatively undisturbed. Wildfire is a common occurrence and can significantly alter the shrub component of the vegetation. The most recent extensive fire on the Hanford Site was in 2000 and burned over 660 $km^2$ (250 $mi^2$).

Trees were planted and irrigated on most of the pre-1943 farms to provide windbreaks and shade. Some of these trees have persisted and serve as nesting platforms for several species of birds (hawks, owls, ravens, magpies, and great blue herons) and as night roosts for wintering bald eagles (DOE 1999) (Figure 8-5).

A total of 727 species representing 90 families of vascular plants are recorded for the Hanford Site (Neitzel et al. 2005). The dominant plants are big sagebrush, rabbitbrush, cheatgrass, and Sandberg's bluegrass, with cheatgrass providing half of the total plant cover on much of the Hanford Site. Cheatgrass and Russian thistle are annuals introduced to the United States from Eurasia in the late 1800s that invade disturbed areas. Big sagebrush and bitterbrush (*Purshia* spp.) are widely spaced and usually provide less than 20 % canopy cover. Bitterbrush provides important browse for the resident mule deer herd. The dominant understory plants are grasses, especially cheatgrass, Sandberg's bluegrass, Indian ricegrass (*Oryzopsis hymenoides*), June grass (*Koeleria macrantha*), and needle-and-thread grass (*Stibacomata*).

**Central Plateau.** The Central Plateau and surrounding areas in the Columbia Basin have been identified as predominantly shrub-steppe (Duranceau 1995). This designation includes communities dominated by big sagebrush and bitterbrush (*Purshia tridentata*) with an understory of cheatgrass or Sandberg's bluegrass. Past wildfires in the Central Plateau have opened up some areas, creating a mosaic of shrub- and grass-dominated areas. More than 100 species of plants have been identified on the Central Plateau (Cushing 1992). Big sagebrush, bitterbrush, cheatgrass, and Sandberg's bluegrass are common species within the 200 Area (Neitzel et al. 2005). Cheatgrass provides approximately 50 % of the total plant cover. Cheatgrass also is common where native plant communities have been disturbed by wildfire or past construction activities. Three vegetation subtypes occurring in the vicinity of the 200 West Area of the Central Plateau are sagebrush and Sandberg's bluegrass, sagebrush and needle-and-thread grass, and spiny hopsage and Sandberg's bluegrass.

The WTP site in and immediately surrounding the 200 East Area is approximately 40 % big sagebrush and rabbitbrush (Figure 8-6). Another 20 % is dominated by Russian thistle, with the remainder being disturbed vegetation or bare gravel (PNL 1994). Other vegetation in the 200 Area includes introduced

perennial grasses planted to revegetate and stabilize disturbed areas, such as waste burial grounds. Introduced perennial grasses (e.g., Siberian wheatgrass [*Agropyron sibericum*]) have been used extensively in the Central Plateau to revegetate and stabilize waste burial grounds against wind and water erosion (DOE 1999). Siberian wheatgrass has proven to be drought tolerant and better adapted to sandy soils than other cultivars used in Central Plateau revegetation efforts (Stegen 1993; WHC 1993).

**Columbia River.** The two major vegetation types occurring along the Hanford Reach of the Columbia River are riparian and upland (NPS 1994). Riparian habitats are found along the shoreline, slack water and slough areas, and on islands in the river. Riparian vegetation at these locations includes both woody and herbaceous species maintained by the high water table immediately adjacent to the river. Common plant species occurring in the riparian zone include water smartweed, sedges, reed canary grass, bulbous bluegrass, common witchgrass, large barnyard, willow, mulberry, and Siberian elm (Neitzel et al. 2005). Sensitive habitats within the riparian zone include islands and cobbled shorelines occurring as a narrow band along the Hanford Reach. Plant species occurring in these areas include perennial, summer-blooming forbs adapted to seasonal changes in water levels (NPS 1994). Upland habitats along the Hanford Reach are composed of shrub-steppe vegetation similar to that found on the Central Plateau (DOE 1999). Sand dunes are often colonized by needle-and-thread grass on the north-facing slopes and a mixture of shrubs and forbs at the crest (Sackschewsky et al. 1992).

In summary, special topographic features on the Hanford Site include Gable Butte and Gable Mountain north of the Central Plateau and an extensive series of active sand dunes in the southeast portion of the area. The dominant plant communities are cheatgrass, sagebrush-bitterbrush and Sandberg's bluegrass, sagebrush and cheatgrass, Sandberg's bluegrass, and riparian plant communities (Sackschewsky et al. 1992). Depending on the location, many of the terrestrial plants occurring in this area are the same as those found in the adjacent Columbia River and Columbia Basin. Big sagebrush, bitterbrush, rabbitbrush, cheatgrass, and Sandberg's bluegrass are common species in the area just north of the 300 Area in the southeast corner of the Hanford Site (Neitzel et al. 2005). Common plants growing in riparian areas along the Columbia River include reed canarygrass, common witchgrass, large barnyard grass, summer-blooming forbs, sandbar willow, poplar, white mulberry, and Russian olive (NPS 1994). Vegetation occurring on scree slopes, outcrops, and scarps such as those on Gable Butte and Gable Mountain is limited to scattered individuals and groups of plants. Plant species include squaw currant, bluebunch wheatgrass, rock buckwheat, and thyme buckwheat. Rigid sagebrush (*Artemesia rigida*) occurs at the Hanford Site only on Gable Mountain and Umtanum Ridge (Downs et al. 1993).

**Wildlife**

Approximately 300 species of terrestrial vertebrates have been observed at the Hanford Site. This number includes 46 species of mammals, 246 species of birds, 6 species of amphibians, and 7 species of reptiles (Neitzel et al. 2005).

**Mammals**. Large herbivorous mammalian species that are found on the Hanford Site and offsite area include mule deer and Rocky Mountain elk. Mule deer (*Odocoileus hemionus*) are most often found near the Columbia River and use Columbia River islands for fawning and nursery areas. Rocky Mountain elk (*Cervus elaphus*) began to appear on the Hanford Site during the early 1970s and are generally restricted to the FEALE Reserve. Elk frequently move off the reserve to private lands to the north and west, particularly during late spring, summer, and early fall (Neitzel et al. 2005).

Black-tailed jackrabbits (*Lepus californicus*) are common on the Hanford Site and offsite area and are most often found in mature stands of sagebrush. Cottontail rabbits (*Sylvilagus* spp.) also are common but are more closely associated with developed areas. Townsend's ground squirrels (*Spermophilus*

*townsendii mollis*) occur in colonies of various sizes scattered across the Hanford Site and offsite areas. The most abundant mammal inhabiting the site is the Great Basin pocket mouse (*Perognathus parvus*). This mouse occurs all across the Columbia Basin and on the slopes of the surrounding ridges. Other small mammals include the western harvest mouse (*Reithrodontomys megalotis*), grasshopper mouse (*Onychomys leucogaster*), deer mouse (*Peromyscus maniculatus*), house mouse (*Mus musculus*), mountain vole (*Microtus montanus*), sagebrush vole (*Lagurus curtatus*), brushy-tailed woodrat (*Neotoma cinerea*), northern pocket gopher (*Thomomys talpoides*), vagrant shrew (*Sorex vagrans*), and Merriam's shrew (*Sorex merriami*) (DOE 1999). In addition to mule deer, mammals occurring primarily in riparian areas include rodents (muskrat [*Ondatra zibethica*]); furbearers (mink [*Mustela vison*], River otter [*Lutra canadensis*], weasel [*Mustela spp.*]); porcupine (*Erithizon dorsatum*); raccoon (*Procyon lotor*); and skunk (*Mephitis mephitis*) (Neitzel et al. 2005).

Common mammalian predators are the coyote (*Canis latrans*), bobcat (*Lynx rufus*), and badger (*Taxidea taxus*). These carnivores feed primarily on the several species of small mammals, including the Great Basin pocket mouse, western harvest mouse, grasshopper mouse, deer mouse, house mouse, Townsend's ground squirrel, mountain vole, sagebrush vole, black-tailed jackrabbit, brushy-tailed woodrat, and northern pocket gopher. Coyotes have been a major predator of Canada goose (*Branta canadensis leucopareia*) nests on Columbia River islands, especially upstream from the abandoned Hanford townsite (DOE 1999). There was a reported sighting of a cougar (*Felis concolor*) on ALE Reserve by experienced biologists during the elk relocation effort in March 2000 (Neitzel et al. 2005).

Up to 14 species of bats are known to be or have the potential to be present on or in the vicinity of the Hanford Site. They include the pallid bat (*Antrozous pallidus*), hoary bat (*Lasiurus cinereus*), silver-haired bat (*Lasionycteris noctivagan*), little brown bat (*Myotis lucifugus*), California brown bat (*Myotis californicus*), Yuma brown bat (*Myotis yamanensis*), and Pacific western big-eared bat (*Plecotus townsendii*) (Fitzner and Gray 1991). The pallid bat, which roosts in abandoned buildings, is considered to be the most abundant. All of these bat species feed on flying insects.

**Birds.** Nearly 250 species of birds occur on or near the Hanford Site as year-round residents, seasonal residents, migrants, and accidentals. There are 144 bird species considered common to the Hanford Site (Neitzel et al. 2005).

Eleven raptors have been documented as nesting on the Hanford Site. These include the northern harrier (*Circus cyaneus*), red-tailed hawk (*Buteo jamaicensis*), Swainson's hawk (*Buteo swainsoni*), golden eagle (*Aquila chrysaetos*), prairie falcon (*Falco mexicanus*), American kestrel (*Falco sparverius*), barn owl (*Tyto alba*), great horned owl (*Bubo virginianus*), long-eared owl (*Asio otus*), short-eared owl (*Asio flammeus*), and burrowing owl (*Athene cunicularia*) (Fitzner and Gray 1991, Rickard et al. 1988). Raptors use a variety of habitats for nesting and foraging on the Hanford Site. Nesting habitats include outcrops, cliffs, trees, marshes, fields, and utility towers. Depending on raptor species, prey may include small mammals, birds, reptiles (i.e., snakes), and insects.

Great blue herons (*Ardea herodias*) and black-crowned night herons (*Nycticorax nycticorax*) are associated with trees in riparian habitat along the Columbia River and use groves or individual trees for perching and nesting. On occasion, great blue herons have constructed nests in the large metal powerline towers that are present on the shores of the Columbia River (Neitzel et al., 2005).

Several songbird species occur in the shrub-steppe vegetation throughout the region. These include the western meadowlark (*Sturnella neglecta*), grasshopper sparrow (*Ammodramus savannarum*), horned lark (*Eremophila alpestris*), and sage thrasher (*Oreoscoptes montanus*) (Downs et al. 1993). The western

meadowlark and horned lark are the most abundant breeding bird species within the shrub-steppe habitat (Rickard and Poole 1989). These two species nest on the ground in the open, while other species (such as sage sparrow, sage thrasher, and loggerhead shrike) require sagebrush or bitterbrush as nesting structures. Long-billed curlews (*Numenius americanus*) and vesper sparrows (*Pooecetes gramineus*) have also been noted as commonly occurring species in shrub-steppe habitat. Songbird species that occur in riparian habitats include the red-winged blackbird (*Agelaius phoeniceus*), American robin (*Turdus migratorius*), black-billed magpie (*Pica pica*), song sparrow (*Melospiza melodia*), and dark-eyed junco (*Junco hyemalis*). Species known or expected to nest in riparian habitat are Brewer's blackbird (*Euphagus cyanocephalus*), mourning dove (*Zenaida macroura*), black-billed magpie (*Pica pica*), northern oriole (*Icterus galbula*), lazuli bunting (*Passerina amoena)*, eastern kingbird (*Tyrannus tyrannus*), western kingbird (*Tyrannus verticalis*), and western wood peewee (*Contopus sordidulus*) (Neitzel et al. 2005).

The Hanford Reach serves as a resting area for neotropical migrant birds, migratory waterfowl, and shorebirds. The area between the old Hanford townsite and Vernita Bridge is closed to recreational hunting, and large numbers of migratory waterfowl find refuge in this portion of the river. Other species observed during this period include American white pelicans (*Pelecanus erythrorhynchos*), egrets (*Casmerodius albus*), doublecrested cormorants (*Phalacrocorax auritus*), coots (*Fulica americana*), and common loons (*Gavia immer*). Shoreline riparian communities are seasonally important for a variety of species. Willows trap food for waterfowl (e.g., Canada geese [*Branta canadensis*]) and birds that use shoreline habitat (e.g., Forster's tern [*Sterna forsteri*]) as well as providing nesting habitat for passerines (e.g., mourning doves [*Zenaida macroura*]) (Neitzel et al. 2005).

Common upland game bird species include the chukar partridge (*Alectoris chukar*), grey partridge (*Perdix perdix*), California quail (*Callipepla californicus*), and Chinese ring-necked pheasant (*Phasianus colchicus*). Greater sage grouse (*Centrocercus urophasianus*) and scaled quail (*Callipepla squamata*) are less common and are rarely seen. Greater sage grouse were observed on ALE Reserve during 1999 and 2000; however, a fire in 2000 destroyed potential greater sage grouse habitat, and it is unlikely that greater sage grouse will return in numbers until the vegetation has recovered to a point where it can support them (Neitzel et al. 2005). None of the upland birds are native to the area except the sage grouse.

**Reptiles and Amphibians.** Seven species of reptiles and six species of amphibians are found at the Hanford Site. The side-blotched lizard (*Uta stansburiana*) is the most abundant reptile (Neitzel et al. 2005). The short-horned lizard (*Phrynosoma douglassii*) and northern sagebrush lizard (*Sceloporous graciosus*) are also common in mature sagebrush habitats with sandy soil. Commonly encountered snakes include the gopher snake (*Pituophis melanoleucus*), yellow-bellied racer (*Coluber constrictor*), and western rattlesnake (*Crotalus viridis*). Less common is the striped whipsnake (*Masticophis taeniatus)*. Amphibians on the Hanford Site are associated with riparian habitats located along the Columbia River or other permanent water bodies (Fitzner and Gray 1991). Species include the Great Basin spadefoot toad (*Spea intermontana*), western toad (*Bufo boreas*), Woodhouses toad (*Bufo woodhouseii*), the Pacific tree frog (*Hyla regilla*), tiger salamander (*Ambystoma tigrinum*), and bullfrog (*Rana catesbeiana*) (Neitzel et al. 2005). These reptiles and amphibians also occur at offsite locations.

**Terrestrial Invertebrates.** Most of the terrestrial invertebrate species on the Hanford Site and offsite locations are insects and spiders. Butterflies, grasshoppers, and darkling beetles represent some of the more conspicuous insect groups. The populations of all three of these species of insects are subject to seasonal changes and weather variations (Rogers and Rickard 1977). Many of the insect species are important in the food web of birds and mammals found on the Hanford Site. Species like the darkling beetle play an important role in the decomposition process by feeding on decaying plant material, animal

feces, fungi, and live plant tissue (Weiss and Mitchell 1992). Spiders are also abundant, especially in the riparian and shrub-steppe habitat (DOE 2001).

The Nature Conservancy has identified nearly 1680 species of insects on the Hanford Site (Evans et al. 2003). A collection of 12,000 specimens in 2003 resulted in the identification of 376 taxa and an estimated 150-200 new findings in the state of Washington, 46 from Hanford studies conducted over the last decade. Numerous species not previously collected at Hanford, especially in the orders Trichoptera (caddisflies) and Lepidoptera (moths), have been added to the invertebrate fauna of the Hanford Site. The actual number of insect species occurring on the Hanford Site may reach as high as 15,500 (Neitzel et al. 2005).

**Distribution of Wildlife.** Because the habitats of the Central Plateau are considerably different from those near the Columbia River, terrestrial animals are described separately for those locations in the following paragraphs.

*Central Plateau:* A characterization study of small mammals performed south of the 200 East Area resulted in the trapping of the following five species: the Great Basin pocket mouse, deer mouse, northern grasshopper mouse, sagebrush vole, and western harvest mouse (Rogers and Rickard 1977). The Great Basin pocket mouse represented more than 90 % of the individuals caught. Medium- and large-size mammals that may occur in the Central Plateau include rabbits, coyotes, badgers, and mule deer (Rogers and Rickard 1977). Some of these organisms are receptors in the SLERA. Other mammals potentially using areas associated with ponds and ditches in the 200 Area include muskrats, porcupines, and raccoons (DOE 1999). Many common bird species, such as the western meadowlark and sage sparrow, are likely to occur on the Central Plateau where suitable habitats exist. Thirty-seven species of terrestrial birds were recorded during surveys conducted in the 200 Area in 1986 (Schuler et al. 1993).

Unique habitats can be found on Gable Butte and Gable Mountain situated north of the Central Plateau. These unique habitats include basalt outcrops, scarps, and scree slopes. Birds likely to occur in these habitats are the prairie falcon, rock wren, poorwill, and chukar; small mammals include the yellow-bellied marmot and wood rat; reptiles include rattlesnakes, gopher snakes, and horned lizards (Downs et al. 1993).

*Columbia River:* Terrestrial wildlife species use both shoreline riparian and shrub-steppe habitats occurring along the Columbia River and on the islands. Wildlife reported to use the Hanford Reach includes 184 species of birds, 36 species of mammals, 9 species of reptiles, and 4 species of amphibians (NPS 1994). The Canada goose uses islands along the Hanford Reach extensively for nesting. Monitoring of nesting geese that use the Hanford Site has been ongoing since 1950. These studies indicate that Canada geese nest more frequently on islands in the downstream reach because of heavy predation by coyotes further upstream (Neitzel et al. 2005). Mule deer use the islands and other riparian areas for fawning habitat. Wildlife occurring in shoreline habitat includes 46 species that use willow communities and 49 species that use grass areas (NPS 1994).

The Hanford Reach begins at the foot of Priest Rapids Dam in the northwest portion of the area within a 50-km radius of the WTP stacks. It extends through the Hanford Reservation to the reservoir of McNary Dam, just north of the city of Richland. The Hanford Reach includes a variety of habitat types, including those also found outside the Hanford Reach but within the 50 km radius. Therefore, biota in and outside of the Hanford Reach are expected to be similar. Evaluating risks wherever the concentration in the Columbia River is highest helps ensure that biota in the 50 km radius are protected.

#### 8.1.2.3 Aquatic Ecosystems

Washington State has classified the stretch of the Columbia River that includes the Hanford Reach as Class A, Excellent (Neitzel et al. 2005). Class A waters must be suitable for essentially all uses, including raw drinking water, recreation, and wildlife habitat. Water from the Columbia River is used for both irrigation and municipal water supplies. Federal and state drinking water quality standards apply to the Columbia (Neitzel et al. 2005). Water samples from the Columbia River and three ponds on the Hanford Site are routinely collected and analyzed.

The Columbia River supports an ecosystem of plankton, benthic invertebrates, fish, and other communities. Algae are abundant in the river and provide food for herbivores, such as immature insects, which are then eaten by carnivorous species, such as bass. Aquatic plants in the Hanford Reach include water milfoil, waterweed, pondweed, Columbia yellowcress, watercress, and duckweed. Water milfoil is an aggressive, introduced aquatic plant and is becoming a nuisance in the river. Other aquatic species found in the Hanford Reach include microflora, zooplankton, and benthic invertebrates. Microflora include both sessile types (periphyton) and free-floating types (phytoplankton). Microflora species include diatoms, golden or yellow-brown algae, green algae, blue-green algae, red algae, and dinoflagellates. Dominant zooplankton taxa include *Bosmina*, *Diaptomus*, and *Cyclops*. Benthic invertebrate taxa occurring in the Hanford Reach include insect larvae such as caddisflies (*Trichoptera*), midge flies (*Chironomidae*), and black flies (*Simuliidae*); clams (*Corbicula* spp., *Anodontia* spp.), snails (*Physa* spp.), freshwater sponges (*Spongilla* spp.), limpets (*Fisherola* spp.), and crayfish (*Astacus trowbridgii*) are also present (Neitzel et al. 2005).

The Hanford Reach and adjacent reaches of the Columbia River support over 40 species of fish. The anadromous chinook salmon (*Oncorhynchus tshawytscha*), sockeye salmon (*Oncorhynchus nerka*), coho salmon (*Oncorhynchus kisutch*), and steelhead trout (*Oncorhynchus mykiss*) use the river to migrate to and from upstream spawning areas. Chinook salmon and steelhead trout also spawn in the Hanford Reach in the fall (Figure 8-7). Shad (*Alosa sapidissima*) may also spawn in this stretch of river. Mountain whitefish (*Prosopium williamsoni*), white sturgeon (*Acipenser transmontanus*), smallmouth bass (*Micropterus dolomieui*), crappie (*Pomoxis nigromaculatus*), catfish (*Ictalurus punctatus*), walleye (*Stizostedion vitreum*), and yellow perch (*Perca flavescens*) are important game fish to sport fisherman and American Indians. A healthy rough fish population includes carp (*Cyprinus carpio*), redside shiner (*Richardsonius balteatus*), suckers (*Catostomus macrocheilus*), and northern squawfish (*Ptychocheilus oregonensis*) (Neitzel et al. 2005).

West Lake, near the 200 Area, is created by a rise in the water table under the Central Plateau and is not fed by surface flow. This results in the pond being highly saline, as well as alkaline, and having low species diversity (DOE 1999). West Lake, located southwest of Gable Mountain, fluctuates in size with changes in the water table. The water level and size of the lake have been decreasing over the past several years because of reduced wastewater discharge (Neitzel et al. 2005). Unlike other ponds on the Hanford Site, West Lake does not receive direct effluent discharges from Hanford Site facilities (PNL 1993). Wetland vegetation found at West Lake is limited to scattered patches of emergent macrophytes, such as cattails and bulrushes. No jurisdictional wetland has been identified at West Lake.

Other wetland habitats found on the Hanford Site are associated with man-made ponds and ditches occurring on the Hanford Site, including a small cooling and wastewater pond in the 400 Area and the gravel pit converted to wetland at the 100-B Area. These artificial water bodies, formed by the wastewater discharges from the operation of the separation facilities, no longer receive discharges.

#### 8.1.2.4 Threatened and Endangered Species

Species of concern on the Hanford Site and offsite locations include federally listed threatened and endangered (T&E) species, state-listed T&E species, state-listed candidate species, state-listed plant species of concern, and species of ethnobiological concern to American Indians. There are no federal- or state-listed endangered or threatened mammals, reptiles, amphibians, or invertebrates on the Hanford Site, but there are three species of fish, four species of birds, and thirteen species of plants listed as threatened or endangered by either the state or federal governments (PNNL 2010). Table 8-1 summarizes the endangered, threatened, candidate, and species of concern found at the Hanford Site.

The federal species of concern bald eagle (*Haliaeetus leucocephalus*) is found regularly along the Hanford Reach. The anadromous chinook salmon (*Oncorhynchus tshawytscha*) and the steelhead (*Oncorhynchus mykiss*) species are regulated as evolutionarily significant units (ESUs) by the National Oceanic and Atmospheric Administration (NOAA) Fisheries Service, based on historical geographic spawning areas. One ESU of the chinook salmon, the Upper Columbia River spring-run ESU, is listed as endangered (Federal Register 1999 [64 FR 14308]). The Middle Columbia River steelhead ESU is listed as threatened (Federal Register 1999 [64 FR 14517]). The Upper Columbia River ESU is the portion of the Columbia River between the US-Canada border and the Yakima River, and it includes the Hanford Reach. One additional threatened fish species (bull trout) has been recorded on the Hanford Site but is believed to be transient (Neitzel et al. 2005).

Washington State lists the American white pelican (*Pelecanus erythrorhynchos*) and sandhill crane (*Grus canadensis*) as endangered, and the ferruginous hawk (*Buteo regalis*) and greater sage grouse (*Centrocercus urophasianus*) as threatened. The bald eagle (*Haliaeetus leucocephalus*), the peregrine falcon (*Falco peregrinus*), and the common loon (*Gavia immer*) are listed as sensitive. The American white pelican is a year-round resident (DOE 2001), the sandhill crane is a rare fall and spring visitor (DOE 2001), and the ferruginous hawk is a breeding resident. The bald eagle is a regular winter resident along the Columbia River (Neitzel et al. 2005). The peregrine falcon is a casual migrant to the Hanford Site between November and January (DOE 2001). The common loon is present year-round (DOE 2001); the temporal habits of the rarely seen greater sage grouse on the Hanford Site are not known (WHC 1992a).

Thirteen species of plants listed by Washington State as T&E are found on the Hanford Site. Two are listed as endangered: Columbia yellowcress (*Rorippa columbiae*) and Umtanum desert buckwheat (*Erigonium codium*); eleven plant species are listed as threatened: awned halfchaff sedge (*Lipocarpha (= Hemicarpha) aristulata*), chaffweed (*Anagallis (=Centunculus) minimus*), desert dodder (*Cuscuta denticulate*), Geyer's milkvetch (*Astragalus geyeri*), grand redstem (*Ammannia robusta*), Great Basin gilia (*Gilia leptomeria*), loeflingia (*Loeflingia squarrosa var. squarrosa*), lowland toothcup (*Rotala ramosior*), rosy pussypaws (*Calyptridium roseum*), White Bluffs bladderpod (*Lesquerella tuplashensis*), and white eatonella (*Eatonella nivea*). The awned halfchaff sedge, chaffweed, Columbia yellowcress, grand redstem, and lowland toothcup are restricted to wetlands in the riparian zone of the Columbia River. Other plant species, such as Great Basin gilia, loeflingia, and rosy pussypaws, are small annuals that have been found in relatively undisturbed sagebrush areas in the vicinity of Gable Mountain. The Great Basin gilia has been identified within the Hanford Reach National Monument. The remaining three state threatened-plant species (Geyer's milkvetch, white eatonella, and desert dodder) have been found at various sites on the Wahluke slope. Two species of plants are candidates for federal protection: the Umtanum desert buckwheat, which occurs in several small, highly localized populations on Umtanum Ridge, and the White Bluffs bladderpod, which occurs on the White Bluffs (Neitzel et al. 2005).

Wildlife state-listed candidate species observed or considered likely to be found on or near the Central Plateau include the sage sparrow (*Amphispiza belli*) and the loggerhead shrike (*Lanius ludovicianus*). Both of these birds commonly nest in undisturbed shrub-steppe habitat. The sage sparrow is one of the most common nesting birds on the Hanford Site (Downs et al. 1993). Other state-listed candidate bird species that may be found include the burrowing owl (*Athene cunicularia*), golden eagle (*Aquila chrysaetos*), sage thrasher (*Oreoscoptes montanus*), and merlin (*Falco columbarius*) (Neitzel et al. 2005). Another state-listed candidate species of concern inhabiting the Central Plateau and vicinity is the striped whipsnake (*Mastocophis taeniatus*).

**Central Plateau.** No federally or state-listed T&E plant or animal species occur in the Central Plateau (DOE 1999). Several state-listed plant species are found on the Central Plateau.

Wildlife species of state concern occurring in the 200 Area include the loggerhead shrike and sage sparrow. Both species nest in undisturbed sagebrush habitat in the Central Plateau (PNL 1993). Other listed T&E bird species that may occur in shrub-steppe habitat in the Central Plateau are the burrowing owl and golden eagle. Reptile species of concern using the Central Plateau include the striped whipsnake (*Masticophis taeniatus*) (Rogers and Rickard 1977, Neitzel et al. 2005).

**Columbia River.** No federally listed T&E plant species occur on the Hanford Reach (DOE 1999). State-listed endangered plant species occurring along the Hanford Reach include the Columbia yellowcress. Preferred habitat for persistent sepal yellowcress is shoreline areas with gently sloping, cobbly substrate (PNL 1993). State-listed plant species of concern have been found along the shoreline and on islands of the Hanford Reach between the Vernita Bridge and the 300 Area, including the southern mudwort, dense sedge, and shining flatsedge (WHC 1992b).

State-listed endangered bird species that occur along the Hanford Reach that are considered relatively common include the American white pelican (*Pelecanus erythrorhynchos*) and sandhill crane. State-listed sensitive species include the common loon (*Gavia immer*), the peregrine falcon, and the bald eagle. The common loon is found within the Hanford Reach. The peregrine falcon is an incidental migrant on the Hanford Site (Neitzel et al. 2005). The bald eagle is a relatively common winter resident along the Hanford Reach that occasionally attempts to nest on the Hanford Site. However, bald eagles have never successfully nested on the Hanford Site (Neitzel et al. 2005).

Of the three federally listed fish species, only the upper Columbia River steelhead trout spawns in the Hanford Reach. Upper Columbia River spring Chinook salmon adults pass through the Hanford Reach while migrating to spawning grounds, and the juveniles use the Hanford Reach as a nursery area while they migrate toward the ocean. The bull trout, which primarily inhabits smaller streams at higher elevations, has been observed in the Hanford Reach on very rare occasions, usually associated with the spring freshets. Bull trout are not considered to be residents of the Hanford Site (Neitzel et al. 2005).

#### 8.1.2.5 Sensitive Environments

Sensitive habitats on the Hanford Site include shrub-steppe, and wetlands and riparian habitats (Table 8-1). Shrub-steppe ecosystems are typified by a shrub overstory and a grass and forb understory. Lichens and mosses, often times referred to as “microbiotic or cryptogamic crust,” provide a soil-stabilizing growth on undisturbed soils in the shrub-steppe ecosystem. The dominant vascular plants in the area are big sagebrush, underlain by perennial bunchgrasses and forbs. Over 700 species representing 90 families of vascular plants are recorded for the Hanford Site (Neitzel et al. 2005). Wetlands include those transitional lands occurring between terrestrial and aquatic ecosystems where the

water table is usually close to the surface or where shallow water covers the surface (Cowardin et al. 1979). The primary wetlands found on site occur along the Hanford Reach of the Columbia River and include the riparian habitats located along the river shoreline. Other wetland habitats found on the Hanford Site are associated with man-made ponds and ditches. The variety of habitat on the Hanford Site creates special ecological areas. For example, the Hanford Site includes nesting sites for bird species of concern, salmon and steelhead spawning areas, riparian habitat, and part of the largest remaining tract of shrub-steppe habitat in the Columbia Basin (DOE 1999).

There are also special ecological areas outside the Hanford Site but within the area included in deposition modeling. These include the Columbia National Wildlife Refuge, which extends from approximately 30 km to approximately 50 km north of the WTP, and habitats classified as priority habitats by the state of Washington. Priority habitats near the Hanford Site include in-stream and riparian habitats on the Columbia and Yakima rivers, Crab Creek, and shrub-steppe habitat types surrounding the Hanford Site.

The SLERA will implicitly include the special ecological areas because it will use the maximum soil or sediment and water concentrations either within the Hanford Site or the adjacent offsite area and because it will assume that all representative ecological receptors are present at each of the maximum deposition locations regardless of habitat. Therefore, ecological receptors within the special ecological areas outside the Hanford Site will have lower exposures than the receptors evaluated in the SLERA.

### 8.1.3 Receptor Identification

The receptors present in the ecological setting and habitats at Hanford and offsite locations will be exposed by routes that are defined by how the receptors live and what they eat. Food webs represent the transfer of matter among the components of an ecosystem. This transfer occurs through the uptake and absorption of substances from abiotic media or consumption of animal and plant tissue. Figure 8-8 shows the food web representing the terrestrial organisms and their general trophic relationships. Figure 8-9 shows the food web representing the aquatic organisms and their general trophic relationships. The food webs highlight the selected terrestrial receptors (Figure 8-8) and also aquatic receptors (Figure 8-9).

#### 8.1.3.1 Terrestrial Receptors

Figure 8-8 presents a simplified food web for selected terrestrial receptors. The receptors selected for use in the SLERA and their trophic levels are shown in bold in the figure:

- Plants (Trophic Level 1): cheatgrass, rabbitbrush
- Terrestrial invertebrates (Trophic Level 2): earthworms, darkling beetles
- Herbivorous mammals (Trophic Level 2): mule deer
- Herbivorous birds (Trophic Level 2): mourning dove
- Omnivorous mammals (Trophic Level 3): Great Basin pocket mouse
- Omnivorous birds (Trophic Level 3): western meadowlark
- Carnivorous mammals (Trophic Level 4): coyote
- Carnivorous birds (Trophic Level 4): burrowing owl, red-tailed hawk

The reasons for selecting the representative receptors are given below. The conservative approach is to assume receptors are exposed to air, soil concentrations, and terrestrial plants at the onsite ground maximum, but are only exposed to water at the Columbia River maximum.

**Terrestrial Plants.** Terrestrial plants are essential to the function of any terrestrial ecosystem and are a major route of entry of contaminants into the food web; therefore, terrestrial plant populations will be evaluated in the SLERA. Terrestrial plants are assumed to be exposed to the onsite ground maximum by direct uptake of COPCs in volatile emissions, uptake of COPCs and ROPCs deposited on leaf surfaces, root uptake from soil, external exposure to radionuclides in soil, and external exposure to radionuclides in soil and air.

**Terrestrial Invertebrates.** Terrestrial invertebrates are essential to the function of any terrestrial ecosystem and are a major route of entry of contaminants into the food web. The number of earthworms at the Hanford Site is expected to be low because of the aridity of most of the habitat. However, there are more data available to evaluate exposure of earthworms than there is for other terrestrial invertebrates. Therefore, earthworm populations will be evaluated as representatives of terrestrial invertebrates in the SLERA. Earthworms are assumed to be exposed to the onsite ground maximum by uptake of COPCs and ROPCs deposited on soil and by external exposure to ROPCs in soil and air. There are no uptake factors for transfer of COPCs from air to terrestrial invertebrates that are separate from the experimental soil exposures used to derive the uptake factors.

**Mule Deer.** Mule deer populations are evaluated as representative of herbivorous mammals that consume vegetation contaminated by COPCs and ROPCs. Mule deer are assumed to be exposed by ingestion of COPCs and ROPCs deposited on soil, by ingestion of plants containing COPCs and ROPCs taken up from soil, by ingestion of surface water containing COPCs and ROPCs, and by external radiation from soil and air. The predominant diet of the mule deer is browse.

**Mourning Dove.** Mourning dove populations are evaluated as representative of herbivorous birds that consume vegetation contaminated by COPCs and ROPCs. The mourning dove is assumed to be exposed by ingestion of COPCs and ROPCs deposited on soil, by ingestion of plants (mainly grass seeds) containing COPCs and ROPCs taken up from soil, by ingestion of surface water containing COPCs and ROPCs, and by external radiation from soil and air.

**Great Basin Pocket Mouse.** Great Basin pocket mouse populations are evaluated as representative of omnivorous mammals. The Great Basin pocket mouse is assumed to be exposed by ingestion of COPCs and ROPCs deposited on soil, by ingestion of plants (mainly grass seeds) and terrestrial invertebrates containing COPCs and ROPCs taken up from soil, and by external radiation from soil and air. The mouse is assumed to get its water through food sources and thus ingestion of surface water containing COPCs and ROPCs is not applicable for the mouse.

**Western Meadowlark.** Western meadowlark populations are evaluated as representative of omnivorous birds. The meadowlark is assumed to be exposed by ingestion of COPCs and ROPCs deposited on soil, by ingestion of plants (mainly grass seeds) and terrestrial invertebrates containing COPCs and ROPCs taken up from soil, by ingestion of surface water containing COPCs and ROPCs, and by external radiation from soil and air.

**Coyote.** Coyote populations are evaluated as representative of carnivorous mammals. The coyote is assumed to be exposed by ingestion of COPCs and ROPCs deposited on soil, by ingestion of small mammals and birds containing COPCs and ROPCs taken up from soil, by ingestion of surface water containing COPCs and ROPCs, and by external radiation from soil and air.

**Burrowing Owl.** Burrowing owl populations are evaluated as representative of carnivorous birds. The

burrowing owl is assumed to be exposed by ingestion of COPCs and ROPCs deposited on soil, by ingestion of small mammals and birds containing COPCs and ROPCs taken up from soil, by ingestion of surface water containing COPCs and ROPCs, and by external radiation from soil and air.

**Red-Tailed Hawk.** Red-tailed hawks are evaluated as representative of federal- and state-listed carnivorous birds of special interest, although the bird itself is not a federal- or state-listed species. The red-tailed hawk is assumed to be exposed by ingestion of small mammals and birds containing COPCs and ROPCs taken up from soil, by ingestion of surface water containing COPCs and ROPCs, and by external radiation from soil and air.

#### 8.1.3.2 Aquatic Receptors

Figure 8-9 presents a simplified food web of selected aquatic receptors. The receptors selected for use in the SLERA are shown in bold on the figure and are listed below:

- Plants (Trophic Level 1): aquatic plants and plants rooted in sediment
- Benthic invertebrates (Trophic Level 2): sediment-dwelling clams and insects
- Aquatic organisms, fish, and other aquatic biota (Trophic Levels 2 through 4): bass, salmon, channel catfish, water fleas, other invertebrates
- Herbivorous waterfowl (Trophic Level 2): Canada goose
- Shorebirds (Trophic Level 3): spotted sandpiper
- Piscivorous birds (Trophic Level 4): great blue heron, bald eagle
- Piscivorous mammals (Trophic Level 4): mink

The reasons for selecting the representative receptors are given below. The conservative approach taken is to assume receptors are exposed to water, sediment concentrations, and aquatic plants at the Columbia River maximum, but air concentrations from the onsite maximum.

**Aquatic Plants.** Aquatic plants are important to the function of an aquatic ecosystem. Plankton, floating plants, and emergent plants contribute to the base of the food web. However, because of the lack of toxicity information, their risk is not quantified. They are handled as ingestion exposure to aquatic herbivores and omnivores.

**Benthic Invertebrates.** Benthic invertebrates are essential for the functioning of an aquatic ecosystem and are a major route of entry of contaminants into aquatic food webs. Therefore, benthic invertebrates will be evaluated in the SLERA. Benthic invertebrates are likely to be present in the Columbia River at the location of maximum deposition of COPCs and ROPCs. Benthic invertebrates are assumed to be exposed by uptake from sediment and by external radiation from water and sediment.

**Aquatic Biota.** Aquatic biota are essential for the functioning of an aquatic ecosystem and are a major route of entry of contaminants into aquatic food webs. Therefore, aquatic biota populations will be evaluated in the SLERA. Aquatic biota are likely to be present in the Columbia River at the location of maximum deposition of COPCs and ROPCs. Aquatic biota are assumed to be exposed by uptake from surface water, ingestion of food containing COPCs and ROPCs taken up from water, and by external radiation from water and sediment.

**Salmonids.** Chinook salmon and steelhead trout populations in the Hanford Reach of the Columbia

River have been designated ESUs (Neitzel et al. 2005). Therefore, special care must be taken to prevent harm to these salmonids. Salmonids are also fish species of special interest because of their economic and recreational importance and, as carnivorous fish, they are at the top of aquatic food webs. Salmonids are also of particular cultural importance to the American Indian tribes, whose way of life has inextricably included salmon and trout as food throughout their history. Therefore, salmonid populations will be evaluated in the SLERA. Salmonids are likely to be present in the Columbia River at the location of maximum deposition of COPCs and ROPCs. Salmonids are assumed to be exposed by uptake from surface water, ingestion of food containing COPCs and ROPCs taken up from water, and by external radiation from water and sediment.

**Canada Goose.** Canada goose populations are evaluated as representative of herbivorous birds that consume vegetation contaminated by COPCs and ROPCs from water. Because the Canada goose is a year-round resident at the Hanford Site (DOE 2001), it could be expected to spend its life at the location of maximum deposition of COPCs and ROPCs. The Canada goose is assumed to be exposed by uptake from ingested surface water and sediment, ingestion of vegetation that contains COPCs and ROPCs taken up from sediment and water, and external radiation from water and air.

**Spotted Sandpiper.** Spotted sandpiper populations are evaluated as representative of carnivorous birds that consume benthic invertebrates contaminated by COPCs and ROPCs from near-shore sediment. The spotted sandpiper resides along the shores of the Columbia River, where it preys on aquatic and terrestrial invertebrates and small fish. It represents the group of carnivorous shorebirds that are exposed to contaminants in aquatic biota, benthic organisms, and water. The spotted sandpiper could be expected to spend its life at the location of maximum deposition of COPCs and ROPCs. The spotted sandpiper is assumed to be exposed by uptake from ingested surface water and sediment, ingestion of benthic invertebrates that contain COPCs and ROPCs taken up from sediment, and external radiation from water and air.

**Great Blue Heron.** Great blue heron populations are evaluated as representative of carnivorous birds that consume small fish contaminated by COPCs and ROPCs from water. The great blue heron could be expected to spend its life at the location of maximum deposition of COPCs and ROPCs. The great blue heron is assumed to be exposed by uptake from ingested surface water, ingestion of omnivorous and planktivorous fish that contain COPCs and ROPCs taken up from water, ingestion of benthic invertebrates exposed by uptake from sediment, and external radiation from water and air.

**Bald Eagle.** Bald eagle populations are evaluated as representative of carnivorous birds of special interest that consume omnivorous fish contaminated by COPCs and ROPCs from water. The bald eagle is known to nest along the Columbia River, but often leaves the area before laying eggs (WHC 1994). Resident eagles are exposed to contaminants in fish as well as waterfowl, small mammals, and carrion, on which they prey. The bald eagle is the best representative of top predators of aquatic biota on the Hanford Site. For conservatism in the SLERA, the bald eagle will be assumed to be exposed year-round by ingestion of surface water, fish that contain COPCs and ROPCs taken up from water, and external radiation from water and air.

**Mink.** Mink populations are evaluated as representative of carnivorous mammals that consume omnivorous fish contaminated by COPCs and ROPCs from water. The mink could be expected to spend its life at the location of maximum deposition of COPCs and ROPCs. The mink is assumed to be exposed by uptake from ingested surface water, ingestion of fish that contain COPCs and ROPCs taken up from water, and external radiation from water and air.

#### 8.1.3.3 Species Profiles

Quantitative descriptions of the receptor species are necessary to model exposure to COPCs and ROPCs. The following species profiles for mammals and birds provide the necessary quantitative information for each receptor, as well as text describing the species and its relation to the Hanford Site. Species profiles are not required for plants (cheatgrass and rabbitbrush) and terrestrial invertebrates (earthworms and darkling beetles) because exposures of these receptors are not modeled using receptor-specific parameters. Similarly, species profiles are not required for the following:

- Benthic invertebrates (clams, insects, snails, and worms)
- Planktivorous fish and small invertebrates (small carp, small northern squaw fish, small suckers, water fleas, and other invertebrates)
- Fish (bass, salmon, and channel catfish)

**Mule Deer (*Odocoileus hemionus*)**

Mule deer, with an onsite herd of several hundred, occur just about everywhere on the Hanford Site, but are most often found near the Columbia River. Mule deer use the islands and other riparian areas as fawning habitat. Bitterbrush provides important browse for the resident mule deer herd. Summer browse is chiefly herbaceous plants and the young shoots of woody plants, while winter browse includes twigs of woody plants and trees, including cedar, yew, aspen, willow, dogwood, juniper, and sage. Coyotes are a major predator, along with bobcats to a lesser extent. Mule deer are most active in the mornings and evenings.

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 66.5 | Average of males and females, north central Colorado (Sample et al. 1997) |
| *ED* | Exposure duration (longevity) (yrs) | 20 | (Anderson and Wallmo 1984) |
| *HR* | Home range (ha) | 285 | (Sample et al. 1997) |
| *TUF* | Temporal use factor | 1 | Will be 1 unless specific value exists for a receptor |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.035 | Adjusted from 0.022 kg/kg BW dry weight per day (Sample et al. 1997) by assuming a 37 % moisture content in browse (Neuenschwander 1980) |
| $F_P$ | Plant fraction | 1 | (Sample et al. 1997) |
| $F_A$ | Animal fraction | 0 | (Sample et al. 1997) |
| *SFr* | Soil fraction | 0.02 | (Arthur and Alldredge 1979 in Beyer et al. 1994) |
| $IR_w$ | Water ingestion rate (L/kg BW/d) | 0.044 | Average of mid-range values for winter and summer reported by Sample et al. (1997) |

[a] Food ingestion rate (g/g/d) re-expressed as kg/kg BW/d is assumed not to include ingested soil; therefore, $F_P + F_A = 1.0$.

## Mourning Dove (*Zenaidura macroura*)

The mourning dove has the widest distribution of any North American game bird; it is the only species nesting in all 48 contiguous US states. During the winter it lives in small to large flocks where food is plentiful and good roosting and protective cover are available in nearby trees. The mourning dove feeds mostly on the ground in harvested crop fields, and along railroads and roadsides. About 98 % of its diet in all seasons is seeds. It eats some insects and snails, and picks up grit from gravel roads or sea beaches. It nests from southeastern Alaska to western Panama, and it winters from southern Canada, but mainly from northern California, south into Central America.

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 0.128 | Numerical average of males and females (Martin and Nelson 1952 in Terres 1980) |
| *ED* | Exposure duration (longevity) (yrs) | 19.3 | (Carey and Judge 2001) |
| *HR* | Foraging distance (km) | ≤ 1 | (CDFG 2003) |
| *TUF* | Temporal use factor | 1 | Will be 1 unless specific value exists for a receptor |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.212 | Calculated by allometric equation, $0.398 \times BW(g)^{0.85}/BW(g)$ (EPA 1993a, Eq. 3-4), adjusted to wet-weight basis by assuming a water content of 9.3 % for seeds (EPA 1993a, Table 4-2): 0.192 / (1-0.093) = 0.212 |
| $F_P$ | Plant fraction | 1 | Diet stated to be >98 % seeds and other vegetation (Terres 1980) |
| $F_A$ | Animal fraction | 0 | <2 % invertebrates (Terres 1980) |
| *SFr* | Soil fraction | 0.09 | Assumed to be 10 % of dry weight of diet (EPA 1999) : 0.1 × (1 – 0.093) = 0.09 |
| $IR_w$ | Water ingestion rate (L/kg BW/day) | 0.116 | Calculated by using allometric equation, $0.059 \times BW\ (kg)^{0.67}/BW(kg)$ [EPA 1993a, Eq. 3-15] |

[a] Food ingestion rate (g/g/d) re-expressed as kg/kg BW/d is assumed not to include ingested soil; therefore, $F_P + F_A = 1.0$.

### Great Basin Pocket Mouse (*Perognathus parvus*)

The Great Basin pocket mouse eats mostly seeds, but also eats insects (Fitzner and Gray 1991). It is the principal prey of the burrowing, great horned, long-eared, and barn owls at the Hanford Site (Downs et al. 1993) and serves as a vector for contaminant movement through the food chain. The Great Basin pocket mouse is a nocturnal, burrowing mammal, with most burrows being between 35 cm and 193 cm (1.2 ft to 6.3 ft) deep (Gano and Rickard 1982). The mouse has no need for drinking water, obtaining all its water from its food. Its small home range could cause it to spend all of its time within a contaminated area and obtain all food there (DOE 1999).

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 0.016 | Average, males and females, Washington State (Sample et al. 1997) |
| *ED* | Exposure duration (longevity) (yrs) | 8.0 | Value for pocket mouse (*Perognathus spp.*) (Carey and Judge 2001) |
| *HR* | Home range (ha) | 0.14 | Mid-range for females, Washington State (Sample et al. 1997) |
| *TUF* | Temporal use factor | 1 | (DOE 1999) |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.285 | (Calder 1984 in DOE-RL 1995) |
| $F_P$ | Plant fraction | 0.62[b] | Annual average (based on four seasons normalized to 100% and then averaged), Colorado, short-grass prairie (EPA 1993a) |
| $F_A$ | Animal fraction | 0.38[b] | Annual average, Colorado, short-grass prairie (EPA 1993a) |
| *SFr* | Soil fraction | 0.01 | Estimated 2 % of dry weight of diet (Beyer et al. 1994). Dry weight is estimated to be 57 % of a mixed diet of 55 % seeds with 9.3 % water content and 45 % terrestrial invertebrates with 84 % water content (EPA 1993a, Tables 4-1 and 4-2). |
| $IR_w$ | Water ingestion rate (L/kg BW/d) | 0.0 | (Price 1983) |

[a] Food ingestion rate (g/g/d) expressed as kg/kg BW/d does not include ingested soil; therefore, $F_P + F_A = 1.0$.
[b] Values used for the Great Basin pocket mouse taken from values established for the deer mouse (*Peromyscus maniculatus*). (Flake 1973 in EPA 1993a)

### Western Meadowlark (*Sturnella neglecta*)

The western meadowlark is a ground-nesting bird that nests in cheatgrass and sagebrush-bunchgrass communities (Rickard et al. 1988, Schuler et al. 1988). The western meadowlark is a common, omnivorous bird of open habitats in southeastern Washington State and is abundant in the shrub-steppe ecosystem (Schuler et al. 1988). It feeds on a variety of items, which include both insects and plant material, mostly seeds. One study (Bent 1958 in Sample et al. 1997) reports that the western meadowlark's diet consists of roughly 70 % insects and 30 % plant material. Studies conducted in southeastern Washington State indicate that it is the main bird prey item in the diets of the red-tailed, ferruginous, and Swainson's hawks (Rickard et al. 1988). Adult female western meadowlarks average 94.2 grams in weight and lay three to seven eggs in dome-shaped nests concealed in the grass or weeds.

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 0.094 | Adult female, Washington State (Sample et al. 1997) |
| *ED* | Exposure duration (longevity) (yrs) | 10.0 | Value for captive species (Carey and Judge 2001) |
| *HR* | Home range (ha) | 3.0 | Adult male, Wisconsin, average (Sample et al. 1997) |
| *TUF* | Temporal use factor | 1 | Will be 1 unless specific value exists for a receptor |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.028 | (Sample et al. 1997) |
| $F_P$ | Plant fraction | 0.30 | (Bent 1958 in Sample et al. 1997) |
| $F_A$ | Animal fraction | 0.70 | (Bent 1958 in Sample et al. 1997) |
| *SFr* | Soil fraction | 0.04 | Estimated 10.4 % of dry weight of diet of woodcock (Beyer et al. 1994) was used for the meadowlark. Dry weight is estimated to be 38 % of a mixed diet of 30 % seeds with 9.3 % water and 70 % terrestrial invertebrates with 84 % water content (EPA 1993a, Tables 4-1 & 4-2). |
| $IR_w$ | Water ingestion rate (L/kg BW/d) | 0.13 | Calculated using allometric equation, $0.059 \times BW(kg)^{0.67}/BW(kg)$ [EPA 1993a, Eq. 3-15] |

[a] Food ingestion rate (g/g/d) re-expressed as kg/kg BW/d is assumed not to include ingested soil; therefore $F_P + F_A = 1.0$.

**Coyote (*Canis latrans*)**

The coyote is the most common carnivore on the Hanford Site. They are nocturnal but may be active at any time of day. Primarily carnivorous, coyotes feed mainly on birds and small mammals, but also feed on insects and fruits in season. The typical hunting range is 10 miles, but may extend to 100 miles, reflecting the coyote's variable home range. Being an upper-trophic-level receptor, the coyote could be particularly susceptible to chemicals that bioaccumulate. Coyotes living in the shrub-steppe feed on pocket mice, northern pocket gopher, Nuttall's cottontail, black-tailed jackrabbit, and occasionally small mule deer. Favored den sites are riverbanks and the sides of canyons or gulches.

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 12.4 | Average of adult male and female from Iowa (Sample et al. 1997) |
| *ED* | Exposure duration (longevity) (yrs) | 21.8 | Value for captive species (Carey and Judge 2001) |
| *HR* | Home range (ha) | 3010 | Living singly or in pairs (Sample et al. 1997) |
| *TUF* | Temporal use factor | 1 | Will be 1 unless specific value exists for a receptor |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.018 | Desert coyote adults (Sample et al. 1997) |
| $F_P$ | Plant fraction | 0.02 | Average for western states (Sample et al. 1997) |
| $F_A$ | Animal fraction | 0.98 | Average for western states (Sample et al. 1997) |
| *SFr* | Soil fraction | 0.002 | Estimated soil ingestion rate divided by food ingestion rate |
| $IR_w$ | Water ingestion rate (L/kg BW/d) | 0.077 | Estimated (Sample et al. 1997) by using allometric equation, 0.099 × BW (kg)$^{0.90}$/BW(kg) [EPA 1993a, Eq. 3-17] |

[a] Food ingestion rate (g/g/d) re-expressed as kg/kg BW/d is assumed not to include ingested soil; therefore, $F_P + F_A = 1.0$.

## Burrowing Owl (*Athene cunicularia*)

The burrowing owl is the most abundant of the owls that nest on the Hanford Site. Burrowing owls nest in holes in the ground that are abandoned by burrowing mammals. Their diet consists of pocket mice, deer mice, pocket gophers, mountain voles, black-tailed jackrabbits, Nuttall's cottontail, rock doves, mallards, and American coots.

The burrowing owl is more diurnal than most owls. The female lays five to seven eggs in a long, underground burrow lined with grasses, roots, and dung. The burrows are usually abandoned prairie dog or pocket gopher burrows, but burrowing owls are capable of digging their own.

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 0.15 | Mean, males and females, throughout North America (Sample et al. 1997) |
| *ED* | Exposure duration (longevity) (yrs) | 8.7 | (Carey and Judge 2001) |
| *HR* | Home range (ha) | 241 | Mean, Saskatchewan (Sample et al. 1997) |
| *TUF* | Temporal use factor | 1 | Will be 1 unless specific value exists for a receptor |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.042 | Estimated (Sample et al. 1997) from reported energy requirement, average of winter and summer |
| $F_P$ | Plant fraction | 0 | Colorado (Sample et al. 1997) |
| $F_A$ | Animal fraction | 1 | Colorado (Sample et al. 1997) |
| *SFr* | Soil fraction | 0.1 | Estimated from mean of 5 % of volume (Thomsen 1971 in Sample et al. 1997) |
| $IR_w$ | Water ingestion rate (L/kg BW/d) | 0.11 | Estimated (Sample et al. 1997) by using allometric equation, 0.059 × BW (kg)$^{0.67}$/BW(kg) [EPA 1993a, Eq. 3-15] |

[a] Food ingestion rate (g/g/d) re-expressed as kg/kg BW/d is assumed not to include ingested soil; therefore, $F_P + F_A = 1.0$.

### Red-Tailed Hawk (*Buteo jamaicensis*)

The red-tailed hawk may be found on the Hanford Site year-round (Fitzner and Gray 1991). Forty-one nesting pairs of hawks (red-tailed, Swainson's, and ferruginous) were observed on site during the 1994 breeding season (Neitzel et al. 2005). Nests were constructed in trees, cliffs, basalt outcrops, and high-voltage transmission line towers (Neitzel et al. 2005). The red-tailed hawk is a diurnal predator of rodents and other small mammals, including mice, shrews, voles, rabbits, and squirrels. Generally opportunistic, the red-tailed hawk feeds on whatever is most abundant and readily available. Red-tailed hawks maintain a territory year-round (Brown and Amadon 1968).

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 1.06 | Average of adult male and female, southwest Idaho (Steenhof 1983 in EPA 1993a) |
| *ED* | Exposure duration (longevity) (yrs) | 18 | (Henny and Wight, 1970, 1972 in EPA 1993a) |
| *HR* | Home range (ha) | 1,770 | Adult, both male and female, Colorado upland prairie (Andersen and Rongstad 1989 in EPA 1993a) |
| *TUF* | Temporal use factor | 1 | Will be 1 unless specific value exists for a receptor |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.105 | Average of adult male and female, winter, Michigan, captive, outdoors (Craighead and Craighead 1956 in EPA 1993a) |
| $F_P$ | Plant fraction | 0 | Not stated in EPA 1993a; assumed to be negligible |
| $F_A$ | Animal fraction | 1 | Prey brought to nests in Alberta, Canada, Oregon, and California (EPA 1993a) |
| *SFr* | Soil fraction | 0 | Not stated in EPA (1993a) or Beyer et al. (1994); assumed to be negligible |
| $IR_w$ | Water ingestion rate (L/kg BW/d) | 0.057 | Average of adult male and female rates (EPA 1993a) estimated using the allometric equation, $0.059 \times BW(kg)^{0.67}$ /BW(kg) [EPA 1993a, Eq. 3-15] |

[a] Food ingestion rate (g/g/d) re-expressed as kg/kg BW/d is assumed not to include ingested soil; therefore, $F_P + F_A = 1.0$.

**Canada Goose (*Branta canadensis*)**

Canada geese forage primarily in open fields, feeding on grains, grass sprouts, and some aquatic vegetation. Breeding habitats include tall grass prairies and shortgrass prairies, marshes, ponds, and lakes. Most nesting sites are close to open water, often on islands (EPA 1993a). The Canada goose uses islands along the Hanford Reach extensively for nesting. Studies on the nesting habits of geese that use the Hanford Site have been ongoing since 1953. These studies indicate a general decline over the years in numbers of nests on islands in the Hanford Reach because of heavy predation by coyotes (Cushing et al. 1995).

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 3.72 | Average of adult male and female, Nova Scotia (EPA 1993a) |
| *ED* | Exposure duration (longevity) (yrs) | 24.3 | (Carey and Judge 2002) |
| *HR* | Home range (ha) | 983 | Adult female and brood, Washington State (EPA 1993a) |
| *TUF* | Temporal use factor | 1 | Considered a year-round resident at the Hanford Site (DOE 2001) |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.031 | Average of adult male and female, winter and spring, British Columbia interior (EPA 1993a) |
| $F_P$ | Plant fraction | 1 | North Carolina, lake; and Ontario, bay (EPA 1993a) |
| $F_A$ | Animal fraction | 0 | < 1 % invertebrates (EPA 1993a) |
| *SFr* | Sediment fraction | 0.07 | Estimated 8.2 % of dry weight of diet (Beyer et al. 1994). Dry weight is estimated to be 0.89 × wet weight for grain and seeds (EPA 1993a). |
| $IR_W$ | Water ingestion rate (L/kg BW/d) | 0.038 | Average of adult male and female, estimated (EPA 1993a) by using allometric equation, $0.059 \times BW(kg)^{0.67}/BW(kg)$ (EPA 1993a, Eq. 3-15) |

[a] Food ingestion rate (g/g/d) re-expressed as kg/kg BW/d is assumed not to include ingested soil; therefore, $F_P + F_A = 1.0$.

1 **Spotted Sandpiper (*Actitus macularia*)**

The spotted sandpiper requires open water for drinking, semi-open habitat for nesting, and dense vegetation for breeding (Bent 1929 and Oring et al. 1983 in EPA 1993a). The nest is a grassy scrape near water or in brush with a determinate clutch size of four eggs. Several clutches may be laid during a given breeding season. The diet of the spotted sandpiper consists mostly of terrestrial and aquatic insects (Bent 1929 in EPA 1993a), with adult flying insects making up the bulk of the diet (Oring et al. 1983 in EPA 1993a).

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 0.0425 | Arithmetic mean, adult, males and females, Minnesota (EPA 1993a) |
| *ED* | Exposure duration (longevity) (yrs) | 3.7 | (Oring et al., 1983 in EPA 1993a) |
| *HR* | Home range (ha) | 0.25 | Single value, sex not specified, Nova Scotia (EPA 1993a) |
| *TUF* | Temporal use factor | 1 | Will be 1 unless a specific value exists for a receptor |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.88 | Calculated by allometric equation, $0.648 \times BW(g)^{0.651}/BW(g)$ (EPA 1993a, Eq. 3-3), adjusted to wet- weight basis by assuming food moisture content of 80 % for benthic invertebrates (EPA 1993, Table 4-1). $IR_F = 0.175 / (1-0.8) = 0.88$ |
| $F_P$ | Plant fraction | 0 | None listed as dietary intake in EPA (1993a) |
| $F_A$ | Animal fraction | 1 | Benthic invertebrates, Minnesota, lake (EPA 1993a) |
| *SFr* | Sediment fraction | 0.036 | Estimated 18 % of dry weight of diet (Beyer et al. 1994). Dry weight is estimated to be 0.2 × wet weight for benthic invertebrates (EPA 1993a, Table 4-1). SFr = 0.18 × 0.2 = 0.036. |
| $IR_W$ | Water ingestion rate (L/kg BW/d) | 0.165 | Average of adult male and female rates (EPA 1993a), estimated by using allometric equation, $0.059 \times BW(kg)^{0.67}/BW(kg)$ (EPA 1993a, Eq. 3-15) |

[a] Food ingestion rate (g/g/d) re-expressed as kg/kg BW/d is assumed not to include ingested soil; therefore, $F_P + F_A = 1.0$.

### Great Blue Heron (*Ardea herodius*)

Great blue herons are year-round residents of the Hanford Reach. This bird is relatively common along the Hanford Reach (Fitzner and Gray 1991). Some of the trees planted on pre-1943 farms have persisted and serve as nesting platforms for several species of birds, including the great blue herons (DOE-RL 1995). Its nest is a platform of sticks lined with finer material and is sometimes found on the ground or in a reedbed. Principal prey items of the great blue heron are fish and frogs, although it will also feed on small mammals, reptiles, and occasionally birds.

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 2.39 | Arithmetic mean, adult, both sexes, location not stated (EPA 1993a) |
| *ED* | Exposure duration (longevity) (yrs) | 23.3 | (Carey and Judge 2001) |
| *HR* | Foraging range (km) | 3.1 | Foraging distance, mean, adults, both sexes, South Dakota, stream (EPA 1993a) |
| *TUF* | Temporal use factor | 1 | Will be 1 unless a specific value exists for a receptor |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.18 | (EPA 1993a) |
| $F_P$ | Plant fraction | 0 | None listed as dietary intake in EPA (1993a) |
| $F_A$ | Animal fraction | 1 | 98 % aquatic vertebrates, a river in lower Michigan (EPA 1993a) |
| *SFr* | Sediment fraction | 0 | Not reported in EPA (1993a); assumed to be negligible |
| $IR_W$ | Water ingestion rate (L/kg BW/d) | 0.045 | Estimated (EPA 1993a) by using allometric equation, $0.059 \times BW(kg)^{0.67}/BW$ (kg) (EPA 1993a, Eq. 3-15) |

[a] Food ingestion rate (g/g/d) re-expressed as kg/kg BW/d is assumed not to include ingested soil; therefore, $F_P + F_A = 1.0$.

## Bald Eagle (*Haliaetus leucocephalus*)

The bald eagle is a common winter resident, usually arriving in October. These birds forage throughout the Hanford Reach. Bald eagles use trees during the day for perching and occasionally at night for communal roosts (DOE 1999). Wintering eagles tend to concentrate where food is abundant and human disturbance is minimal. The diet of bald eagles varies locally as well as seasonally. Food may vary from spawned salmon and waterfowl (often killed by other predators or disease) during the winter to fish, small mammals, carrion, and waterfowl during the breeding season (EPA 1993a). Although bald eagles exhibit nesting behavior at the Hanford Site, most leave before laying eggs (WHC 1994).

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 3.75 | Arithmetic mean, adult, both sexes, Florida (EPA 1993a) |
| *ED* | Exposure duration (longevity) (yrs) | 50 | (Snow, 1973 in EPA 1993a) |
| *HR* | Foraging distance (km) | 10 | Territory length, mean, adults, coastal Washington State (EPA 1993a) |
| *TUF* | Temporal use factor | 1 | Will be 1 unless specific value exists for a receptor |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.12 | Adult, both sexes, Washington State, free-flying (EPA 1993a) |
| $F_P$ | Plant fraction | 0 | None listed as dietary intake in EPA (1993a) |
| $F_A$ | Animal fraction | 1 | 53 % birds, 27 % fish, 20 % other, Washington State, river (EPA 1993a) |
| *SFr* | Sediment fraction | 0 | Not reported in EPA 1993a; assumed to be negligible |
| $IR_W$ | Water ingestion rate (L/kgBW/d) | 0.036 | Average of adult male and female rates, estimated (EPA 1993a) by using allometric equation, $0.059 \times BW(kg)^{0.67}/BW$ (kg) (EPA 1993a, Eq. 3-15) |

[a] Food ingestion rate (g/g/d) re-expressed as kg/kg BW/d is assumed not to include ingested soil; therefore, $F_P + F_A = 1.0$.

**Mink (*Mustela vison*)**

The mink is the most abundant and widespread carnivorous mammal in North America. The home range of mink encompasses both their foraging areas around waterways and their dens along the Columbia River. The mink is found in aquatic habitats of all kinds, including waterways such as rivers, streams, lakes, and ditches, as well as swamps, marshes, and backwater areas (Linscombe et al. 1982 in EPA 1993a). Mink are particularly sensitive to certain chemicals. Mink are predominantly nocturnal hunters, although they are sometimes active during the day. They can often be found along the Columbia River. Mammals are the mink's most important prey year-round in many parts of their range (Eagle and Whitman 1987 in EPA 1993a), but mink also hunt aquatic prey (such as fish, amphibians, and crustaceans) and other terrestrial prey (such as birds, reptiles, and insects) depending on the season (Linscombe et al. 1982 in EPA 1993a). Salmon and trout can outmaneuver them, unless the fish are preoccupied with spawning (Eaton 2009).

| Parameter | Definition | Value | Reference/Notes |
|---|---|---|---|
| *BW* | Body weight (kg) | 0.85 | Average of adult male and female (summer and fall) (EPA 1993a) |
| *ED* | Exposure duration (longevity) (yrs) | 11 | Value for captive species (Enders, 1952 in EPA 1993a) |
| *HR* | Foraging distance (km) | 2.24 | Foraging distance, mean, adults, both sexes, Sweden/stream (EPA 1993a) |
| *TUF* | Temporal use factor | 1 | Will be 1 unless specific value exists for a receptor |
| $IR_F$ | Food ingestion rate (g/g/d = kg/kg BW/d)[a] | 0.14 | Michigan (farm raised) (EPA 1993a) |
| $F_P$ | Plant fraction | 0.09 | Michigan/stream, river (% wet wt; stomach contents normalized to 97.5% of contents identified) (EPA 1993a) |
| $F_A$ | Animal fraction | 0.91 | Michigan/stream, river (% wet wt; stomach contents normalized to 97.5% of contents identified) (EPA 1993a) |
| *SFr* | Sediment fraction | 0 | (Sample et al. 1997) |
| $IR_W$ | Water ingestion rate (L/kgBW/d) | 0.11 | Estimated (EPA 1993a) by using allometric equation, $0.099 \times BW(kg)^{0.90}/BW\ (kg)$ (EPA 1993a, Eq. 3-17) |

[a] Food ingestion rate (g/g/d) re-expressed as kg/kg BW/d is assumed not to include ingested soil; therefore, $F_P + F_A = 1.0$.

### 8.1.4 Assessment Endpoints

An assessment endpoint is defined by EPA (1997) to be "an expression of an ecological attribute that is to be protected." Environmental statutes govern the protection of ecological resources, including:

- Preservation and conservation of T&E organisms
- Maintenance and protection of terrestrial organism populations and ecosystems
- Maintenance and protection of aquatic organism populations and ecosystems

To fulfill these requirements, the assessment endpoints were chosen to:

- Protect and conserve individuals and populations of T&E species (Table 8-2, assessment endpoint 1).
- Maintain and protect terrestrial populations and ecosystems, including plants (Table 8-2, assessment endpoint 2), invertebrates (Table 8-2, assessment endpoint 3), herbivorous animals (Table 8-2, assessment endpoint 4), omnivorous animals (Table 8-2, assessment endpoint 5), and terrestrial predators (Table 8-2, assessment endpoint 6).
- Maintain and protect aquatic populations and ecosystems, including sediment-dwelling organisms (Table 8-2, assessment endpoint 7), planktivorous fish and small aquatic invertebrates (Table 8-2, assessment endpoint 8), waterfowl (Table 8-2, assessment endpoint 9), large carnivorous fish (Table 8-2, assessment endpoint 10), and fish-eating predators (Table 8-2, assessment endpoint 11).

The assessment endpoints reflect the conceptual exposure model and are based on the identified receptors and their recognized complete exposure pathways. Critical attributes of identified ecological receptors (population, community, or individual in the case of T&E species) are abundance and productivity, which are functions of survival and reproduction. Protection of receptors' survival and reproduction is assumed to protect the structure and function of the local ecosystem (EPA 1999). Measures of effect are defined as measures of change in critical attributes in response to a stressor to which receptors are exposed. For the Hanford Site risk assessment, modeled exposure concentrations and doses are compared to published concentrations and doses associated with measures of toxicological effect on the identified receptors or related species. Decision criteria prescribe how the endpoints are evaluated using the measures of effect.

Policy goals, assessment endpoints, measures of effect, and decision rules used for the SLERA are presented in Table 8-2.

## 8.2 Exposure Assessment

Estimation of the risk to ecological receptors from COPCs and ROPCs in environmental media at an exposure location requires an estimate of exposure and a toxicity reference value (*TRV*) (i.e., an exposure level associated with little or no adverse effect). Section 8.3 discusses *TRV*s. This section describes how the exposures of ecological receptors are estimated for environmental media at the WTP exposure locations. Exposure locations at the Hanford Site are areas within the deposition grid at which ecological receptors come into contact with COPCs and ROPCs in media contaminated by stack emissions. Contamination at a given location is represented by modeled concentrations of COPCs and ROPCs in environmental media. Receptor locations and emissions data used to compute EPCs are the same as in the human health risk assessment, but are limited to the onsite ground maximum (terrestrial receptors) and Columbia River maximum (aquatic receptors and all water consumption). This approach ensures that a conservative risk assessment results, since exposure at any other location would be lower. If there are no

unacceptable risks at the points of maximum deposition and air concentration, logically there cannot be unacceptable risks at other locations where COPC and ROPC concentrations are lower. Therefore, additional information about exposure at points with lower soil, air, or water concentrations will not be necessary.

The exposure assessments for ecological receptors estimate the exposure from ingestion of food and environmental media containing COPCs and ROPCs under certain assumptions. The ingestion rates of food and environmental media (soil, sediment, and water) and the proportions of different types of food that WTP receptors realistically ingest are given in Section 8.1 of this work plan. The proportions of different types of food that a receptor ingests (i.e., its diet) are an important factor in determining the exposure because different food types have different uptake rates of COPCs and ROPCs and, therefore, different concentrations in tissues. The diets to be used for the SLERA are defined in Section 8.2.1.

The assessment of exposure for ecological receptors requires estimates of the EPCs of COPCs and ROPCs in environmental media, including plants and animals ingested by receptors. Section 8.2 discusses EPCs. The SLERA will use modeled whole-body concentrations in food items to estimate doses to wildlife receptors. All terrestrial receptors are assumed to be exposed to the calculated concentrations of contaminants at the ground maximum or Columbia River maximum, regardless of the likelihood that they occur there.

The equations to be used to estimate exposure for terrestrial and aquatic receptors at the WTP exposure locations are described below (Sections 8.2.3 and 8.2.4). Two types of exposure estimates are required:

- The exposure estimate for receptors living immersed in a medium containing COPCs or ROPCs (such as vegetation and terrestrial invertebrates living in soil, fish and other aquatic life living in surface water, and benthic organisms living in sediment) is the concentration of COPC or ROPC in the medium.
- The exposure estimate for a wildlife receptor that does not live in a medium containing COPCs or ROPCs but is exposed by ingestion is the estimated daily dose (*DD*).

The exposure equations for wildlife are variations of wildlife exposure equations from EPA 1999 and implied in other sources (EPA 1997, 1998). These equations are used to calculate both the concentrations of COPCs and ROPCs in the tissues of receptors that are used for food (and in the case of ROPCs, the tissues of all other wildlife receptors) and the ingested doses of COPCs and ROPCs. The equations for ecological receptors are functionally equivalent to the equations in Section 7.1 of this work plan that are used to quantify exposure of humans by ingestion of contaminated food (EPA 2005). All ingested dose equations calculate the amount of contaminant ingested per unit biomass per unit time by multiplying the concentration of the contaminant in the ingested medium (abiotic medium or food item) by the receptor's ingestion rate for that medium and dividing by the receptor's body weight. The wildlife equations allow for the contaminant concentration in a food item to be calculated as the product of the contaminant concentration in an abiotic medium and the bioaccumulation (uptake or transfer) factor for the medium.

The modeled whole-body concentrations of contaminants in plants and fish consumed by both humans and nonhuman receptors will be calculated by using bioaccumulation factors, ingestion rates, and other parameters (Section 8.2.5) in model equations described by EPA (1999). The SLERA will use these modeled whole-body concentrations to estimate doses to wildlife receptors.

The diets to be used in the PRA and the FRA for WTP receptors are discussed in the following subsection.

### 8.2.1 Diet

The proportions of different types of food that a receptor eats (i.e., its diet) are important factors in determining the exposure because different food types have different concentrations of COPCs and ROPCs. Two general types of diet by which ingestion exposure of omnivores and carnivores can be estimated are discussed in this section. An *exclusive diet* is a diet consisting of a single type of prey or food, and a *realistic diet* is a diet where the fractions of different types of prey or food eaten are more or less the fractions reported to actually occur in one or more cases for the receptor or similar species. In the PRA and in the FRA, the exposure assessment will evaluate an exclusive diet in which the concentration of COPC or ROPC is calculated for each food item, and the higher concentration is used in the exposure evaluation. The exclusive-diet scenario will be evaluated as a worst-case scenario (i.e., it gives the most conservative risk estimate). If use of the exclusive diet results in an $HI > 0.25$ for an omnivore or one of its predators, exposure will be reevaluated using realistic diets subject to Ecology approval. In general, the fractions of prey or food types in a given animal's diet, the body burdens in each prey or food type, and the animal's bioconcentration factor (*BCF*) for the COPC or ROPC determines the animal's body burden and, thus, the exposure of its predator.

For 12 of the WTP receptors, a diet must be specified to quantify the dose of COPCs and ROPCs resulting from ingested food. Three of the receptors (mule deer, mourning dove, and Canada goose) eat only plants; four of the receptors (red-tailed hawk, spotted sandpiper, great blue heron, and bald eagle) eat only animals; the remaining five receptors (Great Basin pocket mouse, Western meadowlark, burrowing owl, coyote, and mink) typically eat a mixed diet of both plants and animals. However, the typical plant fraction for burrowing owl, coyote, and mink is so small that they will be evaluated as strict carnivores. An exclusive diet will be used for each of the omnivores and carnivores. Use of the realistic diet would reduce the ingestion exposure of mice and meadowlarks. It would also reduce the tissue concentrations in mice and meadowlarks and, thereby, reduce the ingestion exposure of the terrestrial carnivores—coyotes, owls, and hawks.

For the omnivores (pocket mouse and western meadowlark) and the top predators (coyote, burrowing owl, and red-tailed hawk), the SLERA will evaluate only the exclusive diet comprising the food type with the higher concentration for a given COPC. For the omnivores, if the plant food has the higher concentration for a given COPC, then the diet of 100 % plants will be evaluated (Figure 8-10), and vice versa should the food of the soil-dwelling invertebrate have higher tissue concentration. In this way, the exclusive diet will bound risk associated with insectivores as well as strict herbivores. For the top predators, if the small mammal prey (pocket mouse) has the highest concentration for a given COPC, then the diet of 100 % pocket mice will be evaluated (Figure 8-11), and vice versa should the western meadowlark have the higher tissue concentration. For mink, the SLERA will evaluate a diet of 100 % fish. This approach always results in the most conservative, highest exposure estimate for a given COPC for omnivores (pocket mouse and meadowlark) and predators (coyote, owl, hawk, and mink) that eat multiple types of food. For ROPCs, the assessment will evaluate only the exclusive diet of the food type resulting in the higher tissue concentration in the receptor.

If use of the exclusive diet results in an $HI > 0.25$ for an omnivore or one of its predators, exposure will be reevaluated using realistic diets subject to Ecology approval.

Concentrations used to estimate exposure for ecological receptors, exposure equations for terrestrial and aquatic receptors, and the variables and parameters used in these equations to estimate exposures for ecological receptors are provided in the following sections.

### 8.2.2 Exposure Point Concentrations in Abiotic Media

Exposure of ecological receptors to COPCs and ROPCs in this work plan will be estimated from the concentrations predicted by the aerial dispersion and other fate and transport models (Section 6). Dispersion model output concentrations will be used to calculate exposure concentrations for gases and particulates in air (μg/m$^3$, pCi/m$^3$) and surface soil (mg/kg, pCi/g) at the onsite ground maximum, and gases and particulates in air (μg/m$^3$, pCi/m$^3$), surface water (mg/L, pCi/L), and sediment (mg/kg, pCi/g) at the Columbia River maximum. For each of these exposure locations on the dispersion grid, the modeled concentration will be used to estimate the exposure to terrestrial (Section 8.2.3) and aquatic (Section 8.2.4) ecological receptors as appropriate. Use of maximum-modeled concentrations represents a conservative estimate of potential exposure due to the WTP operations.

In keeping with the protective approach that will be used in the SLERA, EPCs used to estimate doses of COPCs and ROPCs for the quantitative SLERA will correspond to the maximum concentrations at the locations of maximum deposition, and potential exposure to all ecological receptors will be evaluated there.

### 8.2.3 Quantification of Exposure (Terrestrial Receptors)

Quantifying exposures for receptors exposed by direct contact with air and soil, and ingestion of soil and biota, requires the EPCs of COPCs and ROPCs in air, soil, and biota. The method for calculating EPCs in air and soil is described in Section 8.2.2. The EPCs of COPCs and ROPCs in biota (Section 8.2.3.1) are required in order to calculate the *DD* by ingestion (Sections 8.2.3.2 and 8.2.3.3) and the internal radiation dose for wildlife receptors (Section 8.2.3.4).

Terrestrial receptors at Hanford can find water in many sources, including rain, snow, dew, and incidental surface sources. However, climate in the region results in greater evapotranspiration than precipitation (DOE 1997). Therefore, most potential water sources are ephemeral and are not appropriate for deposition modeling, which assumes a 40-year accumulation of COPCs and ROPCs. It is assumed for the RAWP that the terrestrial receptors ingesting surface water do so at the Columbia River maximum location. Exposure by ingestion of drinking water will also be evaluated for aquatic receptors at the Columbia River maximum location, where the river is also the source of drinking water for Canada goose, spotted sandpiper, great blue heron, bald eagle, and mink.

#### 8.2.3.1 EPCs in Terrestrial Biota

Calculating EPCs for tissues of terrestrial plants and animals exposed by direct contact with air and soil requires the EPCs of COPCs and the ROPCs in air ($C_a$) and soil (*Cs*) and the receptor bioaccumulation and uptake factors for the COPCs and ROPCs (Section 8.2.5.3). The remaining EPCs for receptors are computed using methodology from the SLERAP. Unless specifically stated otherwise, all tissue and body weights are wet or fresh weights (FW), whereas soil weights are dry weights (DW).

**EPCs in Terrestrial Plants (Trophic Level 1)**

The EPC for terrestrial plants ($C_{TP}$) exposed to COPCs and ROPCs in air and surface soil and fed upon by herbivores and omnivores is given by:

$$C_{TP} = Pd + Pv + Pr \qquad \text{(SLERAP Eq. 5-6)}$$

where:

| | | |
|---|---|---|
| $C_{TP}$ | = | concentration of COPC or ROPC in plant tissue (mg/kg or pCi/g) |
| $Pd$ | = | concentration resulting from uptake from particles deposited on leaf surfaces (mg/kg or pCi/g) |
| $Pv$ | = | concentration resulting from uptake of vapors by direct contact with air (mg/kg or pCi/g) |
| $Pr$ | = | concentration resulting from uptake from soil through roots (mg/kg or pCi/g) |

Equations for the calculation of *Pd* and *Pv* are presented in Sections 6.6.1 and 6.6.2, respectively. *Pr* is calculated as:

$$Pr = Cs_{15} \times BCF_r \times 0.12 \qquad \text{(SLERAP Table B-3-3)}$$

where:

| | | |
|---|---|---|
| $Pr$ | = | concentration resulting from uptake from soil through roots (mg/kg or pCi/g) |
| $Cs_{15}$ | = | concentration of constituent in soil (mg/kg or pCi/g), based upon a 15 cm root-zone soil depth |
| $BCF_r$ | = | plant-soil biotransfer factor (mg/kg DW plant per mg/kg DW soil) |
| 0.12 | = | dry-weight to wet-weight conversion (unitless, EPA 1999) |

The values of $BCF_r$ are discussed in Section 8.2.5.3. Values of $BCF_r$ for all COPCs and ROPCs are reported in Supplement 4.

**EPCs in Terrestrial Invertebrates (Trophic Level 2)**

For terrestrial invertebrates exposed to COPCs and ROPCs in air and surface soil and fed upon by omnivores, the tissue EPC will be calculated by using either a measured soil-to-invertebrate uptake factor or a calculated soil porewater concentration and an empirically determined water-to-invertebrate uptake factor. The EPCs for COPCs and ROPCs with measured uptake factors are calculated in accordance with EPA (1999) draft guidance:

$$C_{INV} = Cs_{15} \cdot BCF_S \qquad \text{(SLERAP Table F-1-3)}$$

where:

| | | |
|---|---|---|
| $C_{INV}$ | = | fresh weight concentration of COPC or ROPC in animal tissue (mg/kg or pCi/g) |

$Cs_{15}$ = concentration of COPC or ROPC in soil (mg/kg or pCi/g), based upon a 15 cm soil depth

$BCF_S$ = soil-to-invertebrate uptake factor (mg/kg FW tissue per mg/kg DW soil)

The values of $BCF_S$ are discussed in Section 8.2.5.3. Values of $BCF_S$ for all COPCs and ROPCs are reported in Supplement 4.

For many organic COPCs, measured $BCF_S$ values are not available. Instead, per EPA draft guidance (EPA 1999), values of $BCF_S$ for organic COPCs for which no measured values were available were calculated with an equation (SLERAP Eq. C-1-1) derived by regression analysis of uptake of several organic chemicals from water by aquatic invertebrates as a function of log $K_{ow}$ (Southworth et al., 1978 [see Section 8.2.5.3 for further discussion of the equation]). $K_{ow}$ is the ratio of the molar concentrations (in a dilute solution) of a chemical in n-octanol and in water. Since $K_{ow}$ is the ratio of two molar concentrations, it is a dimensionless quantity. Sometimes $K_{ow}$ is reported as the decadic logarithm (log $K_{ow}$). $K_{ow}$ provides a measure of chemical lipophilicity, that is, the degree to which a chemical dissolves in a lipid (an oily compound). The $K_{ow}$ values for affected organic COPCs are wide-ranging. Based on the equilibrium partitioning approach described in EPA (SLERAP Section 5.3.2.1), $BCF_W$ values for COPCs with higher log $K_{ow}$ values will be used with estimated soil porewater concentrations, rather than soil concentrations, to estimate COPC concentrations in terrestrial invertebrates.

$$\log BCF_W = 0.819 \cdot \log K_{ow} - 1.146 \qquad \text{(SLERAP Eq. C-1-4)}$$

where:

$BCF_W$ = water-to-tissue uptake factor for terrestrial invertebrates (mg/kg FW tissue per mg/L water)

$K_{ow}$ = octanol-water partition coefficient of the compound (unitless)

To be taken up by terrestrial invertebrates, chemicals must be in solution in soil porewater. For most organic COPCs, only a small fraction of the COPC in soil is dissolved in porewater, and the biologically available fraction of these organic COPCs in soil (i.e., the fraction in soil porewater) is small. Chemicals in soil porewater are assumed to be in equilibrium with chemicals bound to soil particles. The ratio of concentration in soil porewater to concentration on soil particles is given by the partitioning coefficient ($Kd_s$) that is characteristic of the chemical and the soil. However, most organic COPCs in soil are bound to organic carbon rather than to the mineral structure of soil particles (EPA 1993b), and $Kd_s$ is not constant for soils with different organic carbon contents. A more useful partitioning coefficient is the ratio of the concentration relative to soil carbon (mg/kg carbon) to the concentration in soil porewater (mg/L) and is designated $K_{oc}$. $K_{oc}$ can be multiplied by the fraction of organic carbon in the soil to derive the porewater-to-soil concentration ratio:

$$Kd_s = K_{oc} \cdot f_{oc} \qquad \text{(SLERAP Eq. A-2-8a)}$$

where:

$Kd_s$ = soil-water partitioning coefficient (L/kg soil)

$K_{oc}$ = soil organic carbon-water partitioning coefficient (L/kg carbon)

$f_{oc}$ = fraction of soil that is organic carbon, 0.0044 (fraction of organic carbon in soil, site-specific value from average organic carbon measurements in Paragon Analytics, Inc., CCN 150854) (kg carbon/kg soil)

The concentration in interstitial water ($C_{IW}$) can be calculated by dividing the concentration in the media of interest ($Cs_{15}$) by $Kd_s$:

$$C_{IW} = \frac{Cs_{15}}{Kd_s}$$

and by substitution (as shown in Eq. 5-5 of EPA draft guidance [EPA 1999]):

$$C_{IW} = \frac{Cs_{15}}{K_{oc} \cdot f_{oc}} \qquad \text{(SLERAP Eq. 5-5)}$$

where:

$C_{IW}$ = concentration of organic COPC in soil interstitial water (mg/L)

$Cs_{15}$ = concentration of organic COPC in soil (mg/kg soil), based upon a 15 cm soil depth

$K_{oc}$ = soil organic carbon-water partitioning coefficient (L/kg carbon)

$f_{oc}$ = fraction of soil that is organic carbon, 0.0044 (fraction of organic carbon in soil, site-specific value from average organic carbon measurements in Paragon Analytics, Inc., CCN 150854) (kg carbon/kg soil)

Thus, the tissue EPC for organic COPCs derived by using the calculated $BCF_W$ would be:

$$C_{INV} = C_{IW} . BCF_W \qquad \text{(SLERAP Eq. 5-4)}$$

and:

$$C_{INV} = \frac{Cs_{15}}{f_{oc} \cdot K_{oc}} \cdot BCF_W$$

where:

$C_{INV}$ = concentration of organic COPC in animal tissue (mg/kg)

$C_{IW}$ = concentration of organic COPC in soil porewater (mg/L)

$BCF_W$ = water-to-tissue uptake factor for terrestrial invertebrates (mg/kg FW tissue per mg/L water)

$Cs_{15}$ = concentration of organic COPC in 15 cm root-zone depth soil (mg/kg)

$f_{oc}$ = fraction of soil that is organic carbon, 0.0044 (fraction of organic carbon in soil, site-specific value from average organic carbon measurements in Paragon Analytics, Inc., CCN 150854) (kg carbon / kg soil)

$K_{oc}$ = soil organic carbon-water partitioning coefficient (L/kg) (Supplement 4)

The SLERAP (Section 5.3.2.1 of EPA 1999) quoted for the use of equilibrium partitioning to estimate porewater concentrations states that the equilibrium-partitioning approach may be applied only when certain conditions are met:

- The fraction of organic carbon in soil ($f_{oc}$) is known.
- The COPCs must be nonpolar hydrophobic organic compounds.
- The COPCs must have mathematically derived water-to-tissue BCFs.

For this work plan, equilibrium partitioning can be applied to the subset of organic COPCs that have log $K_{ow}$ and log $K_{oc}$ values but do not have measured *BCF* values because each the above conditions are met, as described below:

- The approach is considered valid if $f_{oc}$ is > 0.002 (EPA 1993b), whereas it has been accepted that the average of measured $f_{oc}$ values is 0.0044 for the SLRA. This $f_{oc}$ value is based on the sample data provided by Ecology (CCN 150854). This data indicates organic carbon content ranges from 0.21 to 0.77 percent (10 samples plus a duplicate sample, mean = 0.0044, standard deviation = 0.0022). The mean value will be used to model the soil invertebrate tissue concentration for the subset of organic COPCs mentioned in the text.
- The hydrophobic nature of a compound is indicated by its log $K_{ow}$. In the discussion of the technical basis for using equilibrium partitioning to derive sediment quality criteria (EPA 1993b), EPA shows sediment quality criteria for compounds with log $K_{ow}$ above about 2.6, so any compound with a log $K_{ow}$ greater than or equal to 2.6 should be considered sufficiently hydrophobic to meet the requirements of the method. The organic COPCs with $BCF_S$ values calculated by SLERAP Eq. C-1-1 (EPA 1999) that also have log $K_{ow}$ values greater than or equal to 2.6 and thus meet the requirement of being nonpolar, hydrophobic compounds with mathematically derived water *BCF*s are footnoted in Supplement 4.
- The organic COPCs for which $K_{ow}$ is known have mathematically determined water-to-invertebrate *BCF*s.

The equilibrium-partitioning approach will be used for the organic COPCs that do not have measured $BCF_S$ values but have log $K_{ow}$ values ≥ 2.6. For the remaining organic COPCs that do not have measured $BCF_S$ values but have log $K_{ow}$ values < 2.6, the calculated $BCF_W$ will be used to calculate tissue concentrations per SLERAP Table F-1-3. Per EPA (1999) it will be assumed that the pore water concentration is the same as the concentration in bulk soil. Therefore, the calculated $BCF_S$ will be assumed to have units of kg soil/kg tissue.

**EPCs in Mammal and Bird Tissues (Trophic Levels 2, 3, and 4)**

For mammal and bird omnivores that are preyed upon by other predator receptors, the tissue EPC ($C_{OM}$) will be calculated as the sum of the contributions from the different types of material ingested. For transfer of COPCs and ROPCs to receptors by ingestion of plants, water, and soil, *BCF*s are used. For transfer of COPCs and ROPCs from prey to predators by ingestion of prey tissue, the food-chain multiplier (*FCM*) approach (EPA 1999) will be used to model transfer from one trophic level to another. Section 8.2.5.3 discusses *FCM*s. It is assumed that all mammals and birds ingest unfiltered water from the Columbia River maximum location. The equation describing the concentration of COPCs and ROPCs in receptor tissues is adapted from the SLERAP (EPA 1999, Eqs. 5-11, 5-12, and 5-13). The equation has

been modified by simplification of the subscripts and removal of the summation (since exclusive diets are assumed). The equation takes the following form:

| concentration in receptor | = | contaminants consumed from prey | + | contaminants consumed from plants | + | contaminants consumed from soil | + | contaminants consumed from water |
|---|---|---|---|---|---|---|---|---|

where:

| contaminants consumed | = | concentration in food or media | × | bioconcentration factor | × | fraction or proportion of food or media that is contaminated |
|---|---|---|---|---|---|---|

such that the concentration in a omnivore is: (modified SLERAP Eq. 5-12)

$$C_{OM} = C_A \cdot \frac{FCM_{OM}}{FCM_A} \cdot F_A + C_{TP} \cdot BCF_P \cdot F_P + Cs_2 \cdot BCF_S \cdot P_S + C_{wctot} \cdot BCF_W \cdot P_W \cdot CF$$

and the concentration in a herbivore is: (modified SLERAP Eq. 5-11)

$$C_H = C_{TP} \cdot BCF_P \cdot F_P + Cs_2 \cdot BCF_S \cdot P_S + C_{wctot} \cdot BCF_W \cdot P_W \cdot CF$$

and the concentration in a carnivore is: (modified SLERAP Eq. 5-13)

$$C_C = C_A \cdot \frac{FCM_C}{FCM_A} \cdot F_A + Cs_2 \cdot BCF_S \cdot P_S + C_{wctot} \cdot BCF_W \cdot P_W \cdot CF$$

where:

| | | |
|---|---|---|
| $C_{OM/H/C}$ | = | concentration of constituent in omnivorous, herbivorous, or carnivorous (respectively) receptor tissue (mg/kg FW tissue or pCi/g FW tissue) |
| $C_A$ | = | concentration of constituent in ingested animal prey (mg/kg FW tissue or pCi/g FW tissue) |
| $FCM_{OM/C}$ | = | food-chain multipliers for the omnivorous receptor (unitless) |
| $FCM_A$ | = | food-chain multipliers for ingested prey type (unitless) |
| $F_A$ | = | fraction of diet from animal tissue (unitless) |
| $C_{TP}$ | = | concentration of constituent in ingested plant tissue (mg/kg plant or pCi/g plant) |
| $BCF_P$ | = | plant-to-tissue uptake factor for omnivorous/herbivorous receptor (mg/kg FW tissue per mg/kg plant) |
| $F_P$ | = | fraction of diet from plant tissue (unitless) |
| $Cs_2$ | = | concentration of constituent in ingested soil (based upon an untilled 2-cm soil depth) (mg/kg or pCi/g), |
| $BCF_S$ | = | soil-to-tissue uptake factor for omnivorous/herbivorous/carnivorous receptor (mg/kg FW tissue per mg/kg DW soil) |

$P_S$ = proportion of consumed soil or sediment that is contaminated (unitless)

$C_{wctot}$ = concentration of constituent in unfiltered water from Columbia River maximum location (mg/L or pCi/L)

$BCF_W$ = water-to-tissue uptake factor for omnivorous/herbivorous/carnivorous receptor (mg/kg FW tissue per mg/L water)

$P_W$ = proportion of consumed water that is contaminated (unitless)

$CF$ = conversion factor for radionuclides, 0.001 pCi/kg to pCi/g

Per the SLERAP (EPA 1999), the plant-to-tissue, water-to-tissue, and soil-to-tissue *BCF*s are calculated from the receptor's ingestion rate and the published biotransfer factor (*Ba*). The *BCF*s are calculated using a modified version of SLERAP equations. The modification is necessary to derive *BCF*s from ingestion rates reported on a body-weight basis:

$$BCF_F = Ba \cdot IR_F \cdot BW \qquad \text{(modified SLERAP Eq. D-1-1)}$$

and:

$$BCF_M = Ba \cdot IR_M \cdot BW \qquad \text{(modified SLERAP Eq. D-1-2)}$$

where:

$BCF_A$ = food-to-animal bioconcentration factor for receptor (mg COPC/kg FW tissue per mg COPC/kg FW food)

$BCF_M$ = media-to-animal bioconcentration factor for receptor (mg COPC/kg FW tissue per mg COPC/kg DW media)

$Ba$ = ingestion-to-tissue transfer factor (d/kg)

$IR_F$ = daily food ingestion rate (kg/kg BW/d)

$IR_M$ = daily media ingestion rate, such that:

- $IR_w$ = rate of water consumption (L/kgBW/d)
- $IR_S$ = $SFr \cdot IR_F$ = rate of soil consumption (kg/kg BW/d)
- $SFr$ = soil ingested per unit food ingested (unitless)

$BW$ = body weight of receptor (kg)

Soil consumption by receptors is incidental to the consumption of prey and plants. The amount of soil ingested per unit of food ingested (*SFr*) is used in conjunction with the food ingestion rate to determine the soil ingestion rate ($IR_S = IR_F \times SFr$).

According to EPA (1999), the fraction of the diet that is plants ($F_P$) is included in the calculation of $BCF_P$. Thus, an omnivore whose diet is 50 % plants would have a $BCF_P$ half that of an herbivore with the same body weight and food ingestion rate. However, because $F_P$ must be adjusted to either 1 or 0 for the exclusive diet, a fixed value of $F_P$ cannot be included in the calculation of $BCF_P$. Therefore, $F_P$ is not included as a part of $BCF_P$, but appears as a separate term in SLERAP Equation 5-12.

Values for $IR_F$, $F_P$, *BW*, *SFr*, and water ingestion ($IR_w$) for receptors exposed at terrestrial areas are given in the receptor profiles in Section 8.1.3.3. The values of *BCF* are discussed in Section 8.2.5.3. Values of

*Ba*, $BCF_P$ (*BCF* for plants), $BCF_S$ (*BCF* for soil), and $BCF_W$ (*BCF* for water), for all COPCs and ROPCs for each receptor are reported in Supplement 4.

The EPCs for COPCs in plants, terrestrial invertebrates, and mammal and bird receptors that are eaten by other receptors will be used in the equations for modeling intake to terrestrial ecological receptors (i.e., the ingestion *DD*). Tissue EPCs for ROPCs are used for all receptors to calculate internal radiation exposure.

#### 8.2.3.2 Modeling Intake to Terrestrial Ecological Receptors

The ingestion *DD* for terrestrial receptors will be calculated as the sum of the intakes of plant tissue, animal tissue, soil, and water. Thus:

$$DD = \sum IR_F \cdot C_i \cdot P_i \cdot F_i + \sum IR_M \cdot C_M \cdot P_M \qquad \text{(SLERAP Eq. 5-1)}$$

or:

$$DD = DD_A + DD_P + DD_S + DD_W$$

where:

| | | |
|---|---|---|
| $DD$ | = | daily dose by ingestion (mg/kg BW/d) |
| $IR_F$ | = | receptor plant or animal food item ingestion rate (kg/kg BW-day) |
| $C_i$ | = | constituent concentration in $i^{th}$ plant or animal food item (mg/kg) |
| $P_i$ | = | proportion of $i^{th}$ food item that is contaminated (unitless) - assumed to be equal to 1 |
| $F_i$ | = | fraction of diet consisting of plant or animal food item *i* (unitless) |
| $IR_M$ | = | media *M* ingestion rate (kg/kg BW-day [soil or bed sediment] or L/kg BW-day [water]) |
| $C_M$ | = | constituent concentration in media *M* (mg/kg [soil or bed sediment] or mg/L [water]) |
| $P_M$ | = | proportion of ingested media *M* that is contaminated (unitless) |
| $DD_A$ | = | daily dose by animal ingestion (mg/kg BW/d) |
| $DD_P$ | = | daily dose by plant ingestion (mg/kg BW/d) |
| $DD_S$ | = | daily dose by soil or sediment ingestion (mg/kg BW/d) |
| $DD_W$ | = | daily dose by water ingestion (mg/kg BW/d) |

As defined by Equation 5-1 of the SLERAP:

$$DD_A = C_A \cdot IR_F \cdot F_A$$

$$DD_P = C_{TP} \cdot IR_F \cdot F_P$$

$$DD_S = Cs_2 \cdot IR_F \cdot SFr$$

$$DD_W = C_{wctot} \cdot IR_W$$

where:

| | | |
|---|---|---|
| $DD_A$ | = | daily dose by animal ingestion (mg/kg BW/d) |
| $DD_P$ | = | daily dose by plant ingestion (mg/kg BW/d) |
| $DD_S$ | = | daily dose by soil ingestion (mg/kg BW/d) |
| $DD_w$ | = | daily dose by water ingestion (mg/kg BW/d) |
| $C_A$ | = | concentration of constituent in ingested animal tissue (mg/kg) |
| $C_P$ | = | concentration of constituent in ingested plant tissue (mg/kg) |
| $IR_F$ | = | food (plant or prey, as applicable) ingestion rate of receptor (kg/kg BW/d) |
| $F_A$ | = | fraction of diet from animals (unitless) |
| $F_P$ | = | fraction of diet from plants (unitless) |
| $Cs_2$ | = | concentration of constituent in ingested soil (mg/kg ), based upon a 2-cm untilled soil depth |
| $SFr$ | = | soil ingested per unit of food ingested (unitless) |

The plant and animal food fractions sum to 1 ($F_P + F_A = 1$), and $SFr$ is defined as the amount of soil ingested per unit of food ingested. Therefore, the total ingested fraction of food plus soil ($F_P + F_A + SFr$) is greater than 1 (e.g., for the western meadowlark $F_P = 0.3$, $F_A = 0.7$, and $SFr = 0.29$, so the total ingested fraction is 1.29).

Proportion of contaminated food and media ($P_i$ and $P_M$), absorption efficiency ($AE$), the area use factor ($AUF$), and the temporal use factor ($TUF$) are assumed to be equal to 1, so they do not appear in the exposure equations.

#### 8.2.3.3 Receptor-Specific Exposure Equations for Terrestrial Receptors

The complete equations for daily ingestion intake ($DD$) and animal tissue concentration ($C_A$) for each receptor are presented below.

**Herbivores: Mule Deer and Mourning Dove (Trophic Level 2)**

Mule deer and mourning doves are strict herbivores but ingest soil incidentally with their plant food and are assumed to ingest water from the Columbia River (Figure 8-8). Thus,

$$DD = DD_P + DD_S + DD_W, \text{ or} \qquad \text{(Equation 8-1)}$$

$$DD_{Deer} = C_{TP} \cdot IR_F \cdot F_P + Cs_2 \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W$$

$$DD_{Dove} = C_{TP} \cdot IR_F \cdot F_P + Cs_2 \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W$$

where $C_{TP}$, $IR_F$, $Cs_2$, $SFr$, $C_{wctot}$, and $IR_w$ are as given above. The mule deer and mourning dove food ingestion rates ($IR_F$), dietary fractions ($F_P$ and $SFr$), and water ingestion rates ($IR_w$) are given in the receptor profiles in Section 8.1.3.3.

Deer and dove tissue concentrations are calculated by an equation with the applicable exposure routes in SLERAP Equation 5-11:

$$C_{Deer} = C_{TP} \cdot BCF_P \cdot F_P + Cs_2 \cdot BCF_S \cdot P_S + C_{wctot} \cdot BCF_W \cdot P_W \cdot CF$$

$$C_{Dove} = C_{TP} \cdot BCF_P \cdot F_P + Cs_2 \cdot BCF_S \cdot P_S + C_{wctot} \cdot BCF_W \cdot P_W \cdot CF$$

where:

| | | |
|---|---|---|
| $C_{Deer/Dove}$ | = | concentration of COPC or ROPC in receptor tissue (mg/kg or pCi/g) |
| $C_{TP}$ | = | concentration of COPC or ROPC in ingested plant tissue (mg/kg or pCi/g) |
| $F_P$ | = | fraction of diet from plants (unitless) |
| $BCF_P$ | = | plant-to-tissue uptake factor for receptor (mg/kg FW tissue per mg/kg plant) |
| $Cs_2$ | = | concentration of constituent in ingested soil (mg/kg or pCi/g), based upon a 2-cm untilled soil depth |
| $BCF_S$ | = | soil-to-tissue uptake factor for receptor (mg/kg FW tissue per mg/kg DW soil) |
| $C_{wctot}$ | = | total concentration of ROPC in water (mg/kg or pCi/L) |
| $BCF_W$ | = | water-to-tissue uptake factor for receptor (kg FW tissue per mg/L water) |
| $CF$ | = | conversion factor for radionuclides, 0.001 pCi/kg to pCi/g |

The soil-to-tissue concentration factors ($BCF_S$), plant-to-tissue concentration factors ($BCF_P$), and water-to-tissue uptake factors ($BCF_W$), respectively, for mule deer and mourning doves are reported in Supplement 4.

**Omnivores: Great Basin Pocket Mouse and Western Meadowlark (Trophic Level 3)**

Great Basin pocket mouse and western meadowlark are omnivores that ingest plants and invertebrates, and ingest soil incidentally with their food. The western meadowlark is assumed to ingest water from the Columbia River whereas the Great Basin pocket mouse is assumed to obtain water through its food (Figure 8-8). The receptor dose includes the contribution of food, soil and water (for the meadowlark):

$$DD = DD_A + DD_P + DD_S + DD_W, \text{ or} \qquad \text{(Equation 8-2)}$$

$$DD = C_A \cdot IR_F \cdot F_A + C_{TP} \cdot IR_F \cdot F_P + Cs_2 \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W$$

where $C_A$, $C_{TP}$, $IR_F$, $Cs_2$, $SFr$, $C_{wctot}$, and $IR_w$ are as given above (Equation 8-2 first appears in Section 8.2.3.2). The pocket mouse and Western meadowlark food ingestion rates ($IR_F$), water ingestion rates ($IR_w$), and dietary fractions ($F_A$, $F_P$) and soil fraction ($SFr$), are given in the receptor profiles in Section 8.1.3.3. The SLERA will evaluate the exposure of mouse and meadowlark assuming ingestion of only the food type with the highest tissue concentration. Thus, the concentration of each COPC and each ROPC will be calculated for plants and terrestrial invertebrates, and the higher concentration will be used in the exposure calculation. The terrestrial food web (Figure 8-8) shows that the sole animal prey type for the Great Basin pocket mouse and western meadowlark are terrestrial invertebrates. Whether plants or terrestrial invertebrates have the higher tissue concentration is a function of the soil-to-tissue uptake factor for the two food types. For the exclusive diets, if the herbivore diet for a given constituent is the main source of tissue contamination, $F_P$ is one and $F_A$ is zero ($C_P > C_A$). If the carnivore diet for a given

constituent is the main source of tissue contamination, $F_P$ is zero and $F_A$ is one ($C_A > C_P$) (Figure 8-10). Selection of the exclusive diet is made on a constituent-by-constituent basis. The use of the exclusive diet in the evaluation of the worst-case scenario is discussed in Section 8.2.1. The corresponding dose equations are therefore:

for plant consumption (herbivore diet, $C_P > C_A$):

$$DD_{Mouse} = C_{TP} \cdot IR_F \cdot F_P + Cs_2 \cdot IR_F \cdot SFr \qquad \text{(Equation 8-3)}$$

$$DD_{Lark} = C_{TP} \cdot IR_F \cdot F_P + Cs_2 \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W$$

and for consumption of invertebrates (carnivore diet, $C_A > C_P$):

$$DD_{Mouse} = C_{INV} \cdot IR_F \cdot F_A + Cs_2 \cdot IR_F \cdot SFr \qquad \text{(Equation 8-4)}$$

$$DD_{Lark} = C_{INV} \cdot IR_F \cdot F_A + Cs_2 \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W$$

Great Basin pocket mouse and western meadowlark tissue concentrations are calculated by equations adapted from the SLERAP Equations 5-12 and 5-13:

for plant consumption (herbivore diet, $C_P > C_A$):

$$C_{Mouse} = C_{TP} \cdot BCF_P \cdot F_P + Cs_2 \cdot BCF_S$$

$$C_{Lark} = C_{TP} \cdot BCF_P \cdot F_P + Cs_2 \cdot BCF_S + C_{wctot} \cdot BCF_W \cdot CF$$

and for consumption of invertebrates (carnivore diet, $C_A > C_P$):

$$C_{Mouse} = C_{INV} \cdot \frac{FCM_3}{FCM_2} \cdot F_A + Cs_2 \cdot BCF_S$$

$$C_{Lark} = C_{INV} \cdot \frac{FCM_3}{FCM_2} \cdot F_A + Cs_2 \cdot BCF_S + C_{wctot} \cdot BCF_W \cdot CF$$

where:

$C_{Mouse/Lark}$ = concentration of constituent in receptor tissue (mg/kg or pCi/g)
$C_{TP}$ = concentration of constituent in ingested plant tissue (mg/kg or pCi/g)
$BCF_P$ = plant-to-tissue uptake factor for receptor (mg/kg FW tissue per mg/kg plant)
$F_P$ = fraction of diet from plants (unitless)
$C_{INV}$ = concentration of constituent in ingested invertebrate (mg/kg or pCi/g)
$FCM_3$ = food-chain multipliers for the receptor (unitless)
$FCM_2$ = food-chain multipliers for ingested prey type (unitless)
$F_A$ = fraction of diet from animals (unitless)

| | | |
|---|---|---|
| $Cs_2$ | = | concentration of COPC or ROPC in ingested soil (mg/kg or pCi/g), based upon a 2-cm untilled soil depth |
| $BCF_S$ | = | soil-to-tissue bioconcentration factor for receptor (mg/kg FW tissue per mg/kg DW soil) |
| $C_{wctot}$ | = | total concentration of COPC (mg/L) or ROPC (pCi/L) in water |
| $BCF_W$ | = | water-to-tissue bioconcentration factor (kg FW tissue per mg/L water) |
| $CF$ | = | conversion factor, 0.001 pCi/kg to pCi/g |

The FCMs for the pocket mouse and western meadowlark ($FCM_3$) and their prey ($FCM_2$) are reported in Supplement 4. The soil-to-tissue bioconcentration factors ($BCF_S$), plant-to-tissue bioconcentration factors ($BCF_P$), and water-to-tissue bioconcentration factor ($BCF_W$) are reported in Supplement 4.

**Carnivores: Coyote and Burrowing Owl (Trophic Level 4)**

Coyotes and burrowing owls are carnivores that ingest primarily small animals, but also a small fraction of soil incidentally with their food and are assumed to ingest water from the Columbia River (Figure 8-8). The SLERA will evaluate the exposure of coyotes and burrowing owls as carnivores, assuming ingestion of only the animal prey type with the highest tissue concentration. Figure 8-11 shows how the exposure of carnivores is calculated using existing diet for the case where soil invertebrates have a higher estimated tissue concentration than plants. Whether meadowlarks or pocket mice have the highest tissue concentration is a function of the soil-to-tissue and other uptake factors for those prey types. Thus:

$DD = DD_A + DD_S + DD_W$, or (Equation 8-1)

$$DD = C_A \cdot IR_F \cdot F_A + Cs_2 \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W$$

where $C_A$, $IR_F$, $Cs_2$, $SFr$, $C_{wctot}$, and $IR_w$ are as given above. $DD_A$ is calculated for the prey type with the highest expected body burden for a given constituent. The value of $F_A$ ($F_{Mouse}$ and $F_{Lark}$) is the value shown in the appropriate table entry in Section 8.1.3.3 for the exclusive diet, or zero for the non-exclusive diet. Because of the exclusive diet assumption, the prey that has the greatest contribution to the accumulation of a given contaminant in the receptor tissue is the sole source of that contaminant. Accordingly, the prey that has the least contribution of a given contaminant is not considered ($F_A$ is set to zero). For example, if the coyote's uptake for constituent "X" is greatest from the mouse, and for constituent "Y" is greatest from the lark, it is assumed that the coyote will be exposed to "X" solely from preying upon mice, and the contribution of constituent "Y" to the coyote's dose will come from exclusive consumption of lark. The corresponding dose equations are therefore:

For a diet exclusive to Great Basin pocket mouse consumption ($C_{Mouse} > C_{Lark}$):

$$DD_{Coyote} = C_{Mouse} \cdot IR_F \cdot F_{Mouse} + Cs_2 \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W \quad \text{(Equation 8-5)}$$

$$DD_{Owl} = C_{Mouse} \cdot IR_F \cdot F_{Mouse} + Cs_2 \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W$$

For a diet exclusive to western meadowlark consumption ($C_{Lark} > C_{Mouse}$):

$$DD_{Coyote} = C_{Lark} \cdot IR_F \cdot F_{Lark} + Cs_2 \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W \qquad \text{(Equation 8-6)}$$

$$DD_{Owl} = C_{Lark} \cdot IR_F \cdot F_{Lark} + Cs_2 \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W$$

Coyote and burrowing owl food ingestion rate ($IR_F$), dietary fraction ($F_A$ and $SFr$), and water ingestion rate ($IR_w$) are given in the receptor profiles in Section 8.1.3.3. The terrestrial food web (Figure 8-8) shows that the sole prey types of the coyote and burrowing owl to be evaluated in the SLERA are the Great Basin pocket mouse and the western meadowlark.

Coyote and burrowing owl tissue concentrations of will be calculated by an equation adapted from the SLERAP Equation 5-13:

for a diet exclusive to Great Basin pocket mouse consumption ($C_{Mouse} > C_{Lark}$):

$$C_{Coyote} = C_{Mouse} \cdot \frac{FCM_4}{FCM_3} \cdot F_{Mouse} + Cs_2 \cdot BCF_S + C_{wctot} \cdot BCF_W \cdot CF$$

$$C_{Owl} = C_{Mouse} \cdot \frac{FCM_4}{FCM_3} \cdot F_{Mouse} + Cs_2 \cdot BCF_S + C_{wctot} \cdot BCF_W \cdot CF$$

for a diet exclusive to western meadowlark consumption ($C_{Lark} > C_{Mouse}$):

$$C_{Coyote} = C_{Lark} \cdot \frac{FCM_4}{FCM_3} \cdot F_{Lark} + Cs_2 \cdot BCF_S + C_{wctot} \cdot BCF_W \cdot CF$$

$$C_{Owl} = C_{Lark} \cdot \frac{FCM_4}{FCM_3} \cdot F_{Lark} + Cs_2 \cdot BCF_S + C_{wctot} \cdot BCF_W \cdot CF$$

where:

| | | |
|---|---|---|
| $C_{Coyote/Owl}$ | = | concentration of in receptor tissue (mg/kg or pCi/g) |
| $C_{Mouse/Lark}$ | = | concentration of in prey tissue (mg/kg or pCi/g) |
| $FCM_4$ | = | food-chain multipliers for the receptor (unitless) |
| $FCM_3$ | = | food-chain multipliers for ingested prey type (unitless) |
| $F_{Mouse}$ | = | fraction of diet from Great Basin pocket mouse tissue (unitless) |
| $F_{Lark}$ | = | fraction of diet from western meadowlark tissue (unitless) |
| $Cs_2$ | = | concentration of constituent in ingested soil (mg/kg or pCi/g), based upon a 2-cm untilled soil depth |
| $BCF_S$ | = | soil-to-tissue uptake factor for receptor (mg/kg FW tissue per mg/kg DW soil) |
| $C_{wctot}$ | = | total concentration of constituent in water (mg/L or pCi/L) |
| $BCF_W$ | = | water-to-tissue uptake factor for receptor (kg FW tissue per mg/L water) |
| $CF$ | = | conversion factor, 0.001 pCi/kg to pCi/g |

For the exclusive diet, $F_A = 1$ for the prey type contributing the most to receptor tissue uptake, and zero for the receptor contributing the least. The *FCM*s for the coyote and burrowing owl ($FCM_{Coyote/Owl}$) and their prey ($FCM_{Mouse/Lark}$) are reported in Supplement 4. Soil-to-tissue uptakes factors ($BCF_S$) and water-to-tissue uptake factors ($BCF_W$) for the coyote and burrowing owls are reported in Supplement 4.

**Carnivore: Red-tailed hawk (Trophic Level 4)**

Red-tailed hawks are carnivores that ingest small animals but do not ingest soil incidentally with their food. They are assumed to ingest water from the Columbia River (Figure 8-8). The SLERA will evaluate the exposure of red-tailed hawks assuming ingestion of only the prey type with the highest tissue concentration (Figure 8-11). Thus, the concentration of each COPC and ROPC in mice and meadowlarks will be calculated, and the higher concentration will be used in the exposure evaluation. Whether meadowlarks or Great Basin pocket mice have the higher tissue concentration is a function of the soil-to-tissue and other uptake factors for those prey types. Thus,

$$DD = DD_A + DD_W, \text{ or} \qquad \text{(Equation 8-7)}$$

$$DD = C_A \cdot IR_F \cdot F_A + C_{wctot} \cdot IR_W$$

where $DD_A$ is calculated for prey type with the highest expected body burden for a given constituent with $F_A = 1$. $DD_W$ is as given above. The corresponding dose equations are therefore:

for a diet exclusive to Great Basin pocket mouse consumption ($C_{Mouse} > C_{Lark}$):

$$DD_{Hawk} = C_{Mouse} \cdot IR_F \cdot F_{Mouse} + C_{wctot} \cdot IR_W \qquad \text{(Equation 8-8)}$$

for a diet exclusive to western meadowlark consumption ($C_{Lark} > C_{Mouse}$):

$$DD_{Hawk} = C_{Lark} \cdot IR_F \cdot F_{Lark} + C_{wctot} \cdot IR_W \qquad \text{(Equation 8-9)}$$

Red-tailed hawk food ingestion rate ($IR_F$) and water ingestion rate ($IR_w$) are given in the receptor profiles in Section 8.1.3.3. The terrestrial food web (Figure 8-8) shows the prey types for the hawk. The hawk prey types to be evaluated in the SLERA are the Great Basin pocket mouse and the western meadowlark.

Red-tailed hawk tissue concentrations of ROPCs are calculated by an equation adapted from the SLERAP Equation 5-13:

for a diet exclusive to Great Basin pocket mouse consumption ($C_{Mouse} > C_{Lark}$):

$$C_{Hawk} = C_{Mouse} \cdot \frac{FCM_4}{FCM_3} \cdot F_{Mouse} + C_{wctot} \cdot BCF_W \cdot CF$$

for a diet exclusive to western meadowlark consumption ($C_{Lark} > C_{Mouse}$):

$$C_{Hawk} = C_{Lark} \cdot \frac{FCM_4}{FCM_3} \cdot F_{Lark} + C_{wctot} \cdot BCF_W \cdot CF$$

where:

$C_{Hawk}$ = concentration of in receptor tissue (mg/kg or pCi/g)
$C_{Mouse/Lark}$ = concentration of in prey tissue (mg/kg or pCi/g)
$FCM_4$ = food-chain multipliers for the receptor (unitless)
$FCM_3$ = food-chain multipliers for ingested prey type (unitless)
$F_{Mouse}$ = fraction of diet from Great Basin pocket mouse tissue (unitless)
$F_{Lark}$ = fraction of diet from western meadowlark tissue (unitless)
$C_{wctot}$ = total concentration of constituent in water (mg/L or pCi/L)
$BCF_W$ = water-to-tissue uptake factor for receptor (kg FW tissue per mg/L water)
$CF$ = conversion factor of 0.001 kg/g (to convert pCi/kg to pCi/g) for ROPCs

For the exclusive diet, $F_A = 1$ for the prey type contributing the most to receptor tissue uptake, and zero for the receptor contributing the least. The FCMs for hawks ($FCM_4$) and their prey ($FCM_3$) are reported in Supplement 4 along with water-to-tissue uptake factors ($BCF_W$).

#### 8.2.3.4 External and Internal Radiation Dose

The total radiological dose to all receptors is calculated as the sum of the external and internal radiation doses for all ROPCs, using methods presented by Sample et al. (1997). External doses to all receptors result from exposure to ROPCs in soil and air. The internal dose to plants and terrestrial invertebrates results from the uptake of radionuclides into their tissues from soil. The internal dose to wildlife receptors results from the uptake of radionuclides into their tissues from ingested food, soil, and water.

The total radiological dose is calculated as:

$$DD = DD_E + DD_I \qquad \text{(Equation 8-10)}$$

where:

$DD$ = total radiation dose to the receptor (rad/day)
$DD_E$ = total external radiation dose (rad/day)
$DD_I$ = internal radiation dose (rad/day)

All radiation damage results from interaction of ionizing radiation with molecules in the tissues. As each ROPC decays, it emits radiation that is characteristic for that ROPC. The energy absorbed by tissues depends on the type and energy of radiation and the amount of tissue that absorbs the energy. Thus, alpha particles and most beta radiation do not penetrate the skin and do not cause damage by external radiation. Also, the fraction of gamma radiation from any ROPC that is absorbed by tissue is higher for large animals than for small animals. Internal alpha radiation does more damage to tissues per unit of energy. To adjust for the additional damage, a quality factor ($QF$) is used: the alpha energy is multiplied by $QF$ in the exposure equations. In a paper by Kocher and Trabalka (2000) it is indicated a quality factor of 5 was suggested by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR 1996), but states that the rationale for the value was not substantiated. Kocher and Trabalka (2000) state that the quality factor probably lies between 5 and 10. Based on this paper, the RAWP will use a quality

factor of 10 (upper end of probable range) for alpha energy. The quality factors for beta and gamma radiation are 1.

**External Dose**

External radiation doses from air and soil will be calculated by methods presented by Eckerman and Ryman (1993, same as EPA 1993c), Blaylock, Frank, and O'Neal (1993), and Sample et al. (1997) because the SLERAP (EPA 1999) does not provide methods to evaluate radiation doses. Sample et al. (1997) is a published report from Oak Ridge National Laboratory and not agency guidance. The method of Sample et al. (1997) to calculate belowground external radiation to terrestrial invertebrates will be adapted for use to calculate external doses to belowground portions of plants. Exposures of terrestrial receptors to external radiation will be calculated as follows:

- **Terrestrial plants** – aboveground parts by immersion in air and contact with the soil surface and belowground parts by immersion in soil
- **Terrestrial invertebrates** – immersion in air and contact with the soil surface while aboveground and immersion in soil while belowground
- **Mule deer** – immersion in air and radiation from the soil surface
- **Mourning dove** – immersion in air and radiation from the soil surface
- **Great Basin pocket mouse** – immersion in air and radiation from the soil surface while aboveground and immersion in soil while belowground
- **Western meadowlark** – immersion in air and radiation from the soil surface
- **Coyote** – immersion in air and radiation from the soil surface while aboveground and immersion in soil while belowground
- **Burrowing owl** – immersion in air and radiation from the soil surface while aboveground and immersion in soil while belowground
- **Red-tailed hawk** – immersion in air and radiation from the soil surface

External irradiation by immersion in air containing ROPCs and by standing, sitting, or lying on the soil surface (aboveground radiation) will be modeled by using external dose conversion factors (*DCF*s) (Eckerman and Ryman 1993) and the activity of ROPCs in the medium.

Aboveground external radiation from soil will be adjusted for the fraction of time that the receptor is assumed to spend on the soil surface, chosen by scientific judgment based on narrative information in published and internet wildlife articles.

There is also a roughness factor of 0.7 to correct for absorption of radiation by uneven soil contours and an elevation correction factor (*ECF*) to adjust *DCF*s to account for most ecological receptors having most of their bodies closer to the ground than the humans for which the *DCF*s were derived. The *ECF* is 2 for all receptors except mule deer, which are large enough to receive radiation at approximately the same height as humans (Sample et al. 1997). The elevation correction factor of 2 for all receptors except the mule deer assumes that these receptors receive twice the exposure from the same concentrations of ROPCs in soil. External radiation *DCF*s are presented in Supplement 4.

Belowground external radiation from soil will be modeled by using the decay energies and tissue absorption fractions presented in Supplement 4. Equations to calculate belowground external exposure are presented by Sample et al. (1997). Belowground exposure is adjusted for the fraction of time that the

receptor is assumed to be exposed underground, chosen by scientific judgment based on narrative information in published wildlife articles.

The fraction of time a receptor spends above ground (on the ground surface) and belowground are assumed to be:

| | Fraction of time above ground | Fraction of time below ground |
|---|---|---|
| Plants | 0.5 | 0.5 |
| Terrestrial invertebrates | 0.5 | 0.5 |
| Mule deer | 1 | 0 |
| Mourning dove | 1 | 0 |
| Great Basin pocket mouse | 0.3 | 0.7 |
| Western meadowlark | 1 | 0 |
| Coyote | 0.7 | 0.3 |
| Burrowing owl | 0.5 | 0.5 |
| Red-tailed hawk | 0.05[a] | 0 |

[a] It is assumed the red-tailed hawk spends 95 % of the time in flight or perched such that there is negligible exposure due to radionuclides in the soil.

As presented implicitly by Eckerman and Ryman (1993) and in Blaylock, Frank, and O'Neal (1993), the external dose (rad/day) to all receptors for a given radionuclide in the will be calculated as:

$$DD_E = \sum (\text{ROPC Concentration} \cdot \text{Dose Conversion Factor})$$

where $DD_E$ is the external radiation dose (rad/day). External radiation exposure occurs from three media types; soil, air, and water. Soil and water exposure can include receptor exposure to the surface of the soil or water, and exposure from immersion in the soil and water. Air exposure is solely due to immersion. For terrestrial receptors, the external dose from water exposure is considered negligible because the receptor contact with the Columbia River is limited.

$$DD_E = DD_{E_{soil}} + DD_{E_{air}}$$

where:

$DD_E$ = total external radiation dose (rad/day)

$DD_{E_{soil}}$ = external radiation dose from sediment (rad/day)

$DD_{E_{air}}$ = external radiation dose from air (rad/day)

The external daily dose due to soil contact is:

$$DD_{E_{soil}} = DD_{abovegrd} + DD_{belowgrd}$$

where:

$DD_{E_{soil}}$ = external radiation dose from soil (rad/day)

$DD_{abovegrd}$ = external dose from exposure to aboveground soil (rad/day)

$DD_{belowgrd}$ = external dose from exposure to belowground soil (rad/day)

The total external dose from all ROPCs in soil is the sum of the external doses from each ROPC. Following the method of Sample et al. (1997), the external dose from exposure to soil ($DD_{abovegrd}$ and $DD_{belowgrd}$ ) will be calculated as:

$$DD_{abovegrd} = F_{above} \cdot F_{ruf} \cdot Cs_2 \cdot DCF_{grd} \cdot CFb \cdot ECF \qquad \text{(Sample et al. (1997), Eq. 9)}$$

$$DD_{belowgrd} = 1.05 \cdot F_{below} \cdot Cs_2 \cdot E_{\gamma} \cdot AbF_{\gamma} \cdot CFa \qquad \text{(modified Sample et al. (1997), Eq. 10)}$$

Factoring $Cs_2$, substituting variables to represent the dose factor yields the following equations:

$$DD_{E_{soil}} = \left(F_{above} \cdot F_{ruf} \cdot DCF_{grd} \cdot CFb \cdot ECF + 1.05 \cdot F_{below} \cdot Cs_2 \cdot E_{\gamma} \cdot AbF_{\gamma} \cdot CFa\right) \cdot Cs_2 \text{, or}$$

$$DD_{E_{soil}} = \left(DCF_{abovegrd} + DCF_{belowgrd}\right) \cdot Cs_2 \text{, or}$$

$$DD_{E_{soil}} = DCF_{soil} \cdot Cs_2$$

Thus, the external dose to a terrestrial receptor due to soil exposure to a ROPC will be calculated according to:

$$DD_{E_{soil}} = DCF_{soil} \cdot Cs_2 \text{, and}$$

$$DF_{soil} = F_{above} \cdot F_{ruf} \cdot DCF_{grd} \cdot CFb \cdot ECF + 1.05 \cdot F_{below} \cdot F_{ruf} \cdot E_{\gamma} \cdot AbF_{\gamma} \cdot CFa$$

where:

$DD_{E_{soil}}$ = external radiation dose from soil (rad/day)

$DCF_{soil}$ = factor for converting activity of radionuclide in soil to external dose from exposure to aboveground and belowground soil in units of rad/day per pCi/g ($DCF_{soil} = DCF_{abovegrd} + DCF_{belowgrd}$)

$DCF_{abovegrd}$ = factor for converting activity of radionuclide in soil to external dose from exposure to aboveground soil in units of rad/day per pCi/g ( $DCF_{abovegrd} = F_{above} \cdot F_{ruf} \cdot DCF_{grd} \cdot CFb \cdot ECF$ )

$DCF_{belowgrd}$ = factor for converting activity of radionuclide in soil to external dose from exposure to belowground soil in units of rad/day per pCi/g ( $DCF_{belowgrd} = 1.05 \cdot F_{below} \cdot F_{ruf} \cdot E_{\gamma} \cdot AbF_{\gamma} \cdot CFa$ )

$DCF_{grd}$ = dose conversion factor for external radiation by the ROPC from soil contaminated to a depth of 5 cm (Sv/s per Bq/m$^3$), using effective doses (dose averaged over all body organs) (Eckerman and Ryman 1993, Table III.4)

$F_{above}$ = fraction of time spent above ground (unitless)

$F_{below}$ = fraction of time spent below ground surface (unitless)

$F_{ruf}$ = dose rate reduction factor accounting for ground roughness (unitless); a value of 0.7 is considered a representative average reduction factor (Eckerman and Ryman 1993)

$CFb$ = $5.115 \times 10^{11}$, factor[1] to convert Sv/s per Bq/m$^3$ to rad/day per pCi/g

$ECF$ = elevation correction factor to adjust dose coefficient for effective height of receptor above ground (unitless, Sample et al. 1997)

1.05 = conversion factor to account for immersion in soil rather than water (unitless)

$E_\gamma$ = photon energy emitted during transition from a higher to a lower energy state × proportion of disintegrations producing γ radiation (MeV/disintegration)

$AbF_\gamma$ = absorbed fraction of energy from gamma energy $E_\gamma$ (unitless)

$CFa$ = unit conversion factor[2], $5.122 \times 10^{-5}$ rad/day per pCi/g per MeV/disintegration

$Cs_2$ = activity of the radionuclide in untilled soil of 2 cm depth (pCi/g)

Note that the modification of Equation 10 of Sample et al. (1997) includes terms for energy emitted and the fraction absorbed. Sample et al. (1997) present absorbed fractions for select radionuclides but neglect to explicitly show them in Equation 10; thus, the equation above has been modified to clearly show application of the absorbed fraction in the dose factor computation. $DCF_{grd}$ for soil, $E_\gamma$, and $AbF_\gamma$ are reported in Supplement 4. To calculate external exposure to radionuclides in soil, $DCF_{soil}$ values will be multiplied by the modeled activities of the corresponding radionuclides in surface soil at each exposure location.

The external dose (rad/day) to all receptors from air will be calculated as:

$$DD_{E_{air}} = DCF_{air} \cdot C_a \qquad \text{(Eckerman and Ryman 1993)}$$

where:

$DD_{E_{air}}$ = external radiation dose from air (rad/day)

$DCF_{air}$ = factor for converting activity of the ROPC in air to external dose from air (rad/day per pCi/m$^3$

$C_a$ = activity of the ROPC in air, calculated as described in Section 6.1 (pCi/m$^3$)

The external dose conversion factor for air ($DCF_{air}$) will be calculated as follows:

$$DCF_{air} = 3.197 \times 10^5 \cdot DCF \qquad \text{(Eckerman and Ryman 1993)}$$

[1] Per Eckerman and Ryman, a soil density conversion factor of $1.6\times10^3$ kg/m$^3$ is applied such that the appropriate conversion factor is: [(Sv/s)/(Bq/m$^3$)] × (100 rad/Sv) × (86400 s/day) × (0.037 Bq/pCi) × ($10^3$ g/kg) × ($1.6\times10^3$ kg/m$^3$) = $5.115\times10^{11}$ (rad·g)/(pCi·day)

[2] Conversion factor: (pCi/g) × (MeV/disintegration) × (0.037 disintegration/s)/pCi × ($1.602\times10^{-8}$ rad/(MeV/g)) × (86400 s/day) = $5.122\times10^{-5}$ rad/day

where:

$3.197 \times 10^5$ = conversion factor[3] to convert Sv/s per Bq/m$^3$ to rad/day per pCi/m$^3$ (Eckerman and Ryman 1993)

$DCF$ = dose conversion factor for external radiation from immersion in air (Sv/s per Bq/m$^3$, Eckerman and Ryman 1993)

For all ROPCs, values of $DCF_{air}$ for air are reported in Supplement 4. To calculate external exposure to radionuclides in air, $DCF_{air}$ values will be multiplied by the modeled activities of the corresponding radionuclides in air at each exposure location.

**Internal Dose**

The internal exposure to radionuclides will be calculated from the activity in the receptor's tissues rather than from the daily ingestion. The internal activities of ROPCs are calculated by using *BCF*s and ROPC activities in soil and food as described in Section 8.2.3.3. Internal radiation doses are calculated by multiplying the activity in tissues by the sum of alpha, beta, and gamma decay energies, where alpha and beta energies are assumed to be completely absorbed. Because gamma rays, like X-rays, may pass through the tissues without depositing their energy, gamma energies are adjusted to account for greater absorption by larger organisms at a given energy level and for greater absorption by all receptors at lower energy levels. Radiation by daughter radionuclides was also included in internal dose calculations. Daughter radionuclides are produced by decay of parent radionuclides and short-lived radionuclides could be expected to contribute to the receptor's internal dose. Exposures are calculated by assuming that the decay products of all short-lived ROPCs are in secular equilibrium and equal to the activity of the parent multiplied by the fraction of the decays in the immediately preceding generation that yield the daughter radionuclide. Decay energies (Eckerman and Ryman 1993) and absorption fractions for gamma radiation (Blaylock, Frank, and O'Neal 1993; Sample et al. 1997) of daughter radionuclides are reported in Supplement 4.

Adapting Equation 11 of Sample et al. (1997), the internal dose (rad/day) to plants, terrestrial invertebrates, and wildlife receptors will be calculated as follows:

$$DD_I = \sum QF \cdot C_{TP \text{ or } INV \text{ or } A} \cdot E \cdot CF \cdot AbF \qquad \text{(modified Sample et al. (1997), Eq. 11)}$$

where:

$DD_I$ = internal radiation dose (rad/day)

$QF$ = quality factor for relative biological effect of radiation (unitless) (Kocher and Trabalka 2000)

$QF_\alpha$ = 10 for alpha radiation

$QF_\beta$ = 1 for beta radiation

$QF_\gamma$ = 1 for gamma radiation

[3] Conversion factor: [(Sv/s)/(Bq/m$^3$)] × (100 rad/Sv) × (86400 s/day) × (0.037 Bq/pCi) = $3.197\times10^3$ (rad·m$^3$)/(pCi·day)

$C_{TP \text{ or } INV \text{ or } A}$ = activity of radionuclide in receptor tissue (terrestrial plant, invertebrate, or animal - see Section 8.2.3.3) (pCi/g)

$CF$ = unit conversion factor, $5.12 \times 10^{-5}$ rad/day per pCi/g per MeV/disintegration

$E$ = average energy emitted as alpha, beta, or gamma radiation by radionuclide $i$ × proportion of disintegrations producing alpha, beta, or gamma radiation (MeV per disintegration)

$AbF$ = absorbed fraction of energy from alpha, beta, or gamma energy (unitless)

The equation above can be expanded to show the variables specific to the types of radiation emitted from the ROPC.

$$DD_I = \left(QF_\alpha \cdot E_\alpha \cdot AbF_\alpha + QF_\beta \cdot E_\beta \cdot AbF_\beta + QF_\gamma \cdot E_\gamma \cdot AbF_\gamma\right) \cdot CF \cdot C_{TP \text{ or } INV \text{ or } A}$$

substituting variables to represent the dose factor yields:

$$DD_I = DCF_I \cdot C_{TP \text{ or } INV \text{ or } A}, \text{ and}$$

$$DCF_I = \left(QF_\alpha \cdot E_\alpha \cdot AbF_\alpha + QF_\beta \cdot E_\beta \cdot AbF_\beta + QF_\gamma \cdot E_\gamma \cdot AbF_\gamma\right) \cdot CF$$

where $QF_\alpha$, $QF_\beta$, and $QF_\gamma$ are the quality factors of alpha, beta, and gamma radiation (respectively), $E_\alpha$, $E_\beta$, and $E_\gamma$ are the product of the average energy emitted and proportion of disintegrations producing alpha, beta, and gamma radiation (respectively), and $AbF_\alpha$, $AbF_\beta$, and $AbF_\gamma$ are absorbed fraction of energy from alpha, beta, and gamma energy (respectively). Other terms are as defined above. Internal exposure dose factors for aquatic receptors include the contribution of daughter products:

$$DCF_{I+D} = DCF_{I_{parent}} + \sum_{i=1}^{n} DCF_{I_D}$$

where

$DCF_{I+D}$ = factor for converting activity of radionuclide (and its daughter products) to internal dose from ingestion of contaminated food in units of rad/day per pCi/g

$DCF_{I_{parent}}$ = factor for converting activity of the parent radionuclide to internal dose from exposure in units of rad/day per pCi/g

$\sum_{i=1}^{n} DCF_{I_D}$ = sum of the daughter product dose factors for each applicable daughter product (1 through $n$) for converting activity of the daughter product to internal dose in units of rad/day per pCi/g

Values of $E$ and $AbF$ for each radiation (alpha, beta and gamma) are reported in Supplement 4 for all ROPCs and their daughters, respectively. To calculate internal exposure to radionuclides in soil, $DCF_I$ values will be multiplied by the modeled activities of the corresponding radionuclides in receptor tissues at each exposure location.

### 8.2.4 Quantification of Exposure at the Columbia River Maximum (Aquatic Receptors)

Calculating ecological screening quotients (*ESQ*s) for receptors exposed to contaminated air, water, and sediment in the Columbia River area by direct contact with air, sediment, and water and by ingestion of water, sediment, and biota requires the EPCs of COPCs and ROPCs in air, water, sediment, and aquatic biota. The EPCs of COPCs in biota (Section 8.2.4.1) are required to calculate the *DD* by ingestion (Sections 8.2.4.2 and 8.2.4.3) and internal radiation dose for predator receptors. The total radiation dose for all receptors exposed to ROPCs is the sum of the external and internal radiation doses for all ROPCs (Section 8.2.4.4).

The exposure of terrestrial predators to terrestrial prey at the onsite ground maximum is higher than exposure at the Columbia River maximum because concentrations in the soil at the onsite ground maximum are by definition higher than concentrations at the Columbia River maximum, which is farther from the emission source than the onsite ground maximum. Ingestion of terrestrial prey by eagles and mink at the Columbia River maximum (which would be modeled by the same exposure pathways as ingestion by coyotes and hawks) will be less than the maximum onsite risk already calculated for ingestion of terrestrial prey by coyotes and hawks at the onsite ground maximum because the concentrations in prey will be much lower in proportion to the soil concentrations. Although body weight, metabolism, and ingestion rate differ among organisms, these differences represent a much smaller contribution to exposure and risk predictions than does the much larger difference in soil concentrations between the On-Site Ground Maximum and the Columbia River Maximum.

The intent of evaluating exposure at the Columbia River maximum is to determine the risks from deposition of COPCs and ROPCs into surface water. To ensure that exposure by deposition of COPCs and ROPCs into surface water is maximized, exposure to aquatic and benthic prey is evaluated at the Columbia River maximum and propagated through the food chain to higher trophic level receptors.

#### 8.2.4.1 EPCs in Aquatic Biota

Calculating EPCs for tissues of aquatic plants and animals exposed by direct contact with air, water, and sediment requires the EPCs of COPCs and the ROPCs in air ($C_a$), dissolved in water ($C_{dw}$), in water column as a total ($C_{wctot}$), in sediment ($C_{sed}$), as well as the receptor bioaccumulation and uptake factors for the COPCs and ROPCs (Section 8.2.5.4).

**EPCs in Aquatic Plants (Trophic Level 1)**

For floating and rooted aquatic plants exposed to COPCs and ROPCs in surface water and sediment, respectively, and fed upon by Canada geese, the tissue EPC will be calculated in accordance with the SLERAP (EPA 1999), using concentration of COPCs and ROPCs in river sediment ($C_{sed}$):

$$C_{AP} = C_M \cdot BCF_M \cdot CF \qquad \text{(SLERAP Eq. 5-3)}$$

where:

$C_{AP}$ = concentration or activity of COPCs or ROPCs in aquatic plant tissue (mg/kg or pCi/g)

$C_M$ = concentration or activity of COPCs or ROPCs in the media of interest ($C_{dw}$, dissolved in surface water in mg/L or pCi/L, or $C_{sed}$, sediment in mg/kg, or pCi/g)

$BCF_M$ = media-to-plant bioconcentration factor (water-to-tissue uptake factor for floating plants in L/kg, or unitless sediment-to-tissue uptake factor for rooted aquatic plants)

$CF$ = conversion factor of 0.001 kg/g (to convert pCi/kg to pCi/g) for ROPCs

In order to be conservative, it is assumed that the diet of the Canada goose is exposed to COPCs and ROPCs in both surface water and sediment (Figure 8-9). The values of $BCF_M$ for the SLERA are discussed in Section 8.2.5.4. For all COPCs and ROPCs, values for $BCF_M$ are reported in Supplement 4.

**EPCs in Benthic Invertebrates (Trophic Level 2)**

For benthic invertebrates exposed to organic COPCs and ROPCs in sediment and fed upon by spotted sandpipers and great blue herons, the tissue EPC will be calculated by using either a measured sediment-to-invertebrate bioconcentration factor or a calculated sediment porewater concentration and an empirically determined water-to-invertebrate bioconcentration factor. The EPCs for constituents with measured bioconcentration factors are calculated in accordance with EPA (1999) draft guidance:

$$C_{INV} = C_{sed} . BCF_S \qquad \text{(SLERAP Eq. F-1-3)}$$

or

$$C_{INV} = C_{IW} . BCF_W \cdot CF \qquad \text{(SLERAP Eq. 5-4)}$$

where:

$C_{INV}$ = concentration or activity of COPCs or ROPCs in benthic invertebrate tissue (mg/kg, pCi/g)

$C_{sed}$ = concentration or activity of COPCs or ROPCs in sediment (mg/kg, or pCi/g)

$BCF_S$ = sediment-to-tissue bioconcentration factor for benthic invertebrates (unitless)

$C_{IW}$ = concentration or activity of COPCs or ROPCs in sediment interstitial water (mg/L, or pCi/L), explained below

$BCF_W$ = water-to-tissue bioconcentration factor for benthic invertebrates (L/kg)

$CF$ = conversion factor of 0.001 kg/g (to convert pCi/kg to pCi/g) for ROPCs

When measured values were not available, the sediment-to-benthic invertebrate $BCF_S$ are the average of all available measured bioconcentration factors, as was done for terrestrial invertebrates. For many organic COPCs, measured sediment-to-tissue $BCF_S$ values are not available. Instead, per EPA draft guidance, values of $BCF_W$ for organic COPCs for which no measured values were available were calculated with an equation, given in SLERAP (Eq. C-1-4) and derived by regression analysis of uptake of several organic chemicals from water by aquatic invertebrates as a function of log $K_{ow}$ (Southworth et al. 1978). According to EPA draft guidance, it is appropriate to use a calculated concentration of an organic COPC in sediment porewater when using an aquatic $BCF_W$ value, as calculated by the regression equation for aquatic invertebrates (SLERAP Eq. C-1-4). The concentration in sediment porewater is determined by equilibrium partitioning, which is explained in Section 8.2.3.1. Thus, the tissue EPC calculated by using the calculated $BCF_W$ (SLERAP Eq. 5-4) will use a sediment interstitial water concentration:

$$C_{IW} = \frac{C_{sed}}{f_{oc,bs} \cdot K_{oc}} \qquad \text{(SLERAP Eq. 5-5)}$$

where:

$C_{IW}$ = concentration of organic COPC in sediment porewater (mg/L)
$C_{sed}$ = concentration of organic COPC in sediment (mg/kg)
$K_{oc}$ = soil organic carbon partitioning coefficient (L/kg)
$f_{oc,bs}$ = fraction of bed sediment that is organic carbon (unitless)

Note: SLERAP Equation 5-5 first appears in Section 8.2.3.1.

The values of $BCF_W$ for the SLERA are discussed in Section 8.2.5.4. For all COPCs and ROPCs, values for $BCF_W$ are reported in Supplement 4.

**EPCs in Aquatic Invertebrates (Trophic Level 2)**

For aquatic invertebrates exposed to COPCs and ROPCs in surface water, the tissue EPC will be calculated in accordance with EPA (1999) draft guidance, using dissolved concentrations ($C_{dw}$) in water:

$$C_{INV} = C_{dw} \cdot BCF_{INV} \cdot CF \qquad \text{(SLERAP Eq. 5-3)}$$

where:

$C_{INV}$ = concentration or activity of COPC/ROPC in aquatic invertebrate tissue (pCi/g)
$C_{dw}$ = concentration or activity of COPC/ROPC dissolved in surface water (pCi/L)
$BCF_{INV}$ = water-to-tissue uptake factor for aquatic invertebrates (L/kg)
$CF$ = conversion factor of 0.001 kg/g (to convert pCi/kg to pCi/g) for ROPCs

For all ROPCs, values for $BCF_{INV}$ are reported in Supplement 4.

**EPCs in Fish (Trophic Levels 2, 3, and 4)**

Trophic-level-specific *FCM*s will be used to calculate the concentrations of COPCs and ROPCs in fish. *FCM*s adjust the calculated concentration in fish tissue to account for bioaccumulation from one trophic level to another (see Section 8.2.5.3). For planktivorous fish (trophic level 2, $FCM_2$), omnivorous fish (trophic level 3, $FCM_3$), and carnivorous fish (trophic level 4, $FCM_4$) exposed to dissolved concentrations ($C_{dw}$) of constituents in surface water, aquatic biota, and fish, the tissue EPC will be calculated, respectively, as follows:

$$C_{F_i} = BCF_W \cdot FCM_i \cdot C_{dw} \cdot CF \qquad \text{(SLERAP Eq. 5-7)}$$

where:

$C_{F_i}$ = concentration or activity of COPC or ROPC in fish tissue for trophic level *i* fish (mg/kg or pCi/g)
$C_{dw}$ = concentration or activity of COPC or ROPC dissolved in surface water (mg/L or pCi/L)
$BCF_W$ = water-to-tissue uptake factor for fish (L/kg)

$FCM_i$ = food chain multiplier for trophic level *i* fish (unitless)

$CF$ = conversion factor of 0.001 kg/g (to convert pCi/kg to pCi/g) for ROPCs

The values of $BCF_W$ and *FCM*s for the SLERA are discussed in Section 8.2.5.4. For all COPCs and ROPCs, values for $BCF_W$ are reported in Supplement 4 with values for $FCM_2$, $FCM_3$ and $FCM_4$.

**EPCs in Mammal and Bird Tissues (Trophic Levels 2, 3, and 4)**

For wildlife receptors, the tissue EPC will be calculated in accordance with EPA (1999) draft guidance as the sum of the contributions from the different types of material ingested. Trophic-level-specific *FCM*s will be used to calculate the concentrations of ROPCs in mammals and birds. The *FCM*s adjust the calculated concentration in animal tissue to account for bioaccumulation from one trophic level to another (see Section 8.2.5.3). The equations are adapted from EPA draft guidance (SLERAP Eq. 5-12),

such that the concentration in a omnivore is: (modified SLERAP Eq. 5-12)

$$C_{OM} = C_A \cdot \frac{FCM_{OM}}{FCM_A} \cdot F_A + C_{AP} \cdot BCF_P \cdot F_P + Cs_{sed} \cdot BCF_S \cdot P_S + C_{wctot} \cdot BCF_W \cdot P_W \cdot CF$$

and the concentration in an herbivore is: (modified SLERAP Eq. 5-11)

$$C_H = C_{AP} \cdot BCF_P \cdot F_P + Cs_{sed} \cdot BCF_S \cdot P_S + C_{wctot} \cdot BCF_W \cdot P_W \cdot CF$$

and the concentration in a carnivore is: (modified SLERAP Eq. 5-13)

$$C_C = C_A \cdot \frac{FCM_C}{FCM_A} \cdot F_A + Cs_{sed} \cdot BCF_S \cdot P_S + C_{wctot} \cdot BCF_W \cdot P_W \cdot CF$$

where:

$C_{OM/H/C}$ = concentration of constituent in omnivorous, herbivorous, or carnivorous (respectively) receptor tissue (mg/kg FW tissue or pCi/g FW tissue)

$C_A$ = concentration of constituent in ingested animal prey (mg/kg FW tissue or pCi/g FW tissue)

$FCM_{OM/C}$ = food-chain multipliers for the omnivorous receptor (unitless)

$FCM_A$ = food-chain multipliers for ingested prey type (unitless)

$F_A$ = fraction of diet from animal tissue (unitless)

$C_{AP}$ = concentration of constituent in ingested aquatic plant tissue (mg/kg plant or pCi/g plant)

$BCF_P$ = plant-to-tissue uptake factor for omnivorous/herbivorous receptor (mg/kg FW tissue per mg/kg plant)

$F_P$ = fraction of diet from plant tissue (unitless)

$Cs_{sed}$ = concentration of constituent in ingested sediment (mg/kg or pCi/g),

$BCF_S$ = soil-to-tissue uptake factor for omnivorous/herbivorous/carnivorous receptor (mg/kg FW tissue per mg/kg DW soil)

$P_S$ = proportion of consumed soil or sediment that is contaminated (unitless)

$C_{wctot}$ = concentration of constituent in unfiltered water from Columbia River maximum location (mg/L or pCi/L)

$BCF_W$ = water-to-tissue uptake factor for omnivorous/herbivorous/carnivorous receptor (mg/kg FW tissue per mg/L water)

$P_W$ = proportion of consumed water that is contaminated (unitless)

$CF$ = conversion factor for radionuclides, 0.001 pCi/kg to pCi/g

The plant-to-tissue uptake factors, water-to-tissue uptake factors, and soil-to-tissue uptake factors, which will be used for sediment, are calculated per EPA (1999) draft guidance from the receptor's ingestion rate and the published biotransfer factor (*Ba*), that is:

$$BCF_F = Ba \cdot IR_F \cdot BW \qquad \text{(modified SLERAP Eq. D-1-1)}$$

and:

$$BCF_M = Ba \cdot IR_M \cdot BW \qquad \text{(modified SLERAP Eq. D-1-2)}$$

where:

$BCF_A$ = food-to-animal bioconcentration factor for receptor (mg COPC/kg FW tissue per mg COPC/kg FW food)

$BCF_M$ = media-to-animal bioconcentration factor for receptor (mg COPC/kg FW tissue per mg COPC/kg DW media)

$Ba$ = ingestion-to-tissue transfer factor (d/kg)

$IR_F$ = daily food ingestion rate (kg/kg BW/d)

$IR_M$ = daily media ingestion rate, such that:

$IR_w$ = rate of water consumption (L/kgBW/d)

$IR_S$ = $SFr \cdot IR_F$ = rate of sediment consumption (kg/kg BW/d)

$SFr$ = sediment ingested per unit food ingested (unitless)

$BW$ = body weight of receptor (kg)

Note: SLERAP Equations D-1-1 and D-1-2 first appear in Section 8.2.3.1.

The values of $Ba$, $BCF_P$, $BCF_S$, and $BCF_W$ are discussed in Section 8.2.5.3 and are reported in Supplement 4. Values for $IR_F$, $F_P$, $BW$, $SFr$, and water ingestion ($IR_w$) for receptors exposed at that Columbia River are given in the receptor profiles in Section 8.1.3.3.

The EPCs for COPCs in aquatic plants, benthic invertebrates, and aquatic biota are used in the equations for modeling intake to aquatic ecological receptors (i.e., the ingestion *DD*). EPCs for ROPCs will be used to calculate internal radiation doses.

#### 8.2.4.2 Modeling Intake to Aquatic Ecological Receptors

Ingestion exposure of aquatic receptors will be evaluated using exclusive diets: floating aquatic plants for the Canada goose, benthic invertebrates for the spotted sandpiper, and fish for the bald eagle and mink. Ingestion of terrestrial food items is not included because the intent is to determine the risk from COPCs and ROPCs in surface water and sediment. The ingestion *DD* (mg/kg/d) for aquatic receptors exposed to COPCs in sediment or surface water will be calculated as the sum of plant tissue, animal tissue, water, and sediment intakes:

$$DD = \sum IR_F \cdot C_i \cdot P_i \cdot F_i + \sum IR_M \cdot C_M \cdot P_M \qquad \text{(SLERAP Eq. 5-1)}$$

or:

$$DD = DD_A + DD_P + DD_{sed} + DD_W$$

where:

| | | |
|---|---|---|
| $DD$ | = | daily dose by ingestion (mg/kg BW/d) |
| $IR_F$ | = | receptor plant or animal food item ingestion rate (kg/kg BW-day) |
| $C_i$ | = | constituent concentration in $i^{th}$ plant or animal food item (mg/kg) |
| $P_i$ | = | proportion of $i^{th}$ food item that is contaminated (unitless) - assumed to be equal to 1 |
| $F_i$ | = | fraction of diet consisting of plant or animal food item *i* (unitless) |
| $IR_M$ | = | media *M* ingestion rate (kg/kg BW-day [soil or bed sediment] or L/kg BW-day [water]) |
| $C_M$ | = | constituent concentration in media *M* (mg/kg [soil or bed sediment] or mg/L [water]) |
| $P_M$ | = | proportion of ingested media *M* that is contaminated (unitless) |
| $DD_A$ | = | daily dose by animal ingestion (mg/kg BW/d) |
| $DD_P$ | = | daily dose by plant ingestion (mg/kg BW/d) |
| $DD_{sed}$ | = | daily dose by sediment ingestion (mg/kg BW/d) |
| $DD_W$ | = | daily dose by water ingestion (mg/kg BW/d) |

As defined by Equation 5-1 of the SLERAP:

$$DD_A = C_A \cdot IR_F \cdot F_A$$

$$DD_P = C_{TP} \cdot IR_F \cdot F_P$$

$$DD_{sed} = C_{sed} \cdot IR_F \cdot SFr$$

$$DD_W = C_{wctot} \cdot IR_W$$

where:

| | | |
|---|---|---|
| $DD_A$ | = | daily dose by animal ingestion (mg/kg BW/d) |
| $DD_P$ | = | daily dose by plant ingestion (mg/kg BW/d) |

$DD_w$ = daily dose by water ingestion (mg/kg BW/d)
$DD_{sed}$ = daily dose by sediment ingestion (mg/kg BW/d)
$C_A$ = concentration of constituent in ingested animal tissue (mg/kg or pCi/g)
$C_P$ = concentration of constituent in ingested plant tissue (mg/kg or pCi/g)
$IR_F$ = food (plant or prey, as applicable) ingestion rate of receptor (kg/kg BW/d)
$IR_W$ = water ingestion rate of receptor (kg/kg BW/d)
$F_A$ = fraction of diet from animals (unitless)
$F_P$ = fraction of diet from plants (unitless)
$C_{sed}$ = concentration of constituent in ingested sediment (mg/kg or pCi/g)
$SFr$ = soil ingested per unit of food ingested (unitless)
$C_{wctot}$ = concentration of constituent in water column (mg/L or pCi/L)

Proportion of contaminated food and media ($P_i$ and $P_M$), absorption efficiency ($AE$), the area use factor ($AUF$), and the temporal use factor ($TUF$) are assumed to be equal to 1, so they do not appear in the exposure equations.

#### 8.2.4.3 Receptor-specific Exposure Equations for Aquatic Receptors

The complete equations for $DD$ and $C_A$ for each receptor are presented below.

**Herbivore: Canada Goose (Trophic Level 2)**

Canada geese are herbivores that ingest aquatic plants, but they ingest water and sediment also with their food. Thus,

$$DD = DD_P + DD_{sed} + DD_W, \text{ or} \qquad \text{(Equation 8-13)}$$

$$DD_{Goose} = C_{TP} \cdot IR_F \cdot F_P + C_{sed} \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W$$

where $DD_P$, $DD_W$, and $DD_{sed}$ are as given above. The Canada goose food ingestion rate ($IR_F$), water ingestion rate ($IR_w$), and dietary fractions ($F_P$ and $SFr$) are given in the receptor profiles in Section 8.1.3.3. The aquatic food web (Figure 8-9) shows the prey types for the Canada goose. The Canada goose will be assumed to have an exclusive diet of aquatic plants.

Canada goose tissue concentrations of ROPCs will be calculated by an equation adapted from the SLERAP Equation 5-11:

$$C_{Goose} = C_{TP} \cdot BCF_P \cdot F_P + C_{sed} \cdot BCF_S + C_{wctot} \cdot BCF_W \cdot P_W \cdot CF$$

where:

$C_{Goose}$ = concentration in receptor tissue (mg/kg or pCi/g)
$C_{TP}$ = concentration in ingested plant tissue (mg/kg or pCi/g)
$F_P$ = fraction of diet from plants (unitless)
$BCF_P$ = plant-to-tissue uptake factor for receptor (mg/kg FW tissue per mg/kg plant)

$C_{sed}$ = concentration of constituent in ingested sediment (mg/kg or pCi/g)
$BCF_S$ = sediment-to-tissue uptake factor for receptor (mg/kg FW tissue per mg/kg DW sediment)
$C_{wctot}$ = concentration of constituent in unfiltered water from Columbia River maximum location (mg/L or pCi/L)
$BCF_W$ = water-to-tissue uptake factor for receptor (kg FW tissue per mg/L water)
$P_w$ = proportion of consumed water that is contaminated (unitless)
$CF$ = conversion factor for radionuclides, 0.001 pCi/kg to pCi/g

For ROPCs, the tissue concentration resulting from ingested water ($C_{wctot} \times BCF_W \times P_W$) must be converted from pCi/kg to pCi/g by multiplying by 0.001 kg/g. Sediment-to-tissue uptake factors ($BCF_S$), plant-to-tissue uptake factors ($BCF_P$), and water-to-tissue uptake factors ($BCF_W$) for the Canada goose are reported in Supplement 4.

**Carnivore: Spotted Sandpiper (Trophic Level 3)**

Spotted sandpipers are carnivores that ingest benthic invertebrates, but they also ingest water and sediment with their food (Figure 8-9). Thus,

$DD = DD_A + DD_{sed} + DD_W$, or (Equation 8-14)

$$DD_{Piper} = C_{INV} \cdot IR_F \cdot F_A + C_{sed} \cdot IR_F \cdot SFr + C_{wctot} \cdot IR_W$$

where $DD_A$, $DD_W$, and $DD_{sed}$ are as given above. The spotted sandpiper food ingestion rate ($IR_F$), water ingestion rate ($IR_w$) and dietary fractions ($F_A$ and $SFr$) are given in the receptor profiles in Section 8.1.3.3. The aquatic food web (Figure 8-9) shows the prey types of the spotted sandpiper. To evaluate exposure specifically from sediment, the spotted sandpiper is assumed to have an exclusive diet of benthic invertebrates.

Sandpiper tissue concentrations of ROPCs will be calculated by an equation adapted from EPA draft guidance (SLERAP Equation 5-13). *FCM*s ($FCM_3$ for trophic level 3) are used to account for bioaccumulation from ingested animal tissue. Bioaccumulation is the process whereby certain toxic substances collect in living tissues, and biomagnification is the transfer and concentration of chemicals through successive trophic levels via ingestion of prey. The *FCM* ratios are used to estimate the biomagnification for ingestion of lower trophic food by higher trophic level animals. See Section 8.2.5.3 for more details on *FCM*s.

$$C_{Piper} = C_{INV} \cdot \frac{FCM_3}{FCM_2} \cdot F_A + C_{sed} \cdot BCF_S + C_{wctot} \cdot BCF_W \cdot CF$$

where:

$C_{Piper}$ = concentration of constituent in receptor tissue (mg/kg or pCi/g)
$C_{INV}$ = concentration of constituent in prey tissue (mg/kg or pCi/g)
$FCM_3$ = food-chain multiplier for the receptor (unitless)
$FCM_2$ = food-chain multipliers for ingested prey type (unitless)

$F_A$ = fraction of diet from benthic invertebrate tissue (unitless)
$C_{sed}$ = concentration of constituent in ingested sediment (mg/kg or pCi/g)
$BCF_S$ = sediment-to-tissue uptake factor for receptor (mg/kg FW tissue per mg/kg DW sediment)
$C_{wctot}$ = concentration of constituent in unfiltered water from Columbia River maximum location (mg/L or pCi/L)
$BCF_W$ = water-to-tissue uptake factor for receptor (kg FW tissue per mg/L water)
$CF$ = conversion factor of 0.001 kg/g (to convert pCi/kg to pCi/g) for ROPCs

For ROPCs, the tissue concentration resulting from ingested water ($C_{wctot} \times BCF_W$) must be converted from pCi/kg to pCi/g by multiplying by a conversion factor ($CF$) of 0.001 kg/g. Because the diet of the spotted sandpiper is assumed to be benthic invertebrates, $F_{INV} = 1$. The *FCM*s for the sandpipers ($FCM_3$) and their invertebrate prey ($FCM_2$), along with sediment-to-tissue bioconcentration factors ($BCF_S$) and water-to-tissue bioconcentration factors ($BCF_W$), are reported in Supplement 4.

**Carnivore: Great Blue Heron (Trophic Level 4)**

Great blue heron are carnivores that ingest planktivorous fish, omnivorous fish, and small invertebrates, but they also ingest water with their food (Figure 8-9). The SLERA will evaluate the exposure of heron assuming ingestion of planktivorous fish (Trophic Level 2), omnivorous fish (Trophic Level 3), benthic invertebrates (Trophic Level 2), and water (Figure 8-9). Thus,

$DD = DD_A + DD_W$, or (Equation 8-15)

$$DD = C_A \cdot IR_F \cdot F_A + C_{wctot} \cdot IR_W$$

where $DD_A$ and $DD_W$ are as given above. In calculating the $DD_A$, it is assumed the heron's diet consists of 5 % Trophic Level 2 fish, 89 % Trophic Level 3 fish, and 6 % Trophic Level 2 benthic invertebrates (EPA 1993c). The expanded equation for the daily dose to the great blue heron ($DD_{Heron}$) is:

$$DD_{Heron} = C_{F_2} \cdot IR_F \cdot F_{F_2} + C_{F_3} \cdot IR_F \cdot F_{F_3} + C_{INV} \cdot IR_F \cdot F_{INV} + C_{wctot} \cdot IR_W$$

where $C_{F_2}$, $C_{F_3}$, $C_{INV}$ and are the tissue concentrations of fish for Trophic Levels 2 and 3, and benthic invertebrates, respectively, as defined in Section 8.2.4.1. $F_{F_2}$ (fraction of diet from Trophic Level 2 fish) is 0.05, $F_{F_3}$ (fraction of diet from Trophic Level 3 fish) is 0.89, and $F_{INV}$ (fraction of diet from benthic invertebrates) is 0.06. The great blue heron food ingestion rate ($IR_F$) and water ingestion rate ($IR_w$) are given in the receptor profiles in Section 8.1.3.3.

The aquatic food web (Figure 8-9) shows the prey types of the great blue heron.

The great blue heron tissue concentrations of ROPCs will be calculated by an equation adapted from EPA draft guidance (SLERAP Equation 5-13). *FCM*s ($FCM_4$ for Trophic Level 4) are used to account for bioaccumulation from ingested animal tissue:

$$C_{Heron} = C_{F_2} \cdot \frac{FCM_4}{FCM_2} \cdot F_{F_2} + C_{F_3} \cdot \frac{FCM_4}{FCM_3} \cdot F_{F_3} + C_{INV} \cdot \frac{FCM_4}{FCM_2} \cdot F_{INV} + C_{wctot} \cdot BCF_W \cdot CF$$

where:

$C_{Heron}$ = concentration of in receptor tissue (mg/kg or pCi/g)

$C_{F_3}$ = concentration or activity of COPC or ROPC in fish tissue for Trophic Level 3 prey type (omnivorous fish) (mg/kg or pCi/g)

$C_{F_2}$ = concentration or activity of COPC or ROPC in fish tissue for Trophic Level 2 prey type (planktivorous fish) (mg/kg or pCi/g)

$C_{INV}$ = concentration of in prey tissue (mg/kg or pCi/g)

$FCM_4$ = food-chain multiplier for the receptor (unitless)

$FCM_3$ = food-chain multipliers for ingested Trophic Level 3 prey type (omnivorous fish) (unitless)

$FCM_2$ = food-chain multipliers for ingested Trophic Level 2 prey type (planktivorous fish and invertebrates) (unitless)

$F_{F_3}$ = fraction of diet from Trophic Level 3 omnivorous fish (unitless)

$F_{F_2}$ = fraction of diet from Trophic Level 2 planktivorous fish (unitless)

$F_{INV}$ = fraction of diet from benthic invertebrate tissue (unitless)

$C_{sed}$ = concentration of constituent in ingested sediment (mg/kg or pCi/g)

$BCF_S$ = sediment-to-tissue uptake factor for receptor (mg/kg FW tissue per mg/kg DW sediment)

$C_{wctot}$ = concentration of constituent in unfiltered water from Columbia River maximum location (mg/L or pCi/L)

$BCF_W$ = water-to-tissue uptake factor for receptor (kg FW tissue per mg/L water)

$CF$ = conversion factor of 0.001 kg/g (to convert pCi/kg to pCi/g) for ROPCs

For ROPCs, the tissue concentration resulting from ingested water ($C_{wctot} \times BCF_W$) must be converted from pCi/kg to pCi/g by multiplying by 0.001 kg/g. Because the diet of the great blue heron is assumed to be exclusively fish and benthic invertebrates, $F_{F_3} + F_{F_2} + F_{INV} = 1$. As with other receptors, the contaminated proportion of prey is assumed to be 100 %, thus $P_A$ is dropped from SLERAP Equation 5-13, as presented above. The *FCM*s for the heron ($FCM_4$) and their planktivorous fish ($FCM_2$), omnivorous fish ($FCM_3$), and benthic invertebrate ($FCM_2$), and water-to-tissue bioconcentration factors ($BCF_W$) for the great blue heron are reported in Supplement 4.

**Carnivores: Bald Eagle and Mink (Trophic Level 4)**

Bald eagles and mink are carnivores that ingest omnivorous and piscivorous fish and other animals, but they also ingest water incidentally with their food (Figure 8-9). Ingestion of terrestrial prey at the Columbia River maximum site will not be evaluated because the concentrations of COPCs and ROPCs in soil near the Columbia River would be less than at the onsite ground maximum. Ingestion of terrestrial prey by red-tailed hawks will be evaluated at the onsite ground maximum, where concentrations in terrestrial prey will be higher than at the Columbia River. Because the exposure of hawks to terrestrial receptors at the onsite ground maximum is more conservative than exposure of predators to terrestrial receptors at the Columbia River, the SLERA will evaluate the exposure of mink assuming ingestion of only omnivorous fish (at Trophic Level 3) and water (Figure 8-9). However, since eagles are known to

consume salmon, their dose due to prey consumption will be based on Trophic Level 4 fish consumption. Thus:

$$DD = DD_A + DD_W, \text{ or} \qquad \text{(Equation 8-16)}$$

$$DD_{Eagle} = C_{F_4} \cdot IR_F \cdot F_A + C_{wctot} \cdot IR_W$$

$$DD_{Mink} = C_{F_3} \cdot IR_F \cdot F_A + C_{wctot} \cdot IR_W$$

where $DD_A$ and $DD_W$ are as given above, and $DD_A$ is calculated for omnivorous fish (Trophic Level 3) and carnivorous fish (Trophic Level 4) for the mink and eagle, respectively ($F_A$ is assumed to be 1). The eagle and mink food ingestion rates ($IR_F$) and water ingestion rate ($IR_w$) are given in the receptor profiles in Section 8.1.3.3. The aquatic food web (Figure 8-9) shows the prey types of the bald eagle and mink.

Bald eagle and mink tissue concentrations of ROPCs will be calculated by an equation adapted from EPA draft guidance (SLERAP Equation 5-13). *FCM*s ($FCM_3$ for Trophic Level 3) are used to account for bioaccumulation from ingested animal tissue:

$$C_{Eagle} = C_{F_4} \cdot \frac{FCM_4}{FCM_4} \cdot F_A + C_{wctot} \cdot BCF_W \cdot CF$$

$$C_{Mink} = C_{F_3} \cdot \frac{FCM_4}{FCM_3} \cdot F_A + C_{wctot} \cdot BCF_W \cdot CF$$

where:

| | | |
|---|---|---|
| $C_{Eagle/Mink}$ | = | concentration of receptor tissue (mg/kg or pCi/g) |
| $C_{F_4}$ | = | concentration of carnivorous fish (Trophic Level 4) tissue (mg/kg or pCi/g) |
| $C_{F_3}$ | = | concentration of omnivorous fish (Trophic Level 3)tissue (mg/kg or pCi/g) |
| $FCM_4$ | = | food-chain multipliers for the receptor (unitless) |
| $FCM_3$ | = | food-chain multipliers for ingested prey type (unitless) |
| $F_A$ | = | fraction of diet from omnivorous fish tissue (unitless) |
| $C_{wctot}$ | = | concentration of constituent in unfiltered water from Columbia River maximum location (mg/L or pCi/L) |
| $BCF_W$ | = | water-to-tissue uptake factor for receptor (kg FW tissue per mg/L water) |
| $CF$ | = | conversion factor of 0.001 kg/g (to convert pCi/kg to pCi/g) for ROPCs |

For ROPCs, the tissue concentration resulting from ingested water ($C_{wctot} \times BCF_W$) must be converted from pCi/kg to pCi/g by multiplying by 0.001 kg/g. Because the diet of the mink and the diet of the eagle are assumed to be exclusively fish, $F_A = 1$. For the mink, a realistic diet would require the addition of a term for ingestion of plants which would result in a less conservative estimate of tissue concentration. The *FCM*s for the eagle and mink ($FCM_4$) and their omnivorous prey ($FCM_3$), and water-to-tissue bioconcentration factors ($BCF_W$) for the bald eagle and mink, are reported in Supplement 4.

#### 8.2.4.4 External and Internal Radiation Dose

The total radiological dose to all receptors is calculated as the sum of the external and internal radiation doses.

$$DD = DD_E + DD_I \qquad \text{(Equation 8-17)}$$

where:

$DD$ = total radiation dose to the receptor (rad/day)
$DD_E$ = total external radiation dose (rad/day)
$DD_I$ = internal radiation dose (rad/day)

External doses to all aquatic biota and benthic invertebrates result from exposure to ROPCs in water and sediment. Wildlife receptors (Canada goose, sandpiper, heron, eagle, and mink) are exposed externally to ROPCs in air and water. The internal dose to plants and benthic invertebrates results from the uptake of radionuclides into their tissues from water and sediment. The internal dose to wildlife and fish receptors results from the uptake of radionuclides into their tissues from ingested food, water, and sediment. The fraction of time receptors spend immersed in sediment, on sediment, immersed in water, and in the proximity of water are scientific judgment based on narrative information about the receptors in published or internet wildlife biology articles, as given below.

**External Dose**

External radiation from water and sediment will be modeled as described by Blaylock, Frank, and O'Neal (1993). Radiation doses will be adjusted for the fraction of time that the receptors are assumed to be immersed in water away from sediment, or near enough to the water to receive external radiation (swimming on the surface or at the river bank), resting on sediment, and immersed in sediment. Those fractions were selected by scientific judgment based on narrative information about the receptors in published or internet wildlife articles. They are assumed to be:

| | Fraction of time on/near water surface | Fraction of time immersed in water | Fraction of time resting on sediment | Fraction of time immersed in sediment |
|---|---|---|---|---|
| Benthic invertebrates | 0 | 0.1 | 0 | 0.9 |
| Aquatic biota (including plants and salmonids) | 0 | 0.9 | 0.1 | 0 |
| Canada goose | 0.5 | 0 | 0 | 0 |
| Spotted sandpiper | 0.5 | 0 | 0 | 0 |
| Great blue heron | 0.5 | 0 | 0 | 0 |
| Bald eagle | 0.05 | 0 | 0 | 0 |
| Mink | 0.2 | 0 | 0 | 0 |

The birds and mink will also be assumed to receive external radiation from air. Note that it is assumed the Canada goose, spotted sandpiper, and great blue heron spend 50 % of the time sufficiently away from water and sediment such that there is negligible exposure due to radionuclides in the water and sediment. Likewise, the mink is assumed to spend 80 % of the time away from water and sediment, and the bald eagle spends 95 % of the time in flight or perched such that there is negligible exposure due to radionuclides in the water and sediment.

As presented in Section 8.2.3.4, the external dose (rad/day) to all receptors for a given radionuclide will be calculated as:

$$DD_E = \sum(\text{ROPC Concentration} \cdot \text{Dose Conversion Factor})$$

where $DD_E$ is the external radiation dose (rad/day). The external doses (rad/day) to all aquatic receptors from water, sediment, and air will be calculated, respectively, as follows:

$$DD_E = DD_{E_{water}} + DD_{E_{sed}} + DD_{E_{air}}$$

where:

$DD_E$ = total external radiation dose (rad/day)

$DD_{E_{water}}$ = external radiation dose from immersion in water (rad/day)

$DD_{E_{sed}}$ = external radiation dose from sediment (rad/day)

$DD_{E_{air}}$ = external radiation dose from air (rad/day)

The external dose from water exposure includes both immersion and time spent on or near the water surface. The external daily dose due to water proximity and contact is:

$$DD_{E_{water}} = DD_{near} + DD_{imm}$$

where:

$DD_{E_{water}}$ = external radiation dose from water (rad/day)

$DD_{near}$ = external dose from exposure on or near water (rad/day)

$DD_{imm}$ = external dose from exposure due to water immersion (rad/day)

Receptors immersed in water will be exposed to beta and gamma radiation. Receptors on the surface or in direct proximity to water will receive exposure to gamma radiation. Alpha radiation (for both near water and immersion exposures) and beta radiation (for near water exposures) are not assumed to contribute to the external dose factor because they do not penetrate enough to cause exposure. For example, Sr-90 and Sr-92 both have beta radiation with an energy of 0.196 MeV; Sr-90 has no gamma radiation, whereas Sr-92 has gamma radiation. The *DCF* associated with a 15-cm thick soil source for Sr-90 is 3.72E-21 Sv/s per Bq/m$^3$, whereas the *DCF* for Sr-92, under the same scenario, is 3.88E-17 Sv/s per Bq/m$^3$. In this case, the external beta radiation causes only about 1/10,000 of the dose. Similarly, for

alpha radiation, Sm-146, Sm-147, Gd-148, and Gd-152 have alpha radiation but neither beta nor gamma, thus their *DCF*s are 0.

The external dose for water immersion ( $DD_{imm}$ ) and near or surface water contact ($DD_{near}$) is derived from Blaylock, Frank, and O'Neal (1993) and is calculated as:

$$DD_{near} = F_{near} \cdot E_{\gamma} \cdot \left(1 - AbF_{\gamma}\right) \cdot CFa \cdot CF \cdot C_{wctot}$$ (Blaylock et. al. (1993), Eq. 2)

$$DD_{imm} = F_{imm} \cdot \left[E_{\beta} \cdot \left(1 - AbF_{\beta}\right) + E_{\gamma} \cdot \left(1 - AbF_{\gamma}\right)\right] \cdot CFa \cdot CF \cdot C_{wctot}$$ (Blaylock et. al. (1993), Eqs. 2 and 6)

Factoring $C_{cwtot}$, substituting variables to represent the dose factor yields the following equations:

$$DD_{E_{water}} = \left(F_{near} \cdot E_{\gamma} \cdot \left(1 - AbF_{\gamma}\right) \cdot CFa \cdot CF + F_{imm} \cdot \left[E_{\beta} \cdot \left(1 - AbF_{\beta}\right) + E_{\gamma} \cdot \left(1 - AbF_{\gamma}\right)\right] \cdot CFa \cdot CF\right) \cdot C_{wctot}$$ , or

$$DD_{E_{water}} = \left(DCF_{near} + DCF_{imm}\right) \cdot C_{wctot}$$ , or

$$DD_{E_{water}} = DCF_{water} \cdot C_{wctot}$$

Thus, the external dose to an aquatic receptor due to water exposure to a ROPC will be calculated according to:

$$DD_{E_{water}} = DCF_{water} \cdot C_{wctot}$$ , and

$$DF_{water} = \left[\left(F_{near} + F_{imm}\right) \cdot E_{\gamma} \cdot \left(1 - AbF_{\gamma}\right) + F_{imm} \cdot E_{\beta} \cdot \left(1 - AbF_{\beta}\right)\right] \cdot CFa \cdot CF$$

where:

$DD_{E_{water}}$ = external radiation dose from water (rad/day)

$DCF_{water}$ = factor for converting activity of radionuclide in water to external dose from exposure to water in units of rad/day per pCi/g ($DCF_{water} = DCF_{near} + DCF_{imm}$)

$DCF_{near}$ = factor for converting activity of radionuclide in water to external dose from exposure near water, or due to surface water in units of rad/day per pCi/g ( $DCF_{near} = F_{near} \cdot E_{\gamma} \cdot \left(1 - AbF_{\gamma}\right) \cdot CFa \cdot CF$ )

$DCF_{imm}$ = factor for converting activity of radionuclide in water to external dose from immersion water, in units of rad/day per pCi/g ( $DCF_{imm} = F_{imm} \cdot \left[E_{\beta} \cdot \left(1 - AbF_{\beta}\right) + E_{\gamma} \cdot \left(1 - AbF_{\gamma}\right)\right] \cdot CFa \cdot CF$ )

$F_{imm}$ = fraction of time receptor spends immersed in water (unitless)

$F_{near}$ = fraction of time receptor spends near or swimming on the surface of the water (unitless)

$AbF_{\beta}$ = absorbed fraction of energy from beta energy $E_{\beta}$ (unitless)

$E_{\beta}$ = average energy emitted as beta radiation × proportion of disintegrations producing a beta-particle (MeV per disintegration)

$AbF_{\gamma}$ = absorbed fraction of energy from gamma energy $E_{\gamma}$ (unitless)

| | | |
|---|---|---|
| $E_\gamma$ | = | photon energy emitted during transition from a higher to a lower energy state × proportion of disintegrations producing gamma radiation (MeV/disintegration) |
| $CFa$ | = | unit conversion factor, $5.12 \times 10^{-5}$ rad/day per pCi/g per MeV/disintegration |
| $CF$ | = | factor to convert L to g (0.001 L/ml × 1 ml/g = 0.001 L/g) |
| $C_{wctot}$ | = | concentration of constituent in unfiltered water from Columbia River maximum location (mg/L or pCi/L) |

$AbF_\gamma$ and $E_\gamma$ for each ROPC are reported in Supplement 4 for all ROPCs. To calculate external exposure to all aquatic receptors from ROPCs in water, $DCF_{water}$ values will be multiplied by the modeled total activities of the corresponding radionuclides in surface water at the Columbia River maximum location. The external dose from sediment exposure includes both exposure to the water/sediment interface, and time spent buried in the sediment. The external daily dose due to sediment immersion and contact is:

$$DD_{E_{sed}} = DD_{s/w} + DD_{sed,imm}$$

where:

| | | |
|---|---|---|
| $DD_{E_{sed}}$ | = | external radiation dose from soil (rad/day) |
| $DD_{s/w}$ | = | external dose from exposure on sediment/water interface (rad/day) |
| $DD_{sed,imm}$ | = | external dose from exposure due to immersion in sediment (rad/day) |

Following the logic of Blaylock, Frank, and O'Neal (1993), the external dose for sediment contact ($DD_{s/w}$) and sediment immersion ($DD_{sed,imm}$) will be calculated as:

$$DD_{s/w} = 0.5F_{s/w} \cdot \left[E_\beta \cdot \left(1 - AbF_\beta\right) + E_\gamma \cdot \left(1 - AbF_\gamma\right)\right] \cdot CFa \cdot C_{sed}$$ (Blaylock et. al. (1993), Eqs. 3 and 7)

$$DD_{sed,imm} = F_{sed,imm} \cdot \left[E_\beta \cdot \left(1 - AbF_\beta\right) + E_\gamma \cdot \left(1 - AbF_\gamma\right)\right] \cdot CFa \cdot C_{sed}$$ (Blaylock et. al. (1993), Eqs. 2 and 6)

Factoring $C_{sed}$, substituting variables to represent the dose factor yields the following equations:

$$DD_{E_{sed}} = \begin{pmatrix} 0.5F_{s/w} \cdot \left[E_\beta \cdot \left(1 - AbF_\beta\right) + E_\gamma \cdot \left(1 - AbF_\gamma\right)\right] \cdot CFa + \\ F_{sed,imm} \cdot \left[E_\beta \cdot \left(1 - AbF_\beta\right) + E_\gamma \cdot \left(1 - AbF_\gamma\right)\right] \cdot CFa \end{pmatrix} \cdot C_{sed} \text{ or}$$

$$DD_{E_{sed}} = \left(DCF_{s/w} + DCF_{sed,imm}\right) \cdot C_{sed}\text{, or}$$

$$DD_{E_{sed}} = DCF_{sed} \cdot C_{sed}$$

Thus, the external dose to an aquatic receptor due to sediment exposure to a ROPC will be calculated according to:

$$DD_{E_{sed}} = DCF_{sed} \cdot C_{sed}\text{, and}$$

$$DCF_{sed} = \left(0.5F_{s/w} + F_{sed,imm}\right) \cdot \left[E_\beta \cdot \left(1 - AbF_\beta\right) + E_\gamma \cdot \left(1 - AbF_\gamma\right)\right] \cdot CFa \cdot C_{sed}$$

where:

| | | |
|---|---|---|
| $DD_{E_{sed}}$ | = | external radiation dose from soil (rad/day) |
| $DCF_{sed}$ | = | factor for converting activity of radionuclide in sediment contact and immersion to external dose from exposure to the sediment in units of rad/day per pCi/g ($DCF_{sed} = DCF_{s/w} + DCF_{sed,imm}$) |
| $DCF_{s/w}$ | = | factor for converting activity of radionuclide in sediment/water interface to external dose from exposure to the sediment/water interface in units of rad/day per pCi/g ($DCF_{s/w} = 0.5F_{s/w} \cdot \left[E_\beta \cdot \left(1 - AbF_\beta\right) + E_\gamma \cdot \left(1 - AbF_\gamma\right)\right] \cdot CFa$) |
| $DCF_{sed,imm}$ | = | factor for converting activity of radionuclide in sediment to external dose from exposure due to burial in sediment, in units of rad/day per pCi/g ($DCF_{sed,imm} = F_{sed,imm} \cdot \left[E_\beta \cdot \left(1 - AbF_\beta\right) + E_\gamma \cdot \left(1 - AbF_\gamma\right)\right] \cdot CFa$) |
| 0.5 | = | factor to account 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 (unitless) |
| $F_{s/w}$ | = | fraction of time receptor spends at the sediment-water interface (unitless) |
| $F_{sed}$ | = | fraction of time receptor spends buried in sediment (unitless) |
| $E_\beta$ | = | average energy emitted as beta radiation × proportion of disintegrations producing a β-particle (MeV per disintegration) |
| $AbF_\beta$ | = | absorbed fraction of energy from beta energy $E_\beta$ (unitless) |
| $E_\gamma$ | = | photon energy emitted during transition from a higher to a lower energy state (MeV) × proportion of disintegrations producing γ radiation (MeV/disintegration) |
| $AbF_\gamma$ | = | absorbed fraction of energy from gamma energy $E_\gamma$ (unitless) |
| $CFa$ | = | unit conversion factor, $5.12 \times 10^{-5}$ rad/day per pCi/g per MeV/disintegration |

Immersion in sediment exposes receptors to a static, direct-contact interface with the contaminated media. Accordingly, a portion of the external dose while immersed in sediment can be attributed to radionuclide daughter products. Because immersion air and river water are not static (i.e., the air and water are in continuous motion), the impact of daughter products is assumed to be insignificant due to limited non-static contact. Terrestrial exposure due to soil contact is not considered a direct-contact interface (that is, continuous and complete immersion of the receptor is disrupted by air and vegetation because of the nature of burrows and dens). Therefore, the external dose due to daughter products in soil is assumed to be negligible.

The exposures of aquatic receptors to ROPCs in sediment is calculated by assuming that the decay products of all short-lived ROPCs in sediment are in secular equilibrium. The activities of each of the daughter radionuclides are, therefore, equal to the activity of the parent multiplied by the fraction of the decays in the immediately preceding generation that yield the daughter radionuclide.

$$DCF_D = DCF_{parent} \cdot y$$

where

$DCF_D$ = the dose factor of the daughter product

$DCF_{parent}$ = the dose factor of the parent isotope

$y$ = yield of the daughter product from the decay of the parent isotope (percent)

For example, the activities of radium-225, actinium-225, francium-221, astatine-217, and bismuth-213 are assumed to be equal to the activity of their parent, thorium-229. However, when bismuth-213 decays, 97.8 % of the decays yield polonium-213 and 2.2 % of the decays yield thallium-209. Therefore, the activities of polonium-213 and thallium-209 are assumed to be 97.8 % and 2.2 %, respectively, of the activity of thorium-229. Exposure factors for the daughter radionuclides were used to calculate the summed exposures from the ultimate parent and all of the daughter radionuclides for both external and internal radiation from exposure to sediments; thus, for an ROPC that undergoes decay:

$$DCF_{sed+D} = DCF_{sed_{parent}} + \sum_{i=1}^{n} DCF_{sed_D}$$

where

$DCF_{sed+D}$ = factor for converting activity of radionuclide (and its daughter products) to external dose from exposure to the sediment in units of rad/day per pCi/g

$DCF_{sed_{parent}}$ = factor for converting activity of the parent radionuclide to external dose from exposure to the sediment in units of rad/day per pCi/g

$\sum_{i=1}^{n} DCF_{sed_D}$ = sum of the daughter product dose factors for each applicable daughter product (1 through $n$) for converting activity of the daughter product to external dose from exposure to the sediment in units of rad/day per pCi/g

The dose factor for each daughter product is calculated using the energy ($E$) and absorption factor ($AbF$) appropriate to the daughter product in accordance with the equations above. The fraction of time a receptor spends immersed in water, near the water, or swimming on the surface of the water must also be applied when computing the contribution of daughter products to a ROPC dose factor.

Values of $F_{s/w}$ and $F_{sed}$ are reported in Supplement 4 along with $AbF_\beta$, $E_\beta$, $AbF_\gamma$, and $E_\gamma$ for all ROPCs and their daughters. To calculate external exposure to all aquatic receptors from ROPCs in water and sediment, $DCF_{water}$ and $DCF_{sed}$, values will be multiplied by the modeled activities of the corresponding radionuclides in surface water and sediment at the Columbia River maximum location.

Per EPA (1993c or Eckerman and Ryman 1993), the external dose (rad/day) to all wildlife receptors from air will be calculated as:

$$DCD_{E_{air}} = DCF_{air} \cdot C_a$$

where:

$DD_{E_{air}}$ = external radiation dose from air (rad/day)

$C_a$ = activity of the ROPC in air, calculated as described in Section 6.1 (pCi/m$^3$)

$DCF_{air}$ = factor for converting activity of the ROPC in air to external dose from air (rad/day per pCi/m$^3$

Note: This equation first appears in Section 8.2.3.4.

The external dose conversion factor for air ($DCF_{air}$) will be calculated as follows:

$$DCF_{air} = 3.2 \times 10^5 \cdot DCF$$

where:

$3.2 \times 10^5$ = conversion factor to convert Sv/s per Bq/m$^3$ to rad/day per pCi/m$^3$ (Eckerman and Ryman 1993)

$DCF$ = dose conversion factor for external radiation from immersion in air (Sv/s per Bq/m$^3$, Eckerman and Ryman 1993)

Note: The equation for $DCF_{air}$ first appears in Section 8.2.3.4.

For all ROPCs, values of *DCF* for air are reported in Supplement 4. To calculate external exposure to all aquatic receptors from ROPCs in air, *DCF* values will be multiplied by the modeled activities of the corresponding radionuclides in air at the Columbia River maximum location.

**Internal Dose**

The internal exposure to radionuclides is calculated from the activity in tissues rather than from the daily ingestion. The internal activities of ROPCs are calculated by using *BCF*s and ROPC activities in sediment, food, and water (see Section 8.2.4.3). Internal radiation doses are calculated by multiplying the activity in tissues by the sum of alpha, beta, and gamma decay energies, where alpha and beta energies are assumed to be completely absorbed. Because gamma rays, like X-rays, may pass through the tissues without depositing their energy, gamma energies are adjusted to account for greater absorption by larger organisms at a given energy level and for greater absorption by all receptors at lower energy levels. For radionuclides in sediment, radiation by daughter radionuclides was also included in internal dose calculations. Daughter radionuclides are produced by decay of parent radionuclides and short-lived radionuclides could be expected to contribute to the receptor's internal dose. As previously discussed, exposures are calculated by assuming that the decay products of all short-lived ROPCs are in secular equilibrium and equal to the activity of the parent multiplied by the fraction of the decays in the immediately preceding generation that yielded the daughter radionuclide. Decay energies (Eckerman and Ryman 1993) and absorption fractions for gamma radiation (Blaylock, Frank, and O'Neal 1993; Sample et al. 1997) of daughter radionuclides are reported in Supplement 4.

Adapting the equations of Sample et al. (1997), the internal dose (rad/day) to aquatic receptors and wildlife receptors will be calculated as follows:

$$DD_I = \sum QF \cdot C_{AP \text{ or } INV \text{ or } A} \cdot E \cdot CF \cdot AbF \qquad \text{(modified Sample et al. (1997), Eq. 11)}$$

where

| | | |
|---|---|---|
| $DD_I$ | = | internal radiation dose (rad/day) |
| $QF$ | = | quality factor for relative biological effect of radiation (unitless) (Kocher and Trabalka 2000) |
| $QF_\alpha$ | = | 10 for alpha radiation |
| $QF_\beta$ | = | 1 for beta radiation |
| $QF_\gamma$ | = | 1 for beta radiation |
| $C_{AP \text{ or } INV \text{ or } A}$ | = | activity of radionuclide in receptor tissue (aquatic plant, benthic invertebrate, or animal - see Section 8.2.4.3) (pCi/g) |
| $CF$ | = | unit conversion factor, $5.12 \times 10^{-5}$ rad/day per pCi/g per MeV/disintegration |
| $E$ | = | average energy emitted as alpha, beta, or gamma radiation by radionuclide $i$ × proportion of disintegrations producing alpha, beta, or gamma radiation (MeV per disintegration) |
| $AbF$ | = | absorbed fraction of energy from alpha, beta, or gamma energy (unitless) |

The equation above can be expanded to show the variables specific to the types of radiation emitted from the ROPC.

$$DD_I = \left(QF_\alpha \cdot E_\alpha \cdot AbF_\alpha + QF_\beta \cdot E_\beta \cdot AbF_\beta + QF_\gamma \cdot E_\gamma \cdot AbF_\gamma\right) \cdot CF \cdot C_{AP \text{ or } INV \text{ or } A}$$

Substituting variables to represent the dose factor yields:

$$DD_I = DCF_I \cdot C_{AP \text{ or } INV \text{ or } A}, \text{ and}$$

$$DF_I = \left(QF_\alpha \cdot E_\alpha \cdot AbF_\alpha + QF_\beta \cdot E_\beta \cdot AbF_\beta + QF_\gamma \cdot E_\gamma \cdot AbF_\gamma\right) \cdot CF$$

where $QF_\alpha$, $QF_\beta$, and $QF_\gamma$ are the quality factors of alpha, beta, and gamma radiation (respectively), $E_\alpha$, $E_\beta$, and $E_\gamma$ are the product of the average energy emitted and proportion of disintegrations producing alpha, beta, and gamma radiation (respectively), and $AbF_\alpha$, $AbF_\beta$, and $AbF_\gamma$ are absorbed fraction of energy from alpha, beta, and gamma energy (respectively). Other terms are as defined above. Note that these equations first appear in Section 8.2.3.4. As with sediment exposures, internal exposure dose factors for aquatic receptors include the contribution of daughter products:

$$DCF_{I+D} = DCF_{I_{parent}} + \sum_{i=1}^{n} DCF_{I_D}$$

where

$DCF_{I+D}$ = factor for converting activity of radionuclide (and its daughter products) to internal dose from ingestion of contaminated food in units of rad/day per pCi/g

$DCF_{I_{parent}}$ = factor for converting activity of the parent radionuclide to internal dose from exposure in units of rad/day per pCi/g

$\sum_{i=1}^{n} DCF_{I_D}$ = sum of the daughter product dose factors for each applicable daughter product (1 through *n*) for converting activity of the daughter product to internal dose in units of rad/day per pCi/g

Values of *E* and *AbF* for each radiation (alpha, beta and gamma) are reported in Supplement 4 for all ROPCs and their daughters, respectively. To calculate internal exposure to all aquatic receptors from ingested ROPCs, $DCF_I$ values will be multiplied by the modeled activities of the corresponding radionuclides in receptor tissues at the Columbia River maximum location.

### 8.2.5 Exposure Variables

The magnitude of exposure of ecological receptors to COPCs and ROPCs in environmental media depends on various parameters and variables in the above exposure equations. These variables are discussed in this section. The exposure variables include space and time factors correcting for the fraction of a receptor's total exposure that can originate at the exposure location (Section 8.2.5.1), variables determining the rate of ingestion and absorption of COPCs and ROPCs (Section 8.2.5.2), and factors accounting for the accumulation in tissues of substances present in exposure media or food (Section 8.2.5.3). The exposure variables for ecological receptors are briefly discussed below.

#### 8.2.5.1 Space and Time Factors for Exposure Calculations

For wildlife receptors that are exposed to COPCs and ROPCs by ingestion and inhalation, the calculation of exposure requires exposure factors that quantify the fraction of a receptor's exposure obtained from the contaminated site. A receptor may obtain only a fraction of its exposure to a contaminant from the exposure location as a result of the receptor foraging over an area larger than the exposure location or spending only a fraction of its lifetime at the exposure location, or both. The exposure assumptions for use and derivation of area-use and temporal-use factors follow.

**Area-Use Factor**

The area-use factor (*AUF*) estimates the fraction of a receptor's exposure that comes from the exposure location. The *AUF* is the smaller of 1 and the ratio of the area of the exposure location and the area in which a receptor lives or forages, whichever is more appropriate to the routes by which the receptor is exposed. The *AUF* is calculated as follows:

$$AUF = 1, \text{ if } A \geq HR$$

$$AUF = A/HR, \text{ if } A < HR$$

where:

$AUF$ = area use factor (unitless)

$A$ = area of exposure (ha)

$HR$ = home range of the receptor (ha)

For the SLERA, the *AUF* will be assumed to be 1 for all receptors. This assumption is highly conservative for wide-ranging receptors such as mule deer, coyote, red-tailed hawk, and bald eagle.

**Temporal-Use Factor**

There are several approaches to dealing with the temporal aspect of exposure. The first approach is to assume, conservatively, that receptors are exposed throughout their lifetime to COPCs and ROPCs present at the exposure location. The second approach is to estimate the temporal-use factor (*TUF*) as the fraction of time each year that a receptor is in the vicinity of the exposure location during which it forages or resides at the exposure location. The remaining time is assumed to be spent in an area free of contamination from the source being evaluated.

For the SLERA, the *TUF* will be assumed to be 1 for all receptors. This assumption is appropriate because some species at each trophic level are nonmigratory year-round residents, even if the specific receptor species evaluated may not be.

### 8.2.5.2 Uptake Variables

The exposure equations for ecological receptors include parameters for body weight, the ingestion rate, and dietary distribution of ecological receptors and the efficiency of absorption of COPCs and ROPCs from ingested media. Where possible, data was taken from published sources (especially EPA 1993a, 1999) and Sample et al. (1997). In some cases, measured values were not available. However, EPA 1993a provides allometric equations that allow various intake parameters to be calculated from the receptor's body weight. These equations were derived by fitting curves to the measured parameters for animals with various body weights but with similar metabolic characteristics. Such parameters as total food ingestion and water ingestion depend on the caloric requirements and metabolic rate of the receptor, both of which are related to body weight. Allometric equations were used to calculate the water ingestion rates for the mourning dove and western meadowlark and the total food ingestion rates for the mourning dove and spotted sandpiper. Source data included allometric calculations of the food ingestion rate of the great blue heron and water ingestion rates of coyote, red-tailed hawk, Canada goose, spotted sandpiper, great blue heron, bald eagle, and mink. Uptake variables are shown in the receptor parameter descriptions in Section 8.1.3.3.

**Ingestion Rates**

The magnitude of exposure of ecological receptors to COPCs and ROPCs in environmental media depends on the rate of intake of the contaminated media. For wildlife receptors exposed by ingestion, receptor-specific ingestion rates are required to estimate exposure. Published values for food ingestion ($IR_F$), soil and sediment fraction (*SFr*), and water ingestion ($IR_W$) will be used to estimate exposure.

**Absorption Efficiency**

Substances ingested or inhaled by ecological receptors are absorbed and taken up into the receptor's cells and organs to varying degrees. The efficiency of absorption depends on the relative affinity of the substance for the environmental medium (soil, particulate, sediment, water, and tissue) and on the relative affinity of the substance for the receptor's tissues. For both the PRA and the FRA, the absorption efficiency (*AE*) for ingested media will be assumed to be the same as or 100 % of the actual absorption of the contaminant in the experiment or field observation used to derive the TRV. Therefore, *AE* does not

appear in the exposure equations. This assumption is conservative for COPCs and ROPCs ingested as soil, sediment, or particulates in water.

#### 8.2.5.3 Bioconcentration Factors for Calculating Terrestrial Exposures

The calculation of exposure for ecological receptors may require one or more bioaccumulation and transfer factors. These factors are used to estimate the concentration in the tissue of an organism from the concentrations in the contaminated media to which it is exposed. Such factors are required to estimate exposure for wildlife receptors, such as mammals and birds, that are exposed to COPCs and ROPCs in soil or water by ingestion of soil, water, plants, or soil-dwelling invertebrates or other wildlife when the concentration in the ingested organism is not measured directly (Figure 8-12). In each case, the numerator of the factor must have units corresponding to the units of concentration in the medium taking up the substance (tissue), and the denominator must have units corresponding to the units of concentration in the "source" medium (soil, water, or tissue). The rules for use and derivation of bioaccumulation or transfer factors follow:

**Direct Deposition-to-Plant Tissue Transfer Factor**

The uptake of COPCs and ROPCs by direct deposition to leaf surfaces, including transfer factors, is discussed in Section 6.6.1.

**Air-to-Plant Tissue Transfer Factor**

The uptake of COPCs in vapor, including transfer factors, is discussed in Section 6.6.2.

**Soil-to-Plant Bioconcentration Factor**

The concentration in aboveground portions of plants through root uptake from soil is a function of the COPC- and ROPC-specific soil concentration (see Section 8.2.3.1) and COPC- or ROPC-specific plant *BCF*. The *BCF* is the ratio of the COPC or ROPC concentration in plant tissue to the COPC or ROPC concentration in soil. The *BCF* will be used to estimate the tissue concentration in plants exposed to COPCs and ROPCs in soil from the concentration of COPC or ROPC in bulk soil. The exposure evaluation will consider three kinds of *BCF*: measured or empirically derived values, mass-limited values, and bioaccumulation equivalency factors (*BEF*s), as well as methods to calculate concentrations of carbon-14 and tritium in plants.

**Measured or Empirically Derived Values**. When measured or empirically derived *BCF*s are used, the concentrations of COPCs and ROPCs from soil in plant tissue are proportional to the concentrations in soil. That is:

$$BCF_r = \frac{C_{TP}}{C_M} \qquad \text{(SLERAP Eq. 5-2)}$$

where:

$BCF_r$ = bioconcentration factor (unitless [soil, sediment], or L/kg [water])

$C_{TP}$ = constituent concentration in plant (mg/kg or pCi/g)

$C_M$ = constituent concentration in media (mg/kg or pCi/g [soil, sediment], or mg/L or pCi/L [water])

Concentrations are estimated for plant tissues that are fed upon by wildlife receptors.

Values of $BCF_r$ are reported in Supplement 4. The first choice for $BCF_r$ values will be EPA (1999) values, and values developed using EPA methods. Per EPA draft guidance (1999), values of $BCF_r$ for organic COPCs for which no field or laboratory data is available are estimated using the Travis and Arms (1988) regression on $K_{ow}$:

$$\log BCF_r = 1.588 - (0.578 \times \log K_{ow}) \qquad \text{(SLERAP Eq. C-1-2)}$$

$K_{ow}$ and log $K_{ow}$ values are reported in Supplement 4. Where $K_{ow}$ values were not available, they were calculated by using EPA's EPI Suite ™ software, which calculates physical properties of organic chemicals from structure/activity relationships.

Travis and Arms (1988) measured soil-to-plant uptake values for 29 organic chemicals (primarily pesticides) to establish a linear relationship between these two parameters. The equations used to calculate *BCF*s rely on empirical data from a few chemicals, plants, and growing media to extrapolate to all other organic chemicals and growing situations. As noted by EPA (1999), this regression equation, derived from experiments conducted on three classes of compounds (pesticides, PCDDs, and PCBs), may not accurately represent the behavior of all organic COPCs under site-specific conditions, and further research is needed to evaluate the applicability and limitations associated with the use of this equation for all classes of compounds.

Per EPA draft guidance (1999), recommended $BCF_r$ values for inorganic elements are values published in Baes et al. (1984), Cappon (1981), and EPA (1992, 2005). For inorganic COPCs and ROPCs with no published measured or estimated data, the arithmetic average of the available $BCF_r$ values for the other inorganics will be used as the *BCF* (EPA 1999).

**Mass-Limited Values**. In some cases, Equation C-1-2 in the SLERAP (Travis and Arms 1988) predicts the accumulation in plants of more organic COPC than is deposited on the soil (see Section 6.6.3.3 for a detailed discussion). Mass-limited *BCF* caps were derived for organic COPCs by (1) assuming that all of the COPC emitted from the WTP and deposited on the soil is taken up by the plants, (2) calculating the concentration of COPC in all of the plants in 1 $m^2$, and (3) dividing that concentration by the concentration of COPC in the soil if all of the COPC is mixed in the rooting zone, which is the upper 15 cm.

The maximum possible uptake factor is calculated as shown in the following equations:

$$\text{Maximum possible bioconcentration factor} = \frac{\text{Total COPC deposited (mg/m}^2\text{) / Plant mass density (kg/m}^2\text{)}}{\text{Total COPC deposited (mg/m}^2\text{) / Soil density (kg/m}^2\text{)}}$$

This equation can be reduced to:

$$\text{Maximum possible bioconcentration factor} = \frac{\text{Soil density (kg soil/m}^2\text{)}}{\text{Plant mass density (kg plant/m}^2\text{)}} \qquad \text{(Equation 8-19)}$$

The mass of soil per $m^2$ is 1300 kg/$m^3$ × 15 cm = 195 kg/$m^2$. The mass of plants used as food for herbivores is assumed to be the yield of forage, which is 0.15 kg/$m^2$. A plant yield value 0.15 kg/$m^2$ for forage was derived from a value of 1,500 kg/ha dry yield for Richland, Washington (Wisiol 1984, refer to

Table 6-5). Therefore, the mass-limited $BCF_r$ is 195/0.15 = 1300. All of the published or calculated values of $BCF_r$ presented in Supplement 4 are less than that upper limit, so the mass-limited $BCF_r$ was not used.

**Bioconcentration Equivalency Factors**. The EPA recommends using *BEF*s to estimate the bioconcentration of PCDD and PCDF congeners for which field or laboratory measurements are not available. The BEF is the predicted ratio of bioaccumulation of a PCDD or PCDF congener in soil to the bioaccumulation of 2,3,7,8-tetrachlorodioxin (TCDD) (EPA 1999). *BEF*s were used by EPA (1999) to calculate the values for *BCF* presented in Supplement 4 for PCDD and PCDF congeners.

$$BCF_i = BCF_{TCDD} \cdot BEF_i \qquad \text{(SLERAP Eq. 2-6)}$$

where:

$BCF_i$ = media-to-animal or media-to-plant bioconcentration factor for $i^{th}$ congener (L/kg [water], unitless [soil and sediment])

$BCF_{TCDD}$ = media-to-receptor *BCF* for 2,3,7,8-TCDD (L/kg [aquatic receptor], unitless [soil and sediment receptor])

$BEF_i$ = bioaccumulation equivalency factor for $i^{th}$ congener (unitless)

**Carbon-14 and Tritium**. *BCF*s are used for all ROPCs except carbon-14 and tritium. Exposure calculations for most ROPCs are based on the assumption that radionuclides are present as particulates or vapors. However, special consideration must be given to carbon-14 and tritium (hydrogen-3), as these ROPCs are processed by vegetation with natural carbon and hydrogen, respectively. Thus, the vegetation pathways for carbon-14 and tritium are dependent on the exchange of carbon and hydrogen between plants and the environment. For this assessment, guidance from Regulatory Guide 1.109 (NRC 1977) is used to account for the bioaccumulation of carbon-14 and tritium in plants. This is done through the use of correction factors, along with the assumption that all carbon-14 is released by the WTP in oxide form (CO or $CO_2$) and tritium is released as water vapor. These correction factors are applied to the air concentration (e.g., pCi/m$^3$) estimated at the point of exposure by the air model.

The concentration of carbon-14 in vegetation is calculated assuming that its ratio to the natural carbon in vegetation is equal to the ratio of carbon-14 to natural carbon in the atmosphere surrounding the vegetation as described in Section 6.6.2.

The concentration of tritium in vegetation is based on the equilibrium between moisture in the air and water in plants as described in Section 6.6.2.

Soil-to-plant uptake values are also used for aboveground protected and unprotected plant parts for human health exposure (Section 6.6.3).

**Soil-to-Terrestrial Invertebrate Bioconcentration Factor**

The concentration in terrestrial invertebrates through uptake from soil is a function of the COPC- or ROPC-specific soil concentration (see Section 8.2.3.1) and COPC- or ROPC-specific invertebrate $BCF_S$. The $BCF_S$ is the ratio of the COPC or ROPC concentration in invertebrate tissue to the COPC or ROPC concentration in soil. The $BCF_S$ will be used to estimate the tissue concentration in invertebrates exposed to COPCs and ROPCs in soil from the concentration of COPC or ROPC in bulk soil. The exposure

evaluation will consider two versions of $BCF_S$: measured or empirically determined values and mass-limited values.

**Measured or Empirically Determined Values**. The soil-to-terrestrial invertebrate tissue transfer factor ($BCF_S$) is the ratio of the COPC or ROPC concentration in terrestrial invertebrate tissue to the COPC or ROPC concentration in soil [(mg/kg$_{tissue}$ wet wt)/(mg/kg$_{soil}$ dry wt)]. The $BCF_S$ is used to estimate the tissue concentration of terrestrial invertebrates exposed to COPCs and ROPCs in soil by all exposure routes (ingestion, direct contact, and inhalation) from the concentration of a COPC or ROPC in bulk soil. That is:

$$BCF_{INV} = \frac{C_{INV}}{C_M} \qquad \text{(SLERAP Eq. 5-2)}$$

where:

$BCF_{INV}$ = bioconcentration factor (unitless [soil, sediment], or L/kg [water])
$C_{INV}$ = constituent concentration in the terrestrial invertebrate (mg/kg or pCi/g)
$C_M$ = constituent concentration in media (mg/kg or pCi/g [soil, sediment], or mg/L or pCi/L [water])

$BCF_{INV}$ is used for soil-dwelling invertebrates, such as worms or insects, that are an important diet item of many omnivores, such as pocket mice and meadowlarks. Tissue concentrations will be estimated for terrestrial invertebrates that are fed upon by wildlife receptors. Although the habitat at most of the Hanford Site is not favorable to earthworms, earthworms are used as a representative of soil invertebrates because most of the data about soil invertebrates pertain to earthworms. This is consistent with EPA draft guidance (EPA 1999), which uses measured uptake factors for earthworms to represent all soil invertebrates.

The first choice for terrestrial soil-to-soil invertebrate bioaccumulation ($BCF_{INV}$) values will be field or laboratory values and calculated values for earthworms reported by EPA (1999). Per EPA draft guidance (1999), recommended $BCF_{INV}$ values for inorganic elements with no published field or laboratory data is arithmetic averages of the $BCF_{INV}$ values available for other inorganics. For organic compounds with no field or laboratory data, $BCF_{INV}$ values will be calculated with a regression equation described by EPA (1999):

$$\log BCF_{INV} = 0.819 \times \log K_{ow} - 1.146 \qquad \text{(SLERAP Eq. C-1-1)}$$

This equation uses values derived from $K_{ow}$s and uptake by daphnids, an aquatic macroinvertebrate, exposed to polycyclic aromatic hydrocarbons (PAHs) (Southworth et al. 1978). Where no appropriate published surrogate data is available, no default $BCF_{INV}$ for organic compounds is used.

The $BCF_{INV}$ values are listed in Supplement 4. Note that the earthworm data serves as proxy for the darkling beetle and other desert terrestrial invertebrates for which there are no known $BCF_{INV}$ values.

**Mass-Limited Values**. In some cases, $BCF_{INV}$ predicts the accumulation in soil invertebrates of more COPC than is deposited on the soil. Mass-limited $BCF_{INV}$ values were derived for organic COPCs by (1) assuming that all of the COPC is taken up by the soil invertebrates, (2) calculating the concentration of COPC in all of the soil invertebrates in 1 $m^2$, and (3) dividing that concentration by the concentration

of COPC in the soil if all of the COPC is mixed in the rooting zone, which is the upper 15 cm. The maximum possible uptake factor can be calculated by assuming that all of the COPC deposited in a unit area of soil in a specified time period is taken up into the mass of soil invertebrates contained in that area as shown in the following equation.

$$\text{Maximum possible bioconcentration factor} = \frac{\text{Total COPC deposited (mg/m}^2\text{) / Soil invertebrate mass density (kg/m}^2\text{)}}{\text{Total COPC deposited (mg/m}^2\text{) / Soil density (kg/m}^2\text{)}}$$

This equation can be reduced to:

$$\text{Maximum possible bioconcentration factor} = \frac{\text{Soil density (kg soil/m}^2\text{)}}{\text{Soil invertebrate mass density (kg soil invertebrate/m}^2\text{)}} \qquad \text{(Equation 8-20)}$$

The mass of soil per $m^2$ is 1300 $kg/m^3$ × 0.15 m = 195 kg/ $m^2$. The mass of soil invertebrates per $m^2$ is assumed to be 0.04 $kg/m^2$ (Gonzalez et al. 1999; average reported for *Dacryodes* community). Therefore, the maximum possible $BCF_{INV}$ is 195/0.04 = 4875. The mass-limited maximum possible value is the same for all organic COPCs because it does not depend on deposition rate or $K_{ow}$, rather soil density and mass density of the receptor. It is mass-limited or deposition-limited because all the mass deposited is accumulated by the receptor. The lesser of the measured or empirically derived $BCF_{INV}$ and the mass-limited $BCF_{INV}$ is used to predict constituent uptake.

**Bioconcentration Factors to Mammal and Bird Tissues**

The transfer factor to tissues ($BCF_A$) is the ratio of the COPC or ROPC concentrations in animal tissue to the COPC or ROPC concentration in the material it ingests [(mg/$kg_{tissue}$ wet wt)/ (mg/kg ingested)]. The $BCF_A$ is used to estimate the tissue concentration of animals exposed to COPCs and ROPCs by ingestion of soil, water, and plants from the concentration of COPC or ROPC in the ingested material. The exposure evaluation will consider two versions of $BCF_A$, measured or empirically derived values and mass-limited values.

**Measured or empirically determined values**. The measured or empirically determined $BCF_A$ is defined as:

$$BCF_A = \frac{C_A}{C_M} \qquad \text{(SLERAP Eq. 5-2)}$$

where:

$BCF_A$ = bioconcentration factor (unitless [soil, sediment], or L/kg [water])

$C_A$ = constituent concentration in the terrestrial receptor (animal of interest) (mg/kg or pCi/g)

$C_M$ = constituent concentration in media (mg/kg or pCi/g [soil, sediment], or mg/L or pCi/L [water])

Tissue concentrations of COPCs are estimated for animals that are fed upon by wildlife receptors, and tissue concentrations of ROPCs are estimated for all animals.

For medium-to-tissue accumulation factors for mammals and birds, EPA draft guidance (1999) calls for the use of Baes et al. (1984) and Travis and Arms (1988) *Ba* values (d/kg) multiplied by the receptor's absolute ingestion rate for the medium (kg-medium/d). Thus, three *BCF* values are calculated for each COPC and ROPC and each receptor. The *BCF*s are calculated using a modified version of SLERAP equations. The modification is necessary to derive *BCF*s from ingestion rates reported on a body-weight basis:

$$BCF_S = Ba \times IR_F \times SFr \times BW \qquad \text{(modified SLERAP Eq. D-1-1)}$$

$$BCF_P = Ba \times IR_F \times BW \qquad \text{(modified SLERAP Eq. D-1-1)}$$

$$BCF_W = Ba \times IR_w \times BW \qquad \text{(modified SLERAP Eq. D-1-2)}$$

where:

| | | |
|---|---|---|
| $BCF_S$ | = | soil-to-tissue uptake factor for receptor (kg soil/kg tissue) |
| $BCF_P$ | = | plant-to-tissue uptake factor for receptor (kg plant/kg tissue) |
| $BCF_W$ | = | water-to-tissue uptake factor (L water/kg tissue) |
| $Ba$ | = | ingestion-to-tissue transfer factor (d/kg) |
| $IR_F$ | = | daily food ingestion rate (kg/kg BW/d) |
| $IR_W$ | = | daily water ingestion rate (kg/kg BW/d) |
| $SFr$ | = | soil ingested per unit of food ingested (unitless) |
| $BW$ | = | body weight of receptor (kg)$IR_w$ = ingestion rate of water by receptor (L/kgBW/d) |

Note: The equations above first appear in Section 8.2.3.1.

*Ba* and *BCF* values are reported in Supplement 4. The first choice for *Ba* values for mammals was EPA draft guidance (1999). *Ba* values for inorganic COPCs and ROPCs that were not included in EPA draft guidance (1999) were taken from Baes et al. (1984), as recommended by EPA (1999). When published field or laboratory values for organic COPCs are not available for mammals, EPA (1999) guidance was followed by using the following regression on $K_{ow}$ (Travis and Arms 1988) for organic compounds (except chlorinated dibenzodioxins and chlorinated dibenzofurans, which use *BEF*s):

$$\log Ba = \log K_{ow} - 7.6 \qquad \text{(SLERAP Eq. D-1-4)}$$

*Ba* values for dioxins/furans presented by EPA (1999) are *Ba* values presented in EPA (1995a). If neither a *Ba* value nor a $K_{ow}$ is available, no tissue concentration will be calculated.

The first choice for *Ba* values for birds was EPA draft guidance (1999). *Ba* values for inorganic COPCs and ROPCs that were not included in EPA draft guidance (1999) are the same as for mammals. For organic COPCs (except dioxins/furans) the Travis and Arms (1988) *Ba* values for mammals were adjusted for the lower fat content of birds. Per EPA (1999) draft guidance, biotransfer factors for uptake of organic compounds by birds were adjusted for body fat content by multiplying the biotransfer factor by 0.8, which is the assumed ratio of body fat in birds to body fat in mammals.

**Mass-Limited Values**. It is possible for *Ba* values to predict the accumulation of more mass of a COPC or ROPC than is ingested by the receptor (see Section 7.1.7 for a detailed discussion). The maximum

possible uptake factor can be calculated by (1) assuming that all of the COPC or ROPC deposited in a unit area of soil in a specified time period is taken up into the food consumed by animals in that area, (2) assuming that the food is consumed by the receptor at a uniform rate during its lifetime, (3) calculating the ingestion rate of COPC or ROPC by the receptor, and (4) assuming that the receptor accumulates all of the COPC or ROPC during its lifetime. The maximum possible uptake factor is calculated as shown in the following equation:

$$\text{Maximum possible bioconcentration factor} = \frac{\text{COPC or ROPC concentration in receptor tissue (mg/kg)}}{\text{Consumption rate of COPC or ROPC (mg/d)}}$$

Assuming 100 % of the COPC or ROPC in the ingested food is transferred to the animal tissue, the tissue concentration can be calculated as:

$$\text{Concentration in receptor tissue} = \frac{\text{Total COPC or ROPC ingested (mg COPC or ROPC)}}{\text{Tissue weight (kg FW)}}$$

The total COPC or ROPC ingested can be expressed as:

$$\text{Total COPC or ROPC ingested} = \text{Concentration in food (mg/kg)} \times \text{Consumption rate of feed (kg/d)} \times \text{Exposure duration (d)}$$

Combining equations, the general equation for calculating the mass limited food-to-receptor tissue uptake factor can be expressed as:

$$\text{Mass limited uptake factor} = \frac{\text{Concentration in food (mg/kg)} \times \text{Consumption rate of feed (kg/d)} \times \text{Exposure duration (d)}}{\text{Tissue weight (kg FW)} \times \text{Concentration in food (mg/kg)} \times \text{Consumption rate of feed (kg/d)}}$$

This equation can be reduced to:

$$\text{Maximum possible bioconcentration factor} = \frac{\text{Exposure duration (d)}}{\text{Tissue weight (kg FW)}} \qquad \text{(Equation 8-21)}$$

The lesser of the measured or empirically derived *Ba* and the mass-limited *Ba* is used to predict constituent uptake. Only receptors with high body weights relative to the lengths of their lives could have mass-limited uptake factors less than the reported or calculated *Ba* values, and in such circumstances, the mass-limited *Ba* value will not be used in lieu of reported or calculated *Ba* values.

**FCMs**. *FCM*s are factors that are used to quantify bioaccumulation through the food chain. As chemicals from the environment pass up the food chain, they may become successively more concentrated at each trophic level. This is especially true of organic chemicals that are not metabolized rapidly. Typically, organic chemicals that dissolve in lipids bioaccumulate because they are stored in body fat, and the more soluble in lipids the chemical is, the more it bioaccumulates. To model this tendency quantitatively, EPA (1995b) measured bioaccumulation factors for organic chemicals taken up through the food chain from water by fish. An *FCM* was derived for each chemical tested by dividing the observed *BAF* by the $K_{ow}$. The EPA 1995b was able to show an orderly relationship between *FCM* and $K_{ow}$ for many organic chemicals taken up by fish at trophic levels 2, 3, and 4. By using this relationship,

the concentration of a chemical in fish tissue, normalized to lipid content, can be calculated by multiplying the concentration of the chemical dissolved in water by the *BCF* of the chemical and by the chemical's *FCM* (refer to SLERAP Section 5.3.2.3).

The EPA (1995b) has published *FCM*s to describe the ratio of bioaccumulation from animal prey to each of the trophic levels of aquatic predators. The EPA (1999) has adopted the use of *FCM*s to estimate the concentrations of organic COPCs in mammals and birds from ingested animal tissue. The *FCM*s will be used to estimate the bioaccumulation of COPCs by omnivores and of ROPCs by all omnivorous and carnivorous receptors. The *FCM*s will be used to calculate bioaccumulation from animal prey only. Bioaccumulation from ingested plants will be calculated by using a $BCF_P$.

The concentration of a contaminant in a predator will be calculated as the concentration in the prey multiplied by the predator's *FCM* and divided by the prey's *FCM*. The *FCM*s for organic COPCs are reported in Supplement 4, along with $K_{ow}$ and log $K_{ow}$. Where $K_{ow}$ values are not available, default values are not used.

All *FCM*s are assumed to equal 1 for both inorganic COPCs and for ROPCs.

#### 8.2.5.4 Bioaccumulation Factors for Calculating Aquatic Exposures

The calculation of exposure for ecological receptors may require one or more bioaccumulation or transfer factors to estimate the concentration in the tissue of an organism from the concentrations in the contaminated media to which it is exposed (Figure 8-13). Such factors are required to estimate exposure for wildlife receptors, such as mammals and birds that are exposed to COPCs and ROPCs in sediment or surface water by ingestion of plants, benthic invertebrates, or aquatic biota, when the concentration in the ingested organism is not measured directly. In each case, the numerator of the factor must have units corresponding to the units of concentration in the medium taking up the substance (tissue), and the denominator must have units corresponding to the units of concentration in the "source" medium (sediment, water, and tissue). The rules for use and derivation of these factors follow.

**Water-to-Plant Bioconcentration Factor**

The water-to-plant bioconcentration factor ($BCF_W$) is the ratio of the COPC or ROPC concentration in aquatic plant tissue to that dissolved in water [(mg/kg$_{plant}$ wet wt)/(mg/L)]. The $BCF_W$ will be used to estimate the tissue concentration of aquatic plants exposed to COPCs and ROPCs in surface water from the concentration of COPC or ROPC dissolved in surface water (Figure 8-13). That is:

$BCF_W$ = water-to-plant transfer factor for aquatic plant tissue for COPC or ROPC dissolved in water (L/kg$_{plant}$ wet wt)

Aquatic plants will be assumed to be exposed only to the dissolved phase of contaminants in surface water. Concentrations will be estimated for aquatic plant tissues that are fed upon by terrestrial receptors (e.g., Canada goose).

$BCF_W$ values presented by EPA (1999) are used if they are available. Per EPA draft guidance (1999), values of $BCF_W$ for organic COPCs for which no measured values were available were calculated with an empirically derived equation for uptake from water by aquatic invertebrates (Southworth, Beauchamp, and Schmieder 1978). The equation is:

$$\log BCF_W = 0.819 \times \log K_{ow} - 1.146 \qquad \text{(SLERAP Eq. C-1-4)}$$

Values of $BCF_W$ for inorganic COPCs and ROPCs for which no measured values were available were calculated as the arithmetic mean of all available inorganic *BCF*s. $BCF_W$ values are presented in Supplement 4.

**Sediment-to-Plant Bioconcentration Factor**

The sediment-to-plant transfer factor ($BCF_{sed}$) is the ratio of the COPC or ROPC concentration in aquatic plant tissue to that in sediment [(mg/kg$_{tissue}$ wet wt)/(mg/kg$_{sediment}$ dry wt)]. The SP will be used to estimate the tissue concentration of aquatic plants exposed to COPCs and ROPCs in sediment (Figure 8-13). That is:

$BCF_{sed}$ = sediment-to-plant transfer factor for aquatic plant tissue (kg$_{sediment}$ dry wt/kg$_{plant}$ wet wt)

Per EPA draft guidance (1999), *BCF*s for the uptake from sediment by aquatic plants are assumed to be the same as *BCF* values for uptake from soil by terrestrial plants. The $BCF_{sed}$ values for organic COPCs are taken from EPA (1999). For organic compounds with no field or laboratory data, $BCF_{sed}$ is estimated using the Travis and Arms (1988) regression on $K_{ow}$:

$$\log BCF_{sed} = 1.588 - (0.578 \times \log K_{ow}) \qquad \text{(SLERAP Eq. C-1-2)}$$

The $BCF_{sed}$ values for inorganic COPCs are taken from EPA draft guidance (1999), Baes et al. (1984), and Cappon (1981) and are provided in Supplement 4. $BCF_{sed}$ values for ROPCs are also taken from Baes et al. (1984) and provided in Supplement 4. Values of $BCF_{sed}$ for inorganic COPCs and ROPCs for which no measured values were available were calculated as the arithmetic mean of all available inorganic $BCF_{sed}$ values.

**Water-to-Fish Tissue Bioconcentration Factor**

The COPCs and ROPCs are taken up by fish both directly from water and through the food chain. Direct uptake will be calculated by using a *BCF*, and trophic transfer through the food chain will be calculated by using *FCM*s. These factors are discussed below.

**Direct Uptake**. The water-to-fish tissue transfer factor (*BCF*) is the ratio of the concentration of COPC or ROPC in the tissue of an aquatic receptor to the concentration in water [(mg/kg$_{tissue}$ wet wt)/(mg/L)]. The fish *BCF* will be used to estimate the tissue concentration of fish from the concentration in the water to which the fish is exposed (Figure 8-13). That is:

$BCF_W$ = water-to-fish tissue transfer factor (L/kg$_{tissue}$ wet wt)

The fish tissue concentrations are estimated because fish are consumed by wildlife receptors such as herons, bald eagles, and mink.

The first choice for *BCF*s for fish are values reported in EPA draft guidance (1999) or developed using EPA methods (EPA 1999). For organic compounds for which no measured data were available, *BCF*s for fish were calculated using the following regression on the $K_{ow}$ (Bintein et al. 1993):

$$\log BCF_W = 0.91 \times \log K_{ow} - 1.975 \times \log (6.8 \times 10^{-7} \times K_{ow} + 1.0) - 0.786 \qquad \text{(SLERAP Eq. C-1-8)}$$

For inorganic COPCs and ROPCs for which no measured data is available, per EPA draft guidance (1999), the $BCF_W$ was estimated as the arithmetic average of available $BCF_{fish}$ values for other inorganics. $BCF_W$ values are presented in Supplement 4.

**FCMs**. The EPA (1995b) has published *FCM*s to describe the ratio of bioaccumulation from entry into the food chain to each of four trophic levels of predators. The concentration of a contaminant in an aquatic predator is calculated as the concentration in the prey multiplied by the predator's FCM and divided by the prey's *FCM*. For example, if a heron, which is a carnivore at Trophic Level 4, has a diet of omnivorous fish at Trophic Level 3, the resulting concentration of COPC or ROPC in the heron is calculated as the concentration of COPC or ROPC in the omnivorous fish multiplied by the *FCM* for Level 4 and divided by the *FCM* for Level 3 (Figure 8-13). The *FCM*s for organic COPCs are reported in Supplement 4. All *FCM*s for inorganic COPCs and for ROPCs are assumed to equal 1.

**Sediment-to-Benthic Invertebrate Tissue Transfer Factor**

The $BCF_{INV}$ is the ratio of the COPC or ROPC concentrations in benthic invertebrate tissue to the COPC or ROPC concentration in bulk sediment [(mg/kg$_{tissue}$ wet wt)/(mg/kg$_{sediment}$ dry wt)]. The $BCF_{INV}$ is used to estimate the tissue concentration of benthic invertebrates exposed to COPCs and ROPCs in sediment by all exposure routes (ingestion, direct contact) from the concentration of COPC or ROPC in bulk sediment (Figure 8-13). The tissue concentration is estimated for animals that are fed upon by wildlife receptors. That is:

$BCF_{INV}$ = sediment-to- benthic invertebrate tissue transfer factor (kg$_{sediment}$ dry wt/kg$_{tissue}$ wet wt)

where the animal is typically a benthic invertebrate, such as a burrowing crustacean or insect, which are important diet items of predators, such as the spotted sandpiper and certain fishes.

The $BCF_{INV}$ values are available in the literature for only a few COPCs and ROPCs. The first choice for $BCF_{INV}$ values is field or laboratory values provided by the EPA (1999). Values of $BCF_{INV}$ values for inorganic COPCs and ROPCs for which no measured values are available are calculated as the arithmetic mean of all available inorganic $BCF_{INV}$ values. $BCF_{INV}$ values are reported in Supplement 4.

For organic COPCs for which no measured data is available, $BCF_{INV}$ values for benthic invertebrates were calculated per EPA (1999) from the octanol water-partitioning coefficient ($K_{ow}$) using the regression equation for daphnids (Southworth et al. 1978):

$$\log BCF_{INV} = 0.819 \times \log K_{ow} - 1.146 \qquad \text{(SLERAP Eq. C-1-9)}$$

where:

$BCF_{INV}$ = sediment-to-tissue transfer factor for benthic invertebrates (kg$_{sediment}$ dry wt/kg$_{tissue}$ wet wt)

$K_{ow}$ = octanol-water partition coefficient of COPC (L/kg)

For organic COPCs with log $K_{ow}$ values $\geq$2.6, the equilibrium partitioning approach will be used (Section 8.2.3.1). Thus, the calculated $BCF_{INV}$ will be multiplied by the calculated sediment porewater

concentration rather than the concentration in sediment to calculate the tissue concentration for the benthic invertebrate.

**Surrogate Bioaccumulation Values**

Surrogate values for $K_{ow}$ were used to calculate *BCF*s for a number of organic COPCs whose structural and chemical properties are similar to those of the COPCs used as surrogates. The use of surrogates in *BCF* calculations is indicated in Supplement 4.

## 8.3 Effects Assessment Calculations

The *TRV*s are concentrations or doses of constituents that are associated with a specified level of adverse effect. The *TRV*s (e.g., ecological soil screening level [Eco-SSL] [EPA 2003a], equilibrium partitioning sediment benchmark [ESB], final chronic value [FCV], or secondary chronic value [SCV] [EPA 2003b, 2008]) may be based on a range of concentration or dose benchmarks, including median lethal concentration (LC50), 20 % effect concentration (EC20), LOAEL, or NOAEL. The *TRV*s are used as the denominator in ecological screening quotients (*ESQ*), as shown in the *ESQ* equations (Section 8.4).

### 8.3.1 Toxicity Reference Values for Terrestrial Receptors

The *TRV*s for receptors dwelling in and, thus, exposed by direct contact with soil (plants, terrestrial invertebrates) are typically values from published sources, if field observations or site-specific toxicity tests of these media are not available. *TRV*s are tabulated in Supplement 4.

#### 8.3.1.1 Single Chemical Toxicity Reference Values for Direct Contact with Soil

The *TRV*s for plants and soil-dwelling invertebrates are derived values associated with some level of inhibition of growth or reproduction based on a review of published single-chemical laboratory studies (e.g., Efroymson et al. 1997a and 1997b).

**Terrestrial Plants**

Toxicity of COPCs to plants is assumed to be a result of uptake from soil into the plant tissues. Therefore, soil concentrations that are associated with toxicity, usually under experimental conditions, are used as *TRV*s. For terrestrial plant *TRV*s, the hierarchy of choices is as follows:

1. Values from Eco-SSL guidance (EPA 2003a)
2. Values from EPA draft guidance (EPA 1999)
3. Values from MTCA (Ecology 2001)
4. Values from Efroymson et al. (1997a)
5. Values in the ECOTOXicology database (ECOTOX) (EPA 2010)

The COPCs with no *TRV*s will not be evaluated for toxicity, and this lack of data will be handled as an uncertainty. The toxicity equivalence factor approach will be used to assess the impact of exposure to constituents for which toxicity equivalence factors have been established (see Sections 8.3.3 and 8.3.4).

EPA draft guidance (1999) presents 21 published plant *TRV*s and 7 surrogate values. For COPCs that are not included in the EPA draft guidance (1999), *TRV*s were based on a review of published single-chemistry laboratory studies (Efroymson et al. 1997a). Surrogate *TRV*s were used for some

COPCs that lack *TRV*s, as shown in Table 8-3.

These substitutions were made because of similarities in chemical structures and properties between each COPC that was assigned a surrogate and the COPC used as its surrogate. The assignment of surrogate values is subject to change as appropriate data becomes available (i.e., empirical studies are published in the future). The derivation of *TRV*s for terrestrial plants is presented in Supplement 4.

**Terrestrial Invertebrates**

Toxicity of COPCs to terrestrial invertebrates is assumed to be a result of uptake from soil into the invertebrate's tissues. Therefore, soil concentrations that are associated with toxicity, usually under experimental conditions, are used as *TRV*s. For terrestrial invertebrate *TRV*s, the hierarchy of choices is as follows:

1 Values from Eco-SSL guidance (EPA 2003a)
2 Values from EPA draft guidance (EPA 1999)
3 Values from the MTCA (Ecology 2001)
4 Values from Efroymson et al. (1997b)
5 Values in the ECOTOXicology database (ECOTOX) (EPA 2010)
6 Values in published literature

The COPCs without *TRV*s will not be evaluated for toxicity, and this lack of data will be handled as an uncertainty. The toxicity equivalence factor approach will be used to assess the impact of exposure to constituents for which toxicity equivalence factors have been established (see Sections 8.3.3 and 8.3.4).

The EPA draft guidance (1999) presents 16 published terrestrial invertebrate *TRV*s and 8 surrogate values. For the many COPCs that are not included in the EPA draft guidance (1999) or Washington State Department of Ecology guidance (Ecology 2001), *TRV*s were based on a review of published single-chemistry laboratory studies (Efroymson et al. 1997b). Surrogate *TRV*s were used for some COPCs that lack *TRV*s (Table 8-3).

These substitutions were made because of similarities in chemical structures and properties between each COPC that was assigned a surrogate and the COPC used as its surrogate. The development of *TRV*s for terrestrial invertebrates is presented in Supplement 4.

#### 8.3.1.2 Toxicity Reference Values for Ingestion Exposure of Terrestrial Receptors

For wildlife receptors, ingestion *TRV*s will be used to calculate *ESQ*s for the ingestion exposure pathway. For terrestrial mammal and bird *TRV*s, the hierarchy of choices is as follows:

1 Values from Eco-SSL guidance (EPA 2003a)
2 Values from EPA (1999) draft guidance
3 Values from Sample et al. (1996)
4 Values from the ECOTOXicology database (ECOTOX)(EPA 2010)

The COPCs with no *TRV*s will not be evaluated for toxicity, and this lack of data will be handled as an uncertainty. The toxicity equivalence factor approach will be used to assess the impact of exposure to constituents for which toxicity equivalence factors have been established (see Sections 8.3.3 and 8.3.4).

The EPA draft guidance (1999) presents 42 published *TRV*s and 3 surrogate values for mammals and 32 published *TRV*s and 4 surrogate values for birds. For the many COPCs that are not included in the EPA draft guidance (1999), *TRV*s were based on a review of published single-chemistry laboratory studies (Sample et al. 1996).

The outputs from the toxicity studies are subchronic or chronic NOAEL or LOAEL doses (mg/kg BW/d) for the test species. Per EPA draft guidance (1999), if the NOAEL is from a subchronic study, the benchmark is adjusted downward by a factor of 10 to estimate the chronic benchmark. If the benchmark is a LOAEL for a mortality or reproduction endpoint, it is adjusted downward by a factor of 10 to estimate the NOAEL. A subchronic LOAEL is adjusted downward by a factor of 100 to estimate the chronic NOAEL. An uncertainty factor of 100 is applied to acute single-point estimates (e.g., LD50 values) to determine a *TRV*. Surrogate *TRV*s were used for some COPCs that lack *TRV*s (Table 8-3).

These substitutions were made because of similarities in chemical structures and properties between each COPC that was assigned a surrogate and the COPC used as its surrogate. The development of *TRV*s for terrestrial receptors is presented in Supplement 4.

If the desired *TRV* corresponds to the NOAEL, then the endpoint observed in the study should be mortality or reproduction. Nonlethal or nonreproductive NOAELs are conservative (i.e., lower than necessary to protect the receptor), but are used if a NOAEL for mortality or reproduction is not available. If the *TRV* is a LOAEL, then the endpoint observed in the study should be nonlethal or a nonreproductive effect. If the observed LOAEL endpoint is mortality or reproduction, then the nonconservative nature of the *TRV* should be considered in the risk characterization.

#### 8.3.1.3 Toxicity Reference Values for Radiation Exposure of Terrestrial Receptors

Exposure to ionizing radiation (alpha particles, beta particles, and gamma rays) will be evaluated for toxicity to ecological receptors. The risk of cancer as a result of exposure to ionizing radiation is not calculated for ecological receptors because low doses of radiation typically do not induce cancer that would be lethal before the receptors are able to reproduce. Doses that would be associated with cancer risks that would cause marked reductions in populations would be extremely high. Instead, naturally occurring exposures that have been associated with little or no damage to populations are used to derive radiological *TRV*s.

The benchmark values for radiation given by the International Atomic Energy Agency (IAEA) (1992) are 1 mGy/d (0.1 rad/day) for terrestrial mammals and birds, and 10 mGy/d (1 rad/day) for plants, invertebrates, and aquatic biota. These benchmarks are confirmed in *Effects of Ionizing Radiation on Terrestrial Plants and Animals: A Workshop Report* (Barnthouse 1995). Alpha radiation has a much higher effect on biological tissue than beta and gamma radiation because of the large mass of the alpha particle. When internal exposure is being evaluated, it is particularly important to consider the relative effectiveness of the radiation (CCN 063808). To adjust for the greater damage done by alpha particles than by beta and gamma radiation, a *QF* of 10 (Kocher and Trabalka 2000) for alpha radiation was included in the dose calculations to evaluate exposure to ROPCs.

### 8.3.2 Toxicity Reference Values for Aquatic Receptors

The *TRV*s for receptors dwelling in and, thus, exposed by direct contact to sediment (benthic invertebrates) or surface water (fish, aquatic biota) are typically values from published sources if field observations or site-specific toxicity tests of these media are not available. The units of these values vary by source and medium (e.g., μg/L for surface water and mg/kg dry wt for sediment).

The toxicity equivalence factor approach will be used to assess the impact of exposure to constituents for which toxicity equivalence factors have been established (see Sections 8.3.3 and 8.3.4). In other cases where *TRV*s are not available, surrogate *TRV*s are assigned as shown in Supplement 4 and described below.

#### 8.3.2.1 Single Chemical Toxicity Reference Values for Direct Contact with Water and Sediment

The *TRV*s for aquatic biota and benthic invertebrates are concentrations of COPCs in the medium to which the receptors are exposed.

**Aquatic Biota**

The *TRV*s for aquatic biota are, in order of preference, FCVs (or SCVs) related to an ESB values (EPA 2003b, 2008), values published in EPA draft guidance (1999) and then other published *TRV*s. The EPA draft guidance (1999) presents 44 published *TRV*s for aquatic biota and 6 surrogate values. The hierarchy of *TRV*s not found in the EPA draft guidance (2003b, 2008, and 1999) is Washington State MTCA values (Ecology 2001), National Ambient Water Quality Criteria (NAWQC) (Suter and Tsao 1996), FCVs (Suter and Tsao 1996), Great Lakes Tier II SCVs (Suter and Tsao 1996), then other toxicity values from recently published aquatic toxicity literature. Surrogate *TRV*s were used for some COPCs that lack *TRV*s (Table 8-3).

These substitutions were made because of similarities in chemical structures and properties between each COPC that was assigned a surrogate and the COPC used as its surrogate. If there is no toxicity value for a COPC, no *TRV* will be listed, and this lack of data will be handled as an uncertainty. The development of *TRV*s for aquatic biota is presented in Supplement 4.

**Chinook Salmon and Other Salmonids**

Salmonids comprise salmon and trout species. These species have special regulatory, economic, and recreational interest in the Columbia River Basin. Chinook salmon and steelhead trout populations in the Hanford Reach of the Columbia River have been designated ESUs. Salmonids are also of particular cultural importance to the American Indian tribes, whose way of life has inextricably included salmon and trout as food throughout their history. Because of their sensitive status, salmonids will be evaluated separately from other aquatic biota, and more stringent *TRV*s were sought for exposure of salmonids in the Columbia River. Available FCVs (or SCVs) from EPA guidance for derivation of ESBs (EPA 2003b and 2008) were selected as first choice TRV values for PAHs and nonionic organics as the data used for the derivation of these values were subject to a quality review not necessarily performed in the derivation of TRVs in older EPA publications. These values account for the varying biological availability of chemicals in different sediments and allow for the incorporation of the appropriate biological effects concentration (EPA 2003b). The EPA draft guidance (EPA 1999) provides data for aquatic receptors for other constituents, but offers no specific *TRV* data for salmonids. Therefore, when aquatic toxicity values were found that were lower than the *TRV*s listed in EPA draft guidance, they were used as alternative

*TRV*s for salmonids. Surrogates were made where similarities in chemical structures and properties between each COPC that was assigned a surrogate and the COPC used as its surrogate justified the use (Table 8-3).

Sensitive species chronic values (Suter and Tsao 1996) were used as *TRV*s for salmonids in preference to *TRV*s for aquatic biota, whether or not they came from tests on salmonids. Other published toxicity values for salmonids were also used as *TRV*s for salmonids if they were lower than the *TRV*s for aquatic biota. These data did not necessarily meet criteria for use to calculate NAWQC, but were used as highly conservative screening *TRV*s. Population $EC_{20}$ values (Suter and Tsao 1996) were also used. The population $EC_{20}$ is a value calculated by a computer model using a variety of toxicity data and is intended to be the lowest chronic exposure that would reduce population recruitment by 20 % (Suter and Tsao 1996). Because the calculation produces a range of concentrations for each COPC, the reported 5th percentile lower bound was used as a conservative *TRV*. The derivation of *TRV*s for salmonids is presented in Supplement 4.

**Benthic Invertebrates**

The *TRV*s for benthic invertebrates are, in order of preference, values from EPA guidance for derivation of ESBs (EPA 2003b and 2008), those published in EPA draft guidance (1999), and then other published *TRV*s. The EPA draft guidance (1999) presents 27 published benthic invertebrate *TRV*s and 19 calculated or surrogate values. The hierarchy of *TRV*s is as follows:

1 Values from EPA guidance for derivation of ESBs (EPA 2003b and 2008)[4]
2 Values from EPA (1999) draft guidance
3 No-effect levels and lowest-effect levels from Persaud et al. (1993)
4 Apparent effects thresholds from Ecology (1994)
5 Values published by Ingersoll et al. (1996)

For COPCs whose values are not available from those sources, values and methods found in Jones, Suter, and Hull (1997) were used. Surrogate *TRV*s were used for some COPCs that lack *TRV*s. In addition to surrogates given in the SLERAP (EPA 1999), surrogates listed in Table 8-3 were used.

These substitutions were made because of similarities in chemical structures and properties between each COPC that was assigned a surrogate and the COPC used as its surrogate. If there is no TRV in these sources, no TRV is listed, and this lack of data will be handled as an uncertainty. The development of *TRV*s for benthic invertebrates is presented in Supplement 4.

#### 8.3.2.2 Toxicity Reference Values for Ingestion Exposure of Predators of Aquatic Biota

The *TRV*s for ingestion exposure of predators of aquatic biota are the same as those for terrestrial mammals and birds (Section 8.3.1.2), with some exceptions[5]. The source of *TRV*s for mammal and bird receptors is presented in Supplement 4.

[4] ESB values (organic carbon based values) are converted to TRVs for benthic invertebrates by multiplying them by the fraction of organic carbon in the bed sediment ($f_{oc,bs}$).

[5] The exception for use of Eco-SSL values is for the burrowing owl, as the guidance (EPA 2003a, Sect. 1.1) cautions the user that SSL exposure pathways may not be complete for burrowing mammals (i.e., inhalation and dermal exposure pathways may not be negligible for burrowing animals for some chemicals)

#### 8.3.2.3 Toxicity Reference Values for Radiation Exposure of Aquatic Biota

Exposure to ionizing radiation (alpha particles, beta particles, and gamma rays) will be evaluated for toxicity to ecological receptors. The risk of cancer as a result of exposure to ionizing radiation is not calculated for ecological receptors because low doses of radiation typically do not induce cancer that would be lethal before the receptors are able to reproduce. Doses that would be associated with cancer risks that would cause marked reductions in populations would be extremely high. Instead, naturally occurring exposures that have been associated with little or no damage to populations are used to derive radiological *TRVs*.

For all sediment and aquatic biota, the TRV for total (external + internal) whole-body radiological dose from combined external and internal exposure for all ROPCs combined is 1.0 rad/day (IAEA 1992). However, the *TRV* for aquatic wildlife receptors (i.e., birds and mammals) is 0.1 rad/day.

### 8.3.3 Toxicity Equivalence Factors for Dioxins, Dibenzofurans, and PCBs

Chlorinated dioxins, chlorinated dibenzofurans, and chlorinated biphenyls are evaluated as a group because they are thought to act through a common mechanism of toxicity. These chemicals are thought to act by binding to a protein known as the arylhydrocarbon receptor (AR) (see ATSDR 1997 or WHO 1998). The AR-ligand complex is responsible for the activation of genes that have a deleterious effect when they are not under proper regulation by the receptor's hormones. Interaction of dioxins and similar compounds with AR, therefore, can cause immunological, neurological, endocrine, embryotoxic, and other effects.

The similarity in action of these compounds is thought to result from their structural similarity. Dioxin is composed of two benzene rings joined by two carbon-oxygen-carbon bonds on two adjacent carbons of each benzene ring. Dibenzofurans have two benzene rings joined by a carbon-oxygen-carbon bond and a carbon-carbon bond on two adjacent carbons of each benzene ring. Biphenyls consist of two benzene rings joined by a single carbon-carbon bond. To form the polychlorinated derivatives, chloro groups are attached at various locations, as designated in the names of the compounds. Benzene rings are planar (flat) in conformation. Because two adjacent carbons on each benzene ring are joined in dioxins and dibenzofurans, both benzene rings are held in the same plane, and the chloro groups are also in that plane. Therefore, these molecules are said to be coplanar. The coplanar structure appears to be essential for interaction with AR. The benzene rings in biphenyl can rotate relative to each other, unless there are added groups that interfere with rotation (such as 2,2',6,6'-chloro groups, which occupy the carbons immediately on both sides of the carbon-carbon bond joining the rings). The PCB congeners that are able to form a coplanar molecule (and are called *coplanar* PCBs) can interact with AR when they are in that configuration. Therefore, coplanar PCBs are included among the COPCs with similar action to dioxins and dibenzofurans.

The EPA has recommended that TEFs be used to evaluate the cumulative toxicity of chlorinated dioxins, chlorinated dibenzofurans, and chlorinated biphenyls. Because these contaminants have a common mechanism of action, it is assumed that their toxicity to biota is additive (WHO 1998, EPA 1999) (i.e., the toxicity of all dioxins, dibenzofurans, and PCBs should be added). Furthermore, their relative potency as chronic toxins is assumed to be related to the degree of affinity for AR, which can be measured much more conveniently than chronic toxic effects. The TEFs have been proposed for several chlorinated dioxins, chlorinated dibenzofurans, and chlorinated biphenyls (WHO 1998, EPA 1999), always assigning the toxicity of TCDD, the most potent chlorinated dioxin, a TEF of 1.0. Separate lists were developed for mammals, birds, and fish, and these lists are presented in Supplement 4.

The TEFs are reported in Supplement 4 for individual PCB congeners (such as 2,3,3',4,4',5-hexachloro-biphenyl), but analytical values for individual congeners in the exposure media are sometimes not available. It is also possible to calculate TEFs for Aroclors, which are mixtures of PCB congeners, using the typical composition of Aroclor mixtures.

Using TEFs, *ESQ*s can be calculated for chlorinated dioxins, chlorinated dibenzofurans, and PCBs for which *TRV*s are not available. The *TRV* for 2,3,7,8-TCDD is divided by a COPC's TEF to calculate an equivalent *TRV* of that corresponds to a dioxin or furan without published *TRV* data. The TCDD-equivalent *TRV* of the COPC is then used to calculate the *ESQ* for the COPC. Because the mechanism of action of these compounds is thought to be the same, the TCDD-equivalent *ESQ*s are added to determine the hazard index (*HI*) for the set of dioxins and dibenzofurans.

### 8.3.4 Toxicity Equivalence Factors for PAHs

As pure chemicals, PAHs generally exist as colorless, white, or pale yellow-green solids. They can have a faint, pleasant odor. They are found throughout the environment in the air, water, and soil. They can occur in the air, either attached to dust particles or as solids in soil or sediment. Studies in animals have also shown that PAHs can cause harmful effects on skin, body fluids, and the body's system for fighting disease after both short- and long-term exposure (ATSDR 1995).

EPA 2003b establishes FCVs for PAHs using the NAWQC Guidelines (Stephan et al., 1985). These values serve as TRVs for aquatic biota and fish. The guidance also provides the corresponding ESBs (equivalent concentration in sediments on an organic carbon basis) as predicted from FCVs using the carbon partition coefficient ($K_{oc}$). These values can be converted to TRVs for benthic invertebrates by multiplying them by the fraction of organic carbon in the bed sediment ($f_{oc,bs}$). The guidance also notes that because PAHs occur in sediments as mixtures and their toxicities in water, tissues, or sediments are additive or nearly additive, their combined toxicities must be considered to assess the impact of PAH mixtures. If the SLERA indicates a potential issue from PAH exposure, the additive effect of the PAH mixture will be assessed as well as their individual impact (see Section 8.4.3).

Additionally, the State of Washington has published TEFs for many of these compounds in MTCA (WAC 173-340-900). These TEFs will be used where appropriate (i.e., for mammals) to calculate equivalent *TRV*s.

## 8.4 Risk Characterization

Risk estimates for a receptor at an exposure location are calculated as the *ESQ*, which is the ratio of the estimated exposure to the *TRV*. That is:

$$ESQ = \frac{EEL}{TRV} \qquad \text{(SLERAP Eq. 6-1)}$$

where:

$ESQ$ = ecological screening quotient (unitless)

*EEL* = constituent estimated exposure level (mass of constituent per mass of media [communities] or mass daily dose constituent ingested per mass body weight-day [class-specific guilds])

*TRV* = toxicity reference value (mass of constituent per mass of media [communities] or mass daily dose ingested per mass body weight-day [class-specific guilds])

The *ESQ* is an index of the total risk to the receptor from exposure to the COPC if the COPC does not occur in the environment from any other source and if the home range of the receptor is smaller than the area of the exposure location, that is, if the *AUF* = 1.

The *ESQ* equation takes different forms depending on how the receptor is exposed, which also determines how the *TRV* is expressed. In the SLERA for the WTP, the exposure to ecological receptors will be a media concentration (EPC), an average daily dose of a COPC (*DD*), or a daily total (external + internal) whole-body radiological dose ($DD_{Rad}$).

There is limited data for developing inhalation *TRV*s and very limited data for developing dermal *TRV*s. Little is known about the actual absorption across the dermal layer of wildlife receptors. There is also uncertainty about the extrapolation of *TRV*s for ingestion to inhalation. Therefore, inhalation and dermal absorption exposures will not be evaluated quantitatively.

### 8.4.1 Terrestrial Receptors

For receptors living in soil (such as plants and terrestrial invertebrates), the *ESQ* will be calculated as the ratio of the concentration of COPC in soil and the *TRV* for the receptor and the COPC. That is:

$$ESQ = \frac{Cs_{15}}{TRV} \qquad \text{(modified SLERAP Eq. 6-1)}$$

where:

*ESQ* = hazard quotient for the receptor at its exposure location for the COPC (unitless)

$Cs_{15}$ = concentration of the COPC in soil at the exposure location based upon a 15 cm root zone soil depth ($mg/kg_{soil}$)

*TRV* = toxicity reference value of the receptor for the COPC ($mg/kg_{soil}$)

The *ESQ* for a wildlife receptor that does not live in the medium containing COPCs, but is exposed by ingestion and other routes, will be calculated as the ratio of the *DD* and the *TRV*. That is:

$$ESQ = \frac{DD}{TRV} \qquad \text{(modified SLERAP Eq. 6-1)}$$

where:

*DD* = daily dose of the COPC or ROPC to the receptor at the exposure location (mg/kg BW/d or rad/day) calculated using the concentration of the COPC or ROPC at the exposure location

*TRV* = toxicity reference value of the COPC or ROPC for the receptor (mg/kg BW/d or rad/day). Note that the ROPC benchmark TRV for terrestrial mammals and birds is 0.1 rad/day and the ROPC benchmark *TRV* for plants and invertebrates is 1.0 rad/day

The second equation will be used to estimate risk for the wildlife receptors in the terrestrial food web: mule deer, mourning dove, Great Basin pocket mouse, western meadowlark, coyote, burrowing owl, and red-tailed hawk.

### 8.4.2 Aquatic Receptors

For receptors living in surface water or sediment (e.g., aquatic life and salmon and other fish living in surface water, and benthic organisms living in sediment), the *ESQ* will be calculated as the ratio of the measured concentration of COPC in the medium and the *TRV*. That is:

$$ESQ = \frac{C_M}{TRV} \qquad \text{(modified SLERAP Eq. 6-1)}$$

where:

*ESQ* = hazard quotient for the receptor at its exposure location for the COPC (unitless)

$C_M$ = concentration of the COPC in the exposure media; dissolved surface water, $C_{dw}$ (for fish), or sediment, $C_{sed}$ (for sediment dwellers) at the exposure location (µg/L, mg/L, µg/kg, or mg/kg)

*TRV* = toxicity reference value of the COPC for the receptor (µg/L, mg/L, µg/kg, or mg/kg)

The *ESQ* for a wildlife receptor that does not live in the surface water or sediment containing the COPCs but is exposed from aquatic food webs by ingestion, inhalation, and other routes is calculated as the ratio of the estimated *DD* (mg/kg BW/d) to the *TRV* (mg/kg BW/d). That is:

$$ESQ = \frac{DD}{TRV} \qquad \text{(modified SLERAP Eq. 6-1)}$$

where:

*DD* = daily dose of the COPC or ROPC to the receptor at the exposure location (mg/kg BW/d or rad/day) calculated using the concentration of the COPC or ROPC at the exposure location

*TRV* = toxicity reference value of the COPC or ROPC for the receptor (mg/kg BW/d or rad/day). Note that the ROPC benchmark *TRV* for aquatic mammals and birds is 0.1 rad/day and the ROPC benchmark *TRV* for aquatic biota, salmonids, and benthic invertebrates is 1.0 rad/day

The above equation will be used to estimate risk for the wildlife receptors in the aquatic food web: Canada goose, spotted sandpiper, great blue heron, bald eagle, and mink.

### 8.4.3 Total Ecological Screening Quotient

The total *ESQ* for a receptor at a given exposure location is the sum of the *ESQ*s for all COPCs with similar modes of toxicity and is an index of the combined risk from exposure to multiple COPCs. A preliminary classification of inorganic COPCs grouped arsenic, antimony, selenium, and vanadium as respiratory inhibitors; lead, manganese, and mercury as central nervous system inhibitors; and aluminum, chromium, and nickel as deoxyribonucleic acid (DNA) and protein reactors. Organic COPCs are typically grouped by chemical structure: volatile organic compounds (VOCs), PAHs, organochloride pesticides, and PCBs. These chemical groupings are based on experience. However, for the SLERA, *ESQ*s for all organic COPCs, all inorganic COPCs, and all ROPCs, regardless of mode of actions, will be grouped and summed because such summing represents the most conservative case. When the total *ESQ* exceeds 0.25, additional *ESQ*s by mode of action will be developed with approval of Ecology if a scientific management decision so indicates. The total *ESQ* for a receptor at an exposure location is calculated from the *ESQ*s for the individual COPCs as follows:

$$ESQ_{Receptor\ COPC\ Total} = \sum ESQ_{COPC\ Specific} \qquad \text{(SLERAP Eq. 6-2)}$$

where:

$ESQ_{Receptor\ COPC\ Total}$ = total ecological screening quotient for the receptor at the exposure location (unitless)

$ESQ_{COPC\ Specific}$ = COPC specific ecological screening quotient for the receptor at the exposure location (unitless)

Similarly, the total *ESQ* for a receptor at an exposure location is calculated from the *ESQ*s for the individual ROPCs as follows:

$$ESQ_{Receptor\ ROPC\ Total} = \sum ESQ_{ROPC\ Specific} \qquad \text{(SLERAP Eq. 6-2)}$$

where:

$ESQ_{Receptor\ ROPC\ Total}$ = total ecological screening quotient for the receptor at the exposure location (unitless)

$ESQ_{ROPC\ Specific}$ = ROPC specific ecological screening quotient for the receptor at the exposure location (unitless)

The *ESQ* equation for receptors exposed to ROPCs is equivalent to the *ESQ* equation for COPCs because the dose from all radionuclides is summed to estimate the total-body dose from internal and external exposures. Calculating the total *ESQ* assumes an additive effect on receptors from the summed COPCs and ROPCs; however, COPCs and ROPCs effects are not additive with one another, thus, COPCs and ROPCs are evaluated separately.

The threshold value for *ESQ*s for COPCs will be 0.25, unless a similar mode of action is demonstrated and approved by Ecology. The threshold value for *ESQ*s for ROPCs will be 1.0 rad/day for lower trophic level species (plants, aquatic biota, salmonids, and terrestrial and benthic invertebrates) and 0.1 rad/day for higher trophic level species (terrestrial and aquatic mammals and birds).

## 8.5 Reporting of Major Ecological Risk Findings

Risk characterization will be reported in such a way as to meet three goals identified in EPA guidance (EPA 1999):

1 Provides the maximum, most conservative exposure estimate
2 "Identifies which pathways are driving risk specific to a COPC and receptor"
3 "Allows risk management efforts to be prioritized"

The characterization will interpret risk findings in terms of the receptor groups represented rather than individual receptor species. For example, if there is excess risk to the Great Basin pocket mouse, the result will be interpreted as indicating potential harm to small omnivorous mammals in general.

The following outline of headings is proposed for the PRA:

I. Risk for Terrestrial Conditions: Central Plateau
   A. Organic COPCs
   B. Inorganic COPCs
   C. ROPCs
II. Risk for Aquatic Conditions: Columbia River
   A. Organic COPCs
   B. Inorganic COPCs
   C. ROPCs
III. Future Risk
   A. Terrestrial Conditions
   B. Aquatic Conditions

At each location, every COPC that equals or exceeds an *ESQ* of 0.25 will be identified along with the receptor for which the exceedance occurs. In addition, locations and receptors for which total *ESQ*s equal or exceed 0.25 will be identified, and for each such combination, COPCs and ROPCs whose *ESQ*s exceed 0.025 will be identified as significant contributors to the total *ESQ*. If the results of the SLERA indicate that one or more COPCs or ROPCs or the sum for a receptor at a given exposure location is a potential hazard (i.e., $ESQ \geq 0.25$), then exposure and toxicity information will be re-evaluated to determine whether the evaluation was overly conservative. Evaluation of sources and pathways will help identify which pathways drive the risk. This information will allow risk managers to prioritize further investigation.

Evaluation of *ESQ*s, sources, and pathways will be done for the PRA as well as the FRA within the SLERA.

## 8.6 Uncertainty in Ecological Risk Assessment

Evaluation of uncertainties is part of the SLERA process (EPA 1998). Uncertainties in each of the four interrelated steps of the EPA approach to the SLERA will be discussed as follows:

- Problem formulation
- Exposure assessment
- Effects assessment
- Risk characterization

Uncertainties about the data will be evaluated in the exposure assessment and the effects assessment steps.

### 8.6.1 Problem Formulation

Environmental concentrations of contaminants deposited on the soil and water at exposure locations will be based on many predictions. A degree of uncertainty exists about the predicted spatial distribution of contaminants. Exposure concentrations could be overestimated or underestimated, depending on how good the model is at predicting contaminant distribution. The assumption that all soil or surface water in a given exposure area contains the COPC concentrations and ROPC activities modeled for the maximum location results in an overestimate of risk to populations.

Because conservative exposure parameters (Section 8.6.2) will be used to calculate *ESQ*s, the estimates of risk from ecological COPCs and ROPCs are conservative (i.e., protective). Using conservative exposure concentrations decreases the likelihood of underestimating the risk posed by each ecological COPC/ROPC and increases the likelihood of overestimating the risk. Note that for wildlife receptors not living in soil, sediment, or surface water, *ESQ* is a function of COPC dose or radiological daily dose (*DD*), which, in turn, depends on a number of exposure factors (in addition to contaminant exposure concentration). Thus, several factors determine how conservative an *ESQ* might be (in addition to contaminant exposure concentration).

The distribution and abundance of organisms comprising the ecological receptors at exposure locations have not been quantified by field studies. The lack of quantitative data introduces uncertainties concerning whether, and to what extent, the risk characterization based on the selected receptor species underestimates, or overestimates, the risk to organisms that are not used in the risk computations but are found at exposure locations.

One (or more) unobserved species at exposure locations is possibly more sensitive than those ecological species for which toxicity data were available. It does not necessarily follow that these unevaluated species are at significantly greater risk of harmful ecological effects than that estimated in the SLERA, because their exposure may be less than the conservatively estimated exposure for WTP receptors.

### 8.6.2 Exposure Assessment

Movement of contaminants from the exposure locations through direct and indirect pathways to ecological receptors will be modeled rather than measured for the SLERA. The lack of site-specific measurements introduces uncertainties about the actual modes and pathways of exposure and the actual exposure concentrations of these contaminants to the ecological receptors. Exposure concentrations can differ from the predicted environmental concentrations as a result of physical and chemical processes during transport from source to receptor. These processes will not be predicted quantitatively in the SLERA.

The modes and pathways used to characterize the exposure of ecological receptors are the most important ones for the relatively large and active species in terrestrial habitats. Soil-dwelling terrestrial animals may

be exposed to contaminants in soil by way of inhalation. However, it is expected that concentrations of VOCs will be very small and that gaseous concentrations in soil interstices, cavities, and burrows do not exist. Inhalation exposures will not be evaluated in the SLERA. Therefore, the exposure to burrowing organisms at the site from contaminated soil and porewater in the soil may be underestimated if gas concentrations are larger than soil concentrations. Overestimating exposure by using conservative exposure concentrations is thought to offset the underestimation of exposure that results from neglecting certain exposure modes and pathways of lesser importance. Additional uncertainties are inherent in ingestion rates and dietary fractions of plants and animals. Likewise, the effects of dermal exposure may be underestimated; uncertainty about those effects will be discussed qualitatively. Exposure concentrations are likely overestimated because of conservative exposure factors. Sources of conservatism in the exposure factors include using published *BAF*s, irrespective of species and environmental conditions.

### 8.6.3 Effects Assessment

Toxicity thresholds are based on concentrations reported to have no, or little, effect on the test organism or are estimated conservatively from published toxicity data. The *TRV*s for wildlife receptors exposed to soils are derived from NOAELs or LOAELs reduced by safety factors of 10 for chronic LOAELs and subchronic NOAELs or 100 for subchronic LOAELs (Sample et al. 1996). These thresholds would underestimate the risks only to organisms at the exposure locations that are considerably more sensitive than the receptor organisms for the specific toxicological endpoint. The thresholds are more likely to overestimate the risk to organisms that are equally or less sensitive than the receptor organisms. The possibility remains that some thresholds are set at levels at or above which some harm would occur to organisms at the exposure locations because receptors may be more sensitive to other toxicological endpoints.

There is limited data for developing inhalation *TRV*s and very limited data for developing dermal *TRV*s. Little is known about the actual absorption across the dermal layer of wildlife receptors. There is also uncertainty about the extrapolation of *TRV*s for ingestion to inhalation. Therefore, inhalation exposures will not be evaluated quantitatively. The uncertainties associated with neglecting dermal contact and inhalation toxicity will be discussed in the PRA.

The risks from exposure to multiple contaminants depend on contaminant interactions; effects could be greater or less than those from a single chemical. This RAWP provides methods for estimating ecological COPC-specific risk estimates and assumes additivity for calculating *ESQ*s. Overall, the effects assessment probably overestimates toxicity because the *TRV*s are based on concentrations that cause no observed effect in test animals rather than an effect that may be observable but is not great enough to threaten populations.

*TRV*s are not available for some COPCs. This lack of *TRV*s is especially true for organic COPCs. This situation likely will result in underestimated risks.

The *TRV*s for radiation exposure were proposed as doses that are unlikely to harm populations (IAEA 1992, Barnthouse 1995). Individual plants or animals, or tissues of plants and animals, may be more sensitive to radiation damage than the populations evaluated by IAEA (1992). For example, rapidly growing tissues such as root hairs may be particularly sensitive to external radiation if they are in close contact with contaminated media. Therefore, the SLERA may underestimate risks from radiation by an unknown amount.

Additional uncertainty exists as to the pertinence of individual organism toxicity for characterizing the risk to individuals, populations, and ecosystems. Populations possibly may compensate for the loss of large

numbers of juveniles or adults with increased survival or birth rates, and habitats or ecosystems may possess functionally redundant species that are less sensitive to contaminants. Although the desert habitat at the exposure locations likely possesses some buffering mechanisms, a conservative risk assessment approach is still justified based on organismal toxicity thresholds (i.e., NOAELs), which probably result in an overestimate of risk.

### 8.6.4 Risk Characterization

The uncertainties described above ultimately produce uncertainty in the quantification of current and future risks to plants and animals at the exposure locations. An additional area of uncertainty in the risk characterization is risk to receptors outside of the exposure areas to be modeled.

It is unlikely that receptors outside the areas of maximum concentration and within the 50 km study area would have lower toxicity thresholds for contaminants than the thresholds used for receptors within those exposure areas. All representative organisms are assumed to be present at the locations of maximum concentration regardless of their actual distribution. In addition, there is little reason to expect that contaminants migrating outside the study area would be concentrated above the concentrations predicted at the exposure locations. In general, the risk to receptors outside the exposure areas is likely to be overestimated rather than underestimated (e.g., bounded) by the risk estimate for receptors at the modeled exposure areas within the 50 km radius of the site.

### 8.6.5 Summary of Uncertainties

The most important uncertainties in the ecological portion of the SLERA for exposure locations are those surrounding the estimates of the contaminant concentrations to which ecological receptors are actually exposed (EPCs) and the concentrations that present an acceptable level of risk or harmful effects (toxicity thresholds or reference values). These uncertainties arise from multiple sources (e.g., the lack of site-specific data on contaminant transport and transformation processes, organismal toxicity, animal behavior and diet, population dynamics, and the response of arid land plant and animal populations to stressors in their environments). Despite these uncertainties, the modeled exposure concentrations and published exposure and effects information will allow risks to be characterized for various exposure locations according to exposure/effects scenarios.

## 8.7 Summary for Screening-Level Ecological Risk Assessment

Risks to ecological receptors from the potential emission of COPCs and ROPCs result from exposure to and ecological toxicity of the COPCs and ROPCs. The SLERA will utilize the estimated emission rates (Section 5) and results of fate and transport modeling (Section 6) to calculate potential ecological receptor exposure to COPCs and ROPCs. This exposure information is combined with toxicity data to estimate the potential for adverse effects to terrestrial and aquatic organisms and populations in the vicinity of the WTP.

The SLERA will use conservative exposure assumptions to compensate for the high level of uncertainty associated with conducting a risk assessment for a facility that is still in the final design phases. The PRA will include a qualitative uncertainty analysis. The exact procedures that may be used to identify and evaluate the primary sources of uncertainty in the FRA will be determined at a later time.

The FRA will include estimated emissions based on engineering calculations (e.g., PT system emissions and vapor-phase organic emissions from WTP process cells) and environmental performance

demonstration tests for the LAW and HLW vitrification systems. Based on the results of the environmental performance demonstration tests, the FRA may involve running new models, modeling additional chemicals, or changing model parameters. Information that will require updating in the FRA, as specified in the WTP DWP (WA7890008967), includes:

- Toxicity data current at the time of the submittal
- Compounds newly identified, or updated emissions data from current waste characterization and emission testing
- Air modeling updated to include stack gas parameters based on most current emissions testing and current WTP unit design
- Physical/transport properties of constituents current at the time of the submittal
- Process description based on current WTP unit design
- Emissions data and all supporting calculations based on current WTP unit design
- Update of receptor locations based on land use or land use zoning, changes, if any

If the risk goals are exceeded in the PRA or the FRA, additional site-specific data will be evaluated for use in the assessments, subject to Ecology approval.

## 8.8 References

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**Table 8-1 Threatened and Endangered Species at Hanford**

| Common Name | Scientific Name | Habitat | Federal Status [(a)] | State Status [(a)] |
|---|---|---|---|---|
| **Plants** | | | | |
| Awned Halfchaff Sedge | *Lipocarpha (= Hemicarpha) aristulata* | Riparian | - | Threatened |
| Beaked Spike-Rush | *Eleocharis rostellata* | Priest Rapids | - | Sensitive |
| Canadian St. John's Wort | *Hypericum majus* | Riparian | - | Sensitive |
| Chaffweed | *Anagallis (=Centunculus) minimus* | Riparian<br>Wetlands | - | Threatened |
| Columbia Milkvetch | *Astragalus columbianus* | Umtanum Ridge<br>Shrub-steppe | Species of concern | Sensitive |
| Columbia Yellowcress | *Rorippa columbiae* | Riparian | Species of concern | Endangered |
| Coyote Tobacco | *Nicotiana attenuata* | Cold Creek Valley | - | Sensitive |
| Desert Cryptantha | *Cryptantha scoparia* | ALE Reserve | - | Sensitive |
| Desert Dodder | *Cuscuta denticulata* | Whaluke Slope | - | Threatened |
| Desert Evening Primrose | *Oenothera caespitosa* | Cobbled soil near Columbia River (China Bar) | - | Sensitive |
| Dwarf Evening Primrose | *Camissonia (= Oenothera) pygmaea* | Shrub-steppe | - | Sensitive |
| Fuzzytongue Penstemon | *Penstemon eriantherus whitedii* | White Bluffs | - | Sensitive |
| Geyer's Milkvetch | *Astragalus geyeri* | Whaluke Slope | - | Threatened |
| Grand Redstem | *Ammannia robusta* | Riparian | - | Threatened |
| Gray Cryptantha | *Cryptantha leucophaea* | Sand dunes<br>Shrub-steppe | Species of concern | Sensitive |
| Great Basin Gilia | *Aliciella (=Gilia) leptomeria* | Gable Mountain<br>Whaluke Slope | - | Threatened |
| Hoover's Desert Parsley | *Lomatium tuberosum* | Umtanum Ridge<br>Basalt outcrops | Species of concern | Sensitive |
| Loeflingia | *Loeflingia squarrosa var. squarrosa* | Gable Mountain | - | Threatened |
| Lowland Toothcup | *Rotala ramosior* | Riparian | - | Threatened |
| Piper's Daisy | *Erigeron piperianus* | Shrub-steppe | - | Sensitive |
| Rosy Pussypaws | *Calyptridium roseum* | Gable Mountain | - | Threatened |

**Table 8-1 Threatened and Endangered Species at Hanford**

| Common Name | Scientific Name | Habitat | Federal Status [a] | State Status [a] |
|---|---|---|---|---|
| Small-Flowered Evening-Primrose | *Camissonia (= Oenothera) minor* | FEALE Reserve<br>Gable Mountain<br>200 Area (gravel pit) | - | Sensitive |
| Snake River Cryptantha | *Cryptantha spiculifera (= C. interrupta)* | White Bluffs | - | Sensitive |
| Suksdorf's Monkey Flower | *Mimulus suksdorfii* | Gable Mountain<br>Gable Butte<br>Vernita grade | - | Sensitive |
| Umtanum Desert Buckwheat | *Eriogonum codium* | Umtanum Ridge | Candidate | Endangered |
| White Bluffs Bladderpod | *Lesquerella tuplashensis* | White Bluffs | Candidate | Threatened |
| White Eatonella | *Eatonella nivea* | Whaluke Slope | - | Threatened |
| **Mollusks** | | | | |
| California Floater | *Anodonta californiensis* | River and streams - Slow current | Species of concern | Candidate |
| Great Columbia River Spire Snail | *Fluminicola columbiana* | Hanford Reach | Species of concern | Candidate |
| Shortfaced Lanx | *Fisherola nuttalli* | Hanford Reach | - | Candidate |
| **Insects** | | | | |
| Columbia River Tiger Beetle [b] | *Cicindela columbica* | see footnote [b] | - | Candidate |
| Silver-Bordered Fritillary | *Boloria selene atrocostalis* | Riparian | - | Candidate |
| **Fish** | | | | |
| Bull Trout [c] | *Salvelinus confluentus* | Hanford Reach | Threatened | Candidate |
| Leopard Dace [c] | *Rhinichthys flacatus* | Hanford Reach | - | Candidate |
| Mountain Sucker [c] | *Catastomus platyrhynchus* | Hanford Reach | - | Candidate |
| Pacific lamprey | *Lampetra tridentata* | Hanford Reach | Species of concern | - |
| River Lamprey [c] | *Lampetra ayresi* | Hanford Reach | Species of concern | Candidate |
| Spring-Run Chinook Salmon | *Oncorhynchus tshawytscha* | Hanford Reach | Endangered | Candidate |
| Steelhead | *Oncorhynchus mykiss* | Hanford Reach | Threatened | Candidate |
| **Amphibians and Reptiles** | | | | |
| Sagebrush Lizard | *Sceloporus graciosus* | Shrub-steppe (low elevations and sandy areas) | Species of concern | Candidate |

**Table 8-1 Threatened and Endangered Species at Hanford**

| Common Name | Scientific Name | Habitat | Federal Status [a] | State Status [a] |
|---|---|---|---|---|
| Striped Whipsnake | *Masticophis taeniatus* | Shrub-steppe | - | Candidate |
| Western Toad | *Bufo boreas* | Riparian | Species of concern | Candidate |
| **Birds** | | | | |
| American White Pelican | *Pelecanus erythrorhynchos* | Riparian | - | Endangered |
| Bald Eagle [d] | *Haliaeetus leucocephalus* | Riparian | Species of concern | Sensitive |
| Burrowing Owl | *Athene cunicularia* | Shrub-steppe | Species of concern | Candidate |
| Common Loon | *Gavia immer* | Riparian | - | Sensitive |
| Ferruginous Hawk | *Buteo regalis* | Transmission line towers<br>Shrub-steppe | Species of concern | Threatened |
| Flamulated Owl [c] | *Otus flammeolus* | Shrub-steppe | - | Candidate |
| Golden Eagle | *Aquila chrysaetos* | Shrub-steppe | - | Candidate |
| Greater Sage Grouse | *Centrocercus urophasianus* | Shrub-steppe | Candidate | Threatened |
| Lewis's Woodpecker [c] | *Melanerpes lewisi* | Riparian<br>Shrub-steppe | - | Candidate |
| Loggerhead Shrike | *Lanius ludovicianus* | Shrub-steppe | Species of concern | Candidate |
| Merlin | *Falco columbarius* | Riparian<br>Shrub-steppe | - | Candidate |
| Northern Goshawk [c] | *Accipter gentilis* | Shrub-steppe | Species of concern | Candidate |
| Olive-Sided Flycatcher | *Contopus cooperi* | Riparian | Species of concern | - |
| Peregrine Falcon | *Falco peregrinus* | Riparian<br>Shrub-steppe | Species of concern | Sensitive |
| Sage Sparrow | *Amphispiza belli* | Shrub-steppe | - | Candidate |
| Sage Thrasher | *Oreoscoptes montanus* | Shrub-steppe | - | Candidate |
| Sandhill Crane | *Grus canadensis* | Islands<br>Riparian<br>Shrub-steppe | - | Endangered |
| Western Grebe | *Aechmorus occidentalis* | Riparian | - | Candidate |
| **Mammals** | | | | |
| Black-Tailed Jackrabbit | *Lepus californicus* | Shrub-steppe | - | Candidate |

**Table 8-1 Threatened and Endangered Species at Hanford**

| Common Name | Scientific Name | Habitat | Federal Status [(a)] | State Status [(a)] |
|---|---|---|---|---|
| Merriam's Shrew | *Sorex merriami* | Shrub-steppe | - | Candidate |
| Townsend's Ground Squirrel | *Spermophilus townsendii* | Benton County Shrub-steppe | Species of concern | Candidate |
| Washington Ground Squirrel [(c)] | *Spermophilus washingtoni* | Shrub-steppe | Candidate | Candidate |
| White-Tailed Jackrabbit | *Lepus townsendii* | Shrub-steppe | - | Candidate |

(a) "-" indicates species is not listed a endangered, threatened, candidate, sensitive, or of concern.

Endangered = Species in danger of extinction within all or a significant portion of its range.

Threatened = Species likely to become endangered in the foreseeable future.

Candidate = Species that are believed to qualify for threatened or endangered species status, but for which listing proposals have not been prepared.

Sensitive = Taxa that are vulnerable or declining and could become endangered or threatened without active management or removal of threats.

Species of concern = Species that are not currently listed or candidates under the Endangered Species Act, but are of conservation concern within specific US Fish and Wildlife Service regions.

(b) Probable, but not observed, on the Hanford Site.

(c) Reported, but seldom observed, on the Hanford Site.

(d) Reclassified January 2008.

Refs: PNNL. 2010. *Hanford Site Environmental Report for Calendar Year 2009*, PNNL-19455, September 2010. Pacific Northwest National Laboratory, Richland, Washington.

PNNL. 2001. *Vascular Plants of the Hanford Site*. PNNL-13688. Pacific Northwest National Laboratory, Richland, Washington.

PNL. 1993. *Habitat Types on the Hanford Site Wildlife and Plant Species of Concern*. PNL-8942 (UC-702). Pacific Northwest National Laboratory, Richland, Washington.

**Table 8-2 Policy Goals, Ecological Assessment Endpoints, Measures, and Decision Rules for 200 Area and Surroundings**

| Policy Goals | Assessment Endpoint | Measures | Decision Rule |
|---|---|---|---|
| **Policy Goal 1:** The conservation of threatened and endangered species and their critical habitats. | **Assessment Endpoint 1:** Protection of individuals of state or federally designated threatened or endangered (T&E) species.<br><br>Endpoint species: redtailed hawk. | **Measure 1:** Modeled contaminant concentrations in prey (such as, deer mouse, western meadowlark, Great Basin pocket mouse, mourning dove, and fish) based on modeled concentrations of vapors in air and particulates, depositions of contaminant particulates to soil and surface water, and measured concentrations of contaminants in abiotic media. These concentrations are used to evaluate exposure of threatened and endangered predators. Chronic exposure concentrations and doses associated with no adverse effect on survival and reproduction. | **Decision Rule for Assessment Endpoint 1:** If threatened or endangered species are not present, or exposure point concentrations in the media do not contribute to the chronic NOAEL, then it is indicated that the contaminant alone is unlikely to cause adverse ecological effects and, therefore, the threatened or endangered species should be preserved. If the HQ $\geq$0.25, lines of evidence will be evaluated to determine the potential for ecological risk and the need for any additional measurements or calculations. |
| **Policy Goal 2:** The protection of terrestrial populations and ecosystems. | **Assessment Endpoint 2:** Stable plant community for erosion control and energy production.<br><br>Endpoint species: cheatgrass, rabbitbrush. | **Measure 2**: Modeled concentrations of vapors in air and particulates and depositions of contaminant particulates to soil. Chronic exposure concentrations associated with no adverse effect on survival and reproduction. | **Decision Rule for Assessment Endpoint 2:** If the HQ is <0.25, then it is indicated that the contaminant alone is unlikely to cause adverse ecological effects and, therefore, the plant populations and communities are maintained. If the HQ $\geq$0.25, lines of evidence will be evaluated to determine the potential for ecological risk and the need for any additional measurements or calculations. |
| **Policy Goal 2:** The protection of terrestrial populations and ecosystems. | **Assessment Endpoint 3:** Stable soil-dwelling invertebrate community for nutrient and energy processing.<br><br>Endpoint species: earthworms and darkling beetles. | **Measure 3:** Modeled concentrations of vapors in air and particulates and depositions of contaminant particulates to soil. Chronic exposure concentrations associated with no adverse effect on survival and reproduction. | **Decision Rule for Assessment Endpoint 3:** If the HQ is <0.25, then it is indicated that the contaminant alone is unlikely to cause adverse ecological effects and, therefore, the terrestrial invertebrate community is maintained. If the HQ $\geq$0.25, lines of evidence will be evaluated to determine the potential for ecological risk and the need for any additional measurements or calculations. |

**Table 8-2 Policy Goals, Ecological Assessment Endpoints, Measures, and Decision Rules for 200 Area and Surroundings**

| Policy Goals | Assessment Endpoint | Measures | Decision Rule |
|---|---|---|---|
| | **Assessment Endpoint 4:** Stable populations of herbivorous animals.<br><br>Endpoint species: mammals - mule deer; birds - mourning dove. | **Measure 4:** Modeled contaminant concentrations in food chain (such as, plants) based on modeled concentrations of vapors in air and particulates and depositions of contaminant particulates to soil. Chronic exposure doses associated with no adverse effect on survival and reproduction. | **Decision Rule for Assessment Endpoint 4:** If the HQ is <0.25, then it is indicated that the contaminant alone is unlikely to cause adverse ecological effects and, therefore, populations of the herbivores (such as, mule deer and mourning dove) are maintained. If the HQ ≥0.25, lines of evidence will be evaluated to determine the potential for ecological risk and the need for any additional measurements or calculations. |
| | **Assessment Endpoint 5:** Stable populations of animals that eat both plants and animals (omnivores).<br><br>Endpoint species: bird - western meadowlark. | **Measure 5:** Modeled contaminant concentrations in earthworms, plants, and other prey based on modeled concentrations of vapors in air and particulates and depositions of contaminant particulates to soil. Chronic exposure doses associated with no adverse effect on survival and reproduction. | **Decision Rule for Assessment Endpoint 5:** If the HQ is <0.25, then it is indicated that the contaminant alone is unlikely to cause adverse ecological effects and, therefore, populations of omnivores (such as, western meadowlark) are maintained. If the HQ ≥0.25, lines of evidence will be evaluated to determine the potential for ecological risk and the need for any additional measurements or calculations. |
| **Policy Goal 2:** The protection of terrestrial populations and ecosystems. | **Assessment Endpoint 6:** Stable populations of terrestrial predators.<br><br>Endpoint species: mammal - coyote; bird - burrowing owl and red-tailed hawk. | **Measure 6:** Modeled contaminant concentrations in prey (such as, western meadowlark and Great Basin pocket mouse) based on modeled concentrations of vapors in air and particulates and depositions of contaminant particulates to soil. These concentrations are used to evaluate exposure of predators. Chronic exposure doses associated with no adverse effect on survival and reproduction. | **Decision Rule for Assessment Endpoint 6:** If the HQ is <0.25, then it is indicated that the contaminant alone is unlikely to cause adverse ecological effects and, therefore, populations of terrestrial predators are maintained. If the HQ ≥0.25, lines of evidence will be evaluated to determine the potential for ecological risk and the need for any additional measurements or calculations. |

**Table 8-2 Policy Goals, Ecological Assessment Endpoints, Measures, and Decision Rules for 200 Area and Surroundings**

| Policy Goals | Assessment Endpoint | Measures | Decision Rule |
|---|---|---|---|
| **Policy Goal 3:** The protection of aquatic populations and ecosystems. | **Assessment Endpoint 7:** Stable populations of sediment-dwelling organisms.<br><br>Endpoint species: clams, insects, snails, and worms. | **Measure 7:** Modeled sediment contaminant concentrations from dispersion and deposition. Chronic exposure concentrations associated with no adverse effect on survival and reproduction. | **Decision Rule for Assessment Endpoint 7:** If the HQ is <0.25, then it is indicated that the contaminant alone is unlikely to cause adverse ecological effects and, therefore, populations of sediment-dwelling organisms are maintained. If the HQ ≥0.25, lines of evidence will be evaluated to determine the potential for ecological risk and the need for any additional measurements or calculations. |
| | **Assessment Endpoint 8:** Stable populations of planktivorous fish and small invertebrates.<br><br>Endpoint species: water fleas and other invertebrates. | **Measure 8:** Modeled surface water contaminant concentrations. Chronic exposure concentrations associated with no adverse effect on survival and reproduction. | **Decision Rule for Assessment Endpoint 8:** If the HQ is <0.25, then it is indicated that the contaminant alone is unlikely to cause adverse ecological effects and, therefore, populations of small invertebrates are maintained. If the HQ ≥0.25, lines of evidence will be evaluated to determine the potential for ecological risk and the need for any additional measurements or calculations. |
| **Policy Goal 3:** The protection of aquatic populations and ecosystems. | **Assessment Endpoint 9:** Stable waterfowl and shorebird populations.<br><br>Endpoint species: Canada goose, spotted sandpiper. | **Measure 9:** Modeled contaminant concentrations in benthic invertebrates or aquatic plants based on modeled contaminant concentrations in surface water or sediments from dispersion and deposition. These concentrations are used to evaluate exposure of predators. Chronic exposure doses associated with no adverse effect on survival and reproduction. | **Decision Rule for Assessment Endpoint 9:** If the HQ is <0.25, then it is indicated that the contaminant alone is unlikely to cause adverse ecological effects and, therefore, populations of waterfowl and shorebirds are maintained. If the HQ ≥0.25, lines of evidence will be evaluated to determine the potential for ecological risk and the need for any additional measurements or calculations. |
| | **Assessment Endpoint 10:** Stable populations of large carnivorous fish population for regulation.<br><br>Endpoint species: salmon, bass, channel catfish. | **Measure 10:** Modeled surface water and sediment contaminant concentrations. Chronic exposure concentrations associated with no adverse effect on survival and reproduction. | **Decision Rule for Assessment Endpoint 10:** If the HQ is <0.25, then it is indicated that the contaminant alone is unlikely to cause adverse ecological effects and, therefore, populations of large carnivorous fish are maintained. If the HQ ≥0.25, lines of evidence will be evaluated to determine the potential for ecological risk and the need for any additional measurements or calculations. |

**Table 8-2 Policy Goals, Ecological Assessment Endpoints, Measures, and Decision Rules for 200 Area and Surroundings**

| Policy Goals | Assessment Endpoint | Measures | Decision Rule |
|---|---|---|---|
| | **Assessment Endpoint 11:** Stable fish-eating terrestrial predator populations for population regulation.<br><br>Endpoint species: mammal - mink; birds - great blue heron, bald eagle. | **Measure 11:** Modeled contaminant concentrations in large carnivorous fish and planktivorous fish and small invertebrates based on modeled surface water and sediment concentrations. These concentrations are used to evaluate exposure of predators. Chronic exposure doses associated with no adverse effect on survival and reproduction. | **Decision Rule for Assessment Endpoint 11:** If the HQ is <0.25, then it is indicated that the contaminant alone is unlikely to cause adverse ecological effects and, therefore, populations of fish-eating terrestrial predators are maintained. If the HQ ≥0.25, lines of evidence will be evaluated to determine the potential for ecological risk and the need for any additional measurements or calculations. |

T&E = Threatened and endangered.

NOAEL = No observed adverse effects level.

HQ = Hazard quotient.

**Table 8-3 Ecological Assessment TRV Surrogates**

| Constituent | Surrogate |
|---|---|
| 5-nitroacenaphthene (CAS #602-87-9) | acenaphthene (CAS #83-32-9) |
| methyl isocyanate (CAS #624-83-9) | acrylonitrile (CAS #107-13-1) |
| trichlorofluoroethane (CAS #27154-33-2) | trichlorofluoromethane (CAS #75-69-4) |
| 2,4-toluene diisocyanate (CAS #584-84-9) | 2,4-dinitrotoluene (CAS #121-14-2) |
| hydrogen chloride (CAS #7647-01-0) | chlorine (CAS #7782-50-5) |

**Figure 8-1 Ecological Resources Conceptual Exposure Model**

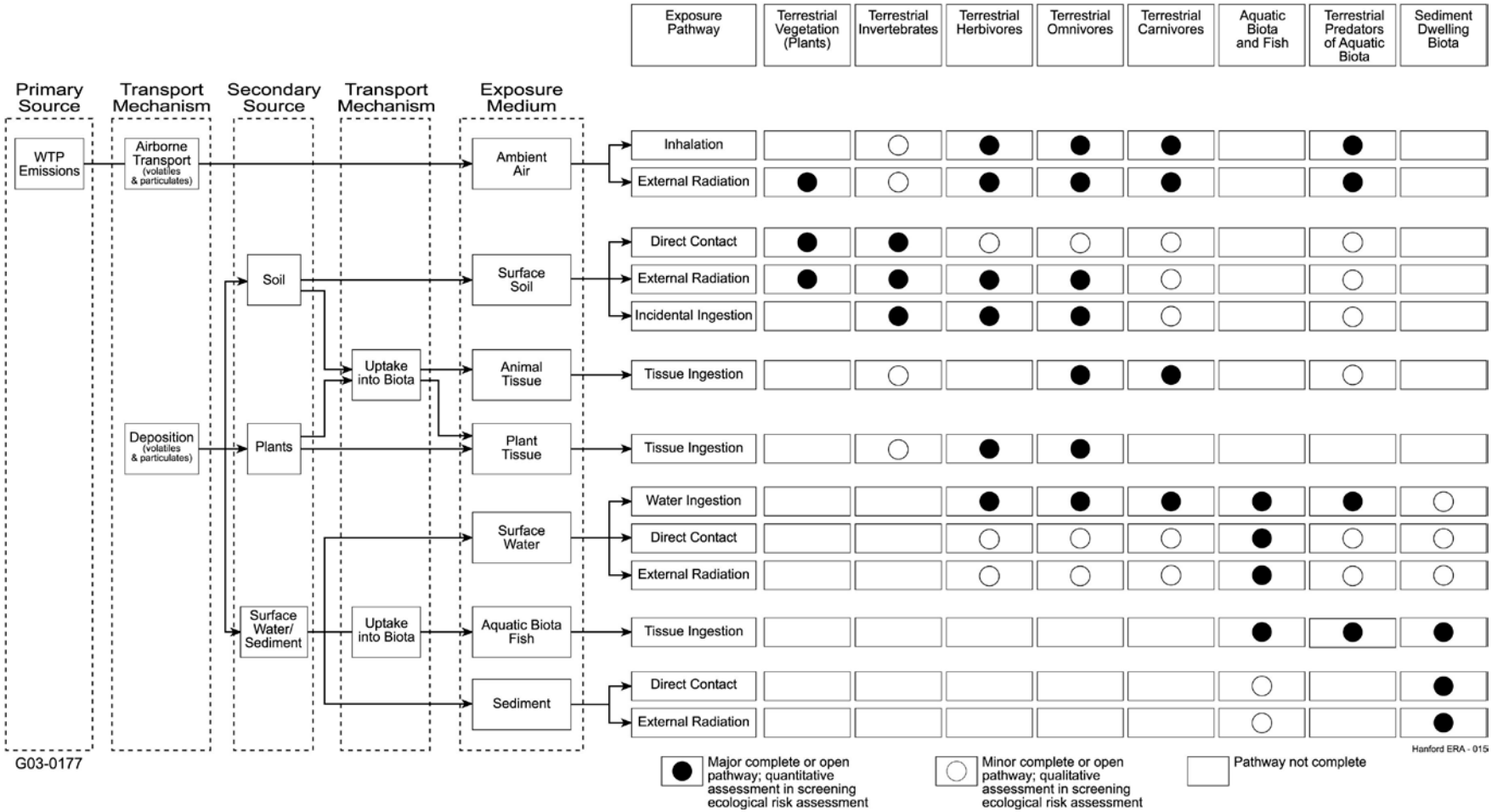


**Figure 8-2 Recreation and Wildlife Areas and the Hanford Reach**

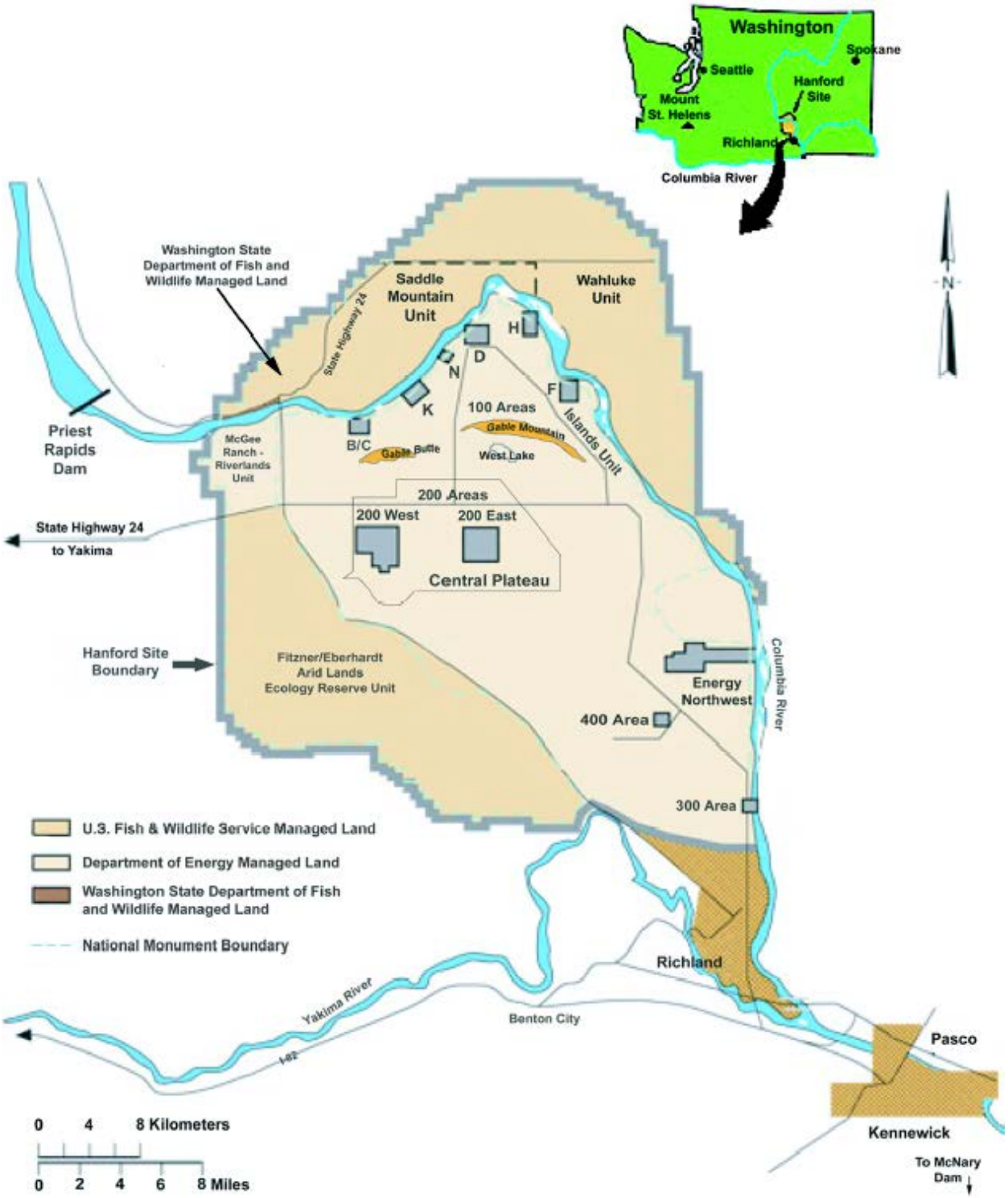
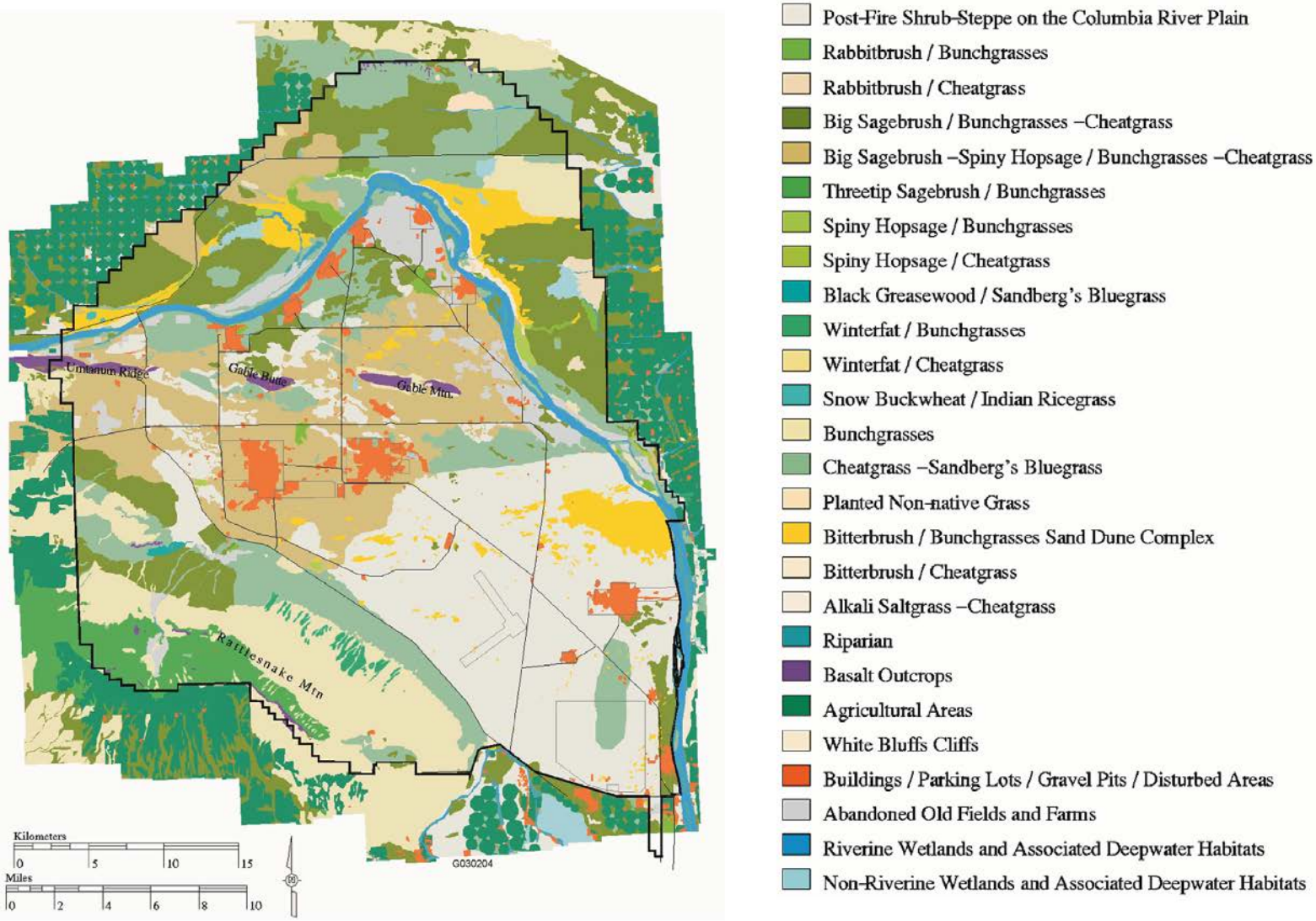


**Figure 8-3 Regional Geography, Water Bodies, Roads, and Communities**

©2010 MapQuest - Portions ©2010 NAVTEQ, Intermap

**Figure 8-4 Vegetation Types of the Hanford Site**




BHI:rpp 04/22/96 clup/brmap d02.aml Database: 07-DEC-1998

**Figure 8-5 Selected Raptor Nesting and Perching Locations on the Hanford Site**

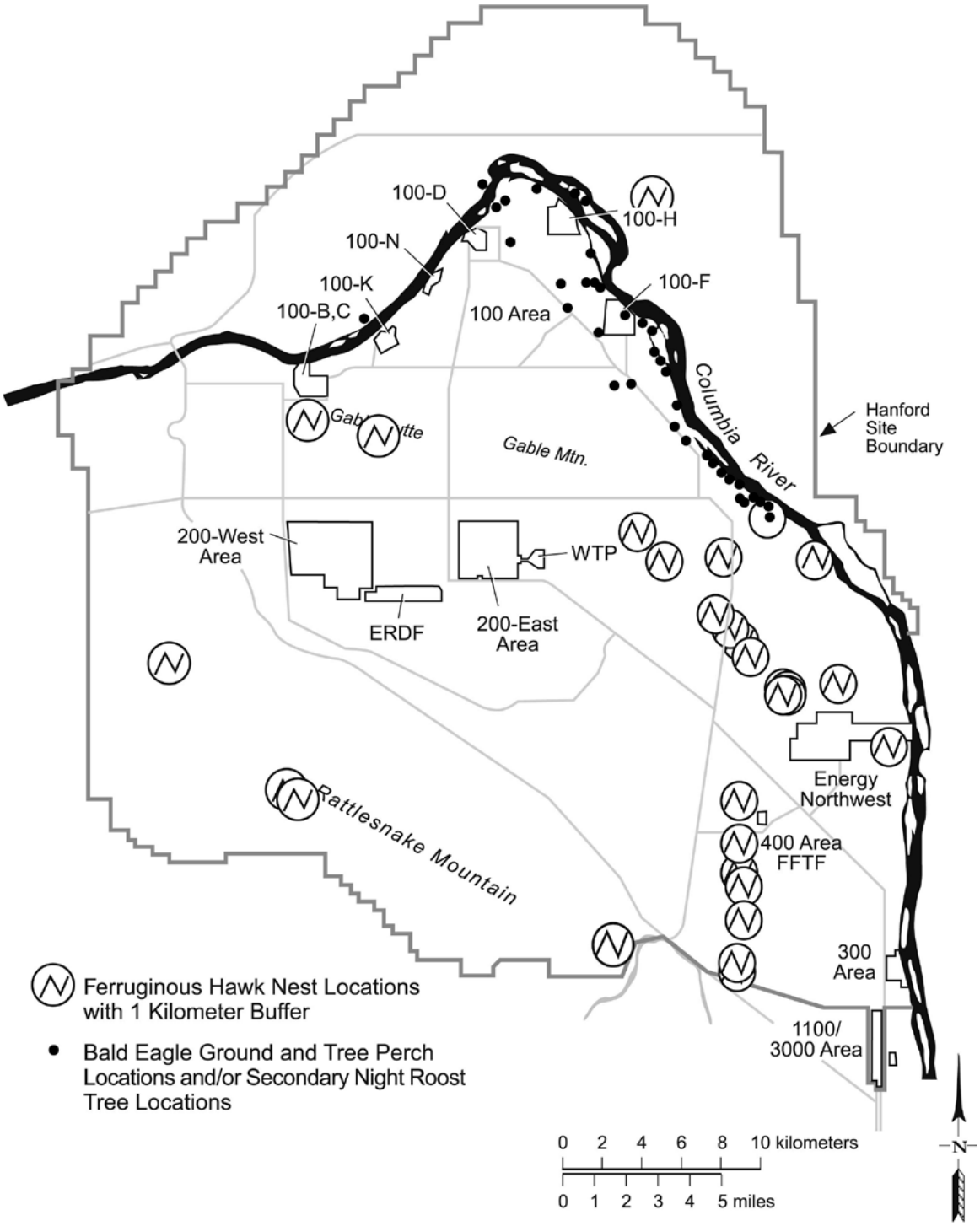


**Figure 8-6 WTP Areas Vegetation Types (Simplified)**

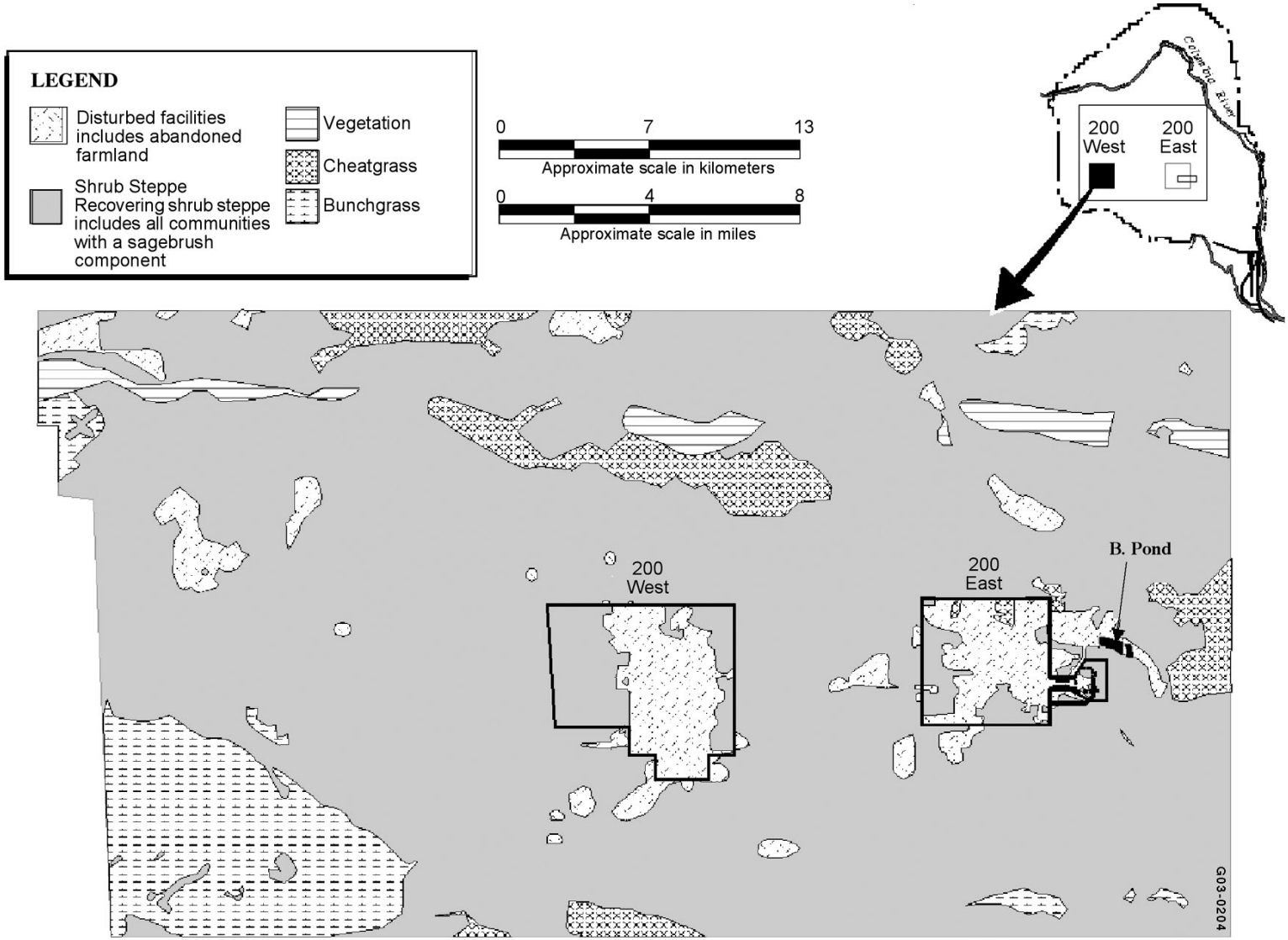


**Figure 8-7 Fall Chinook Salmon Spawning Areas Along the Columbia River**

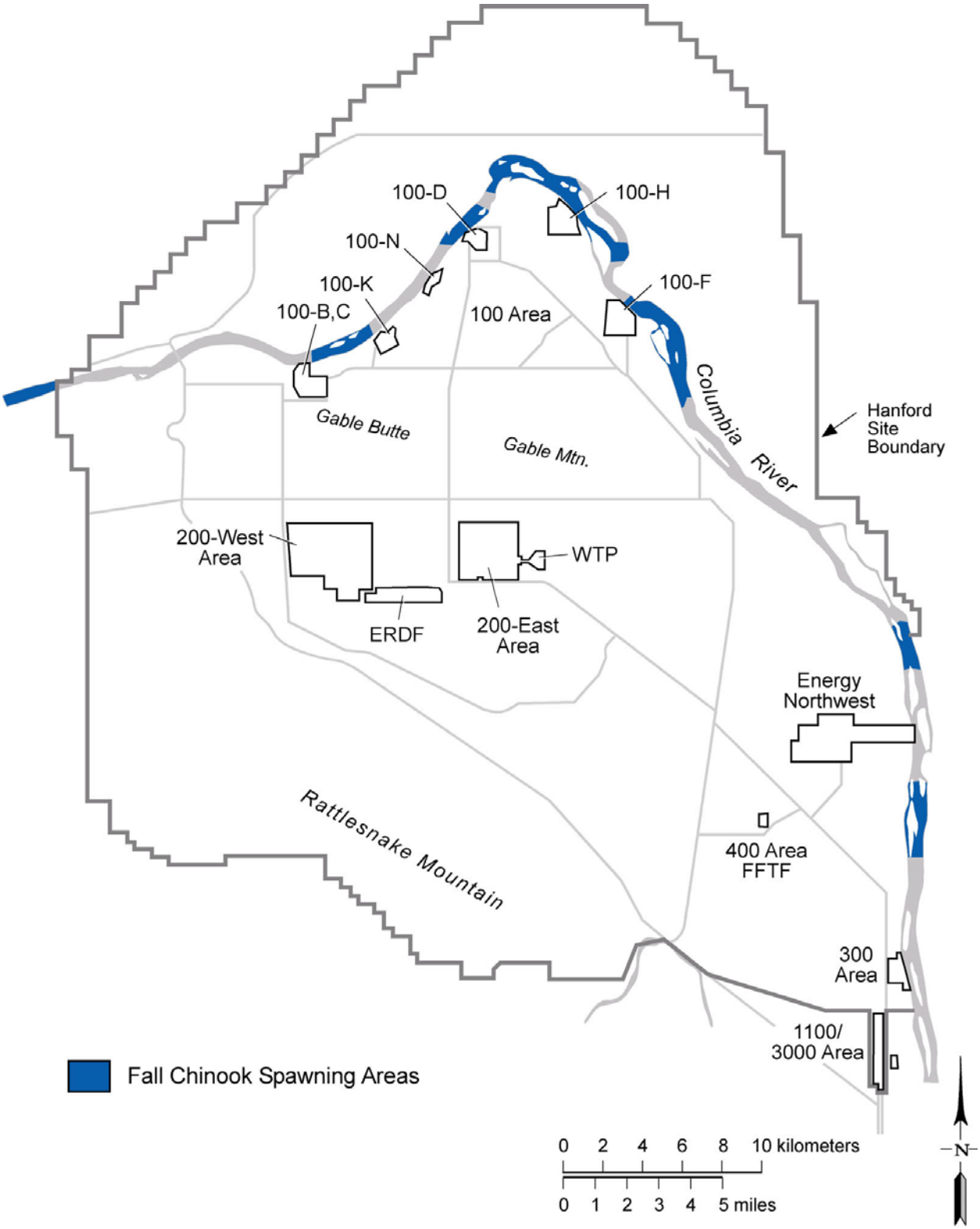
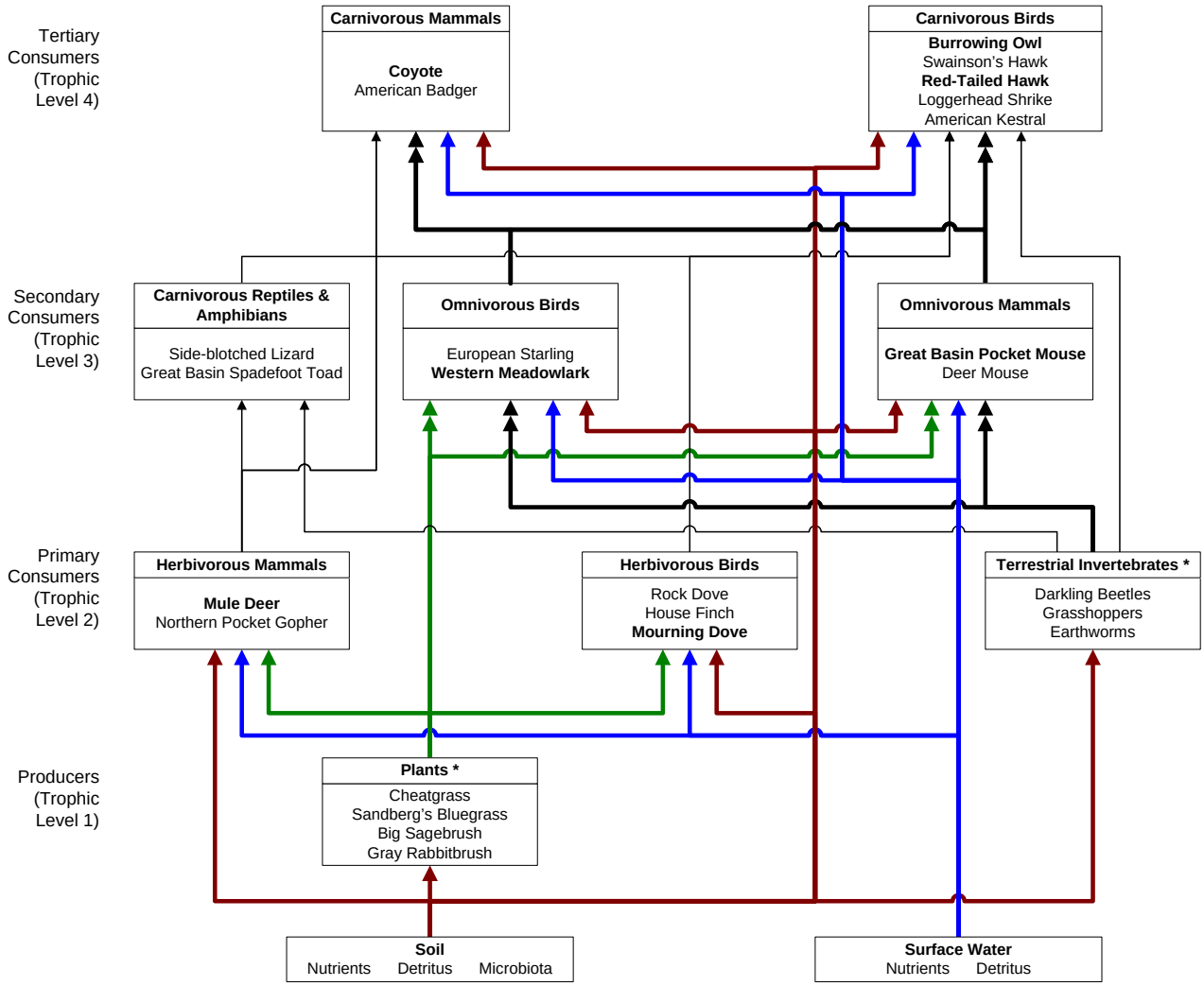


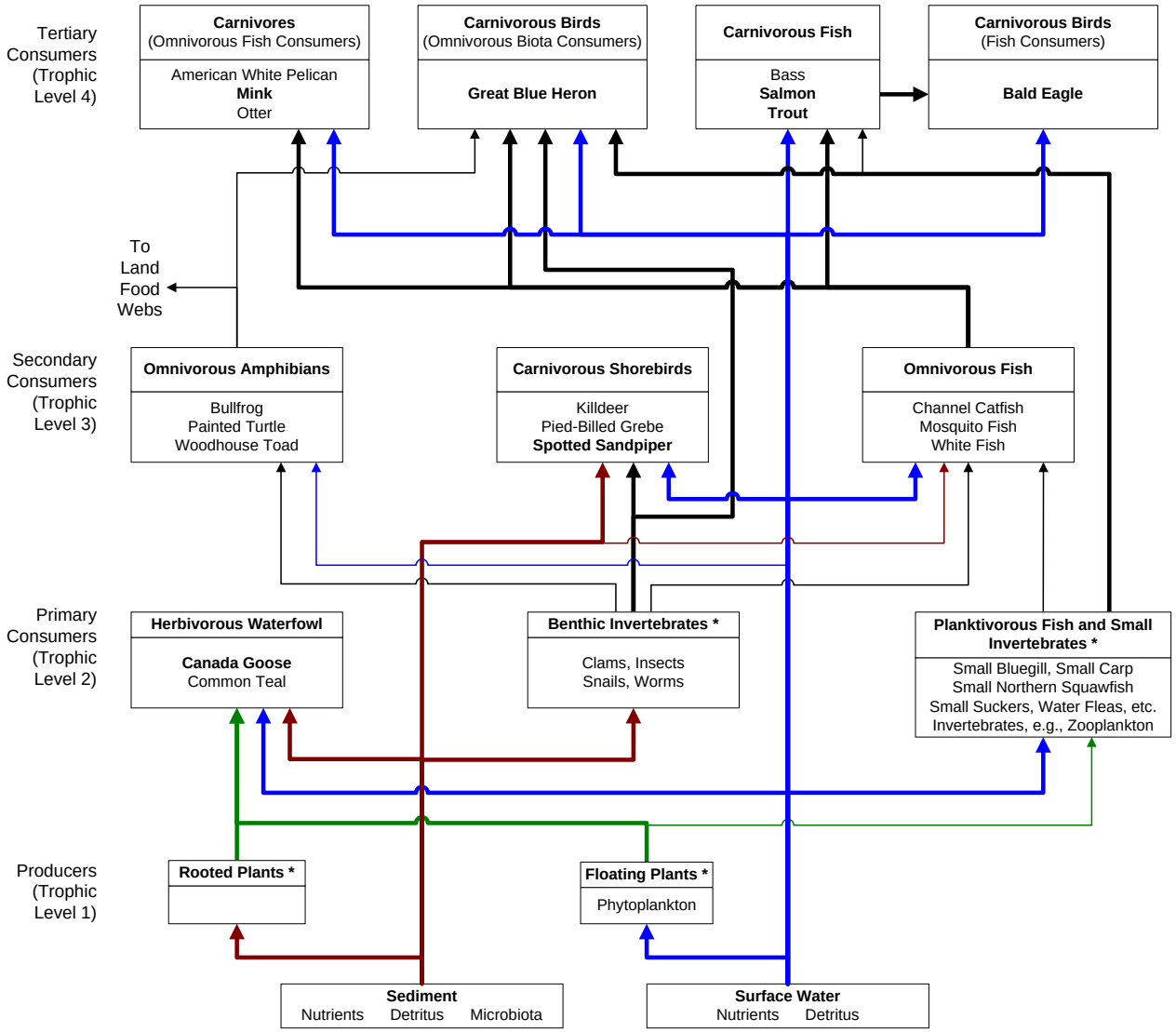
**Figure 8-8 Trophic Levels and Measurement Receptor Species Evaluated in the Hanford Site and Vicinity Terrestrial Conceptual Exposure Model**




Bold-faced type indicates measurement receptors for which exposure will be evaluated quantitatively.
Heavy lines indicate exposure pathways that will be evaluated quantitatively.
Thin lines indicate exposure pathways that will not be evaluated quantitatively.
Double arrow heads indicate food source is one of two potential exclusive food sources (see Section 8.2.1).
Line color indicates ingestion pathway (brown=soil, blue=water, green=plant, black=prey).

* Species specific measurement receptors not identified because the group is evaluated on a community level.

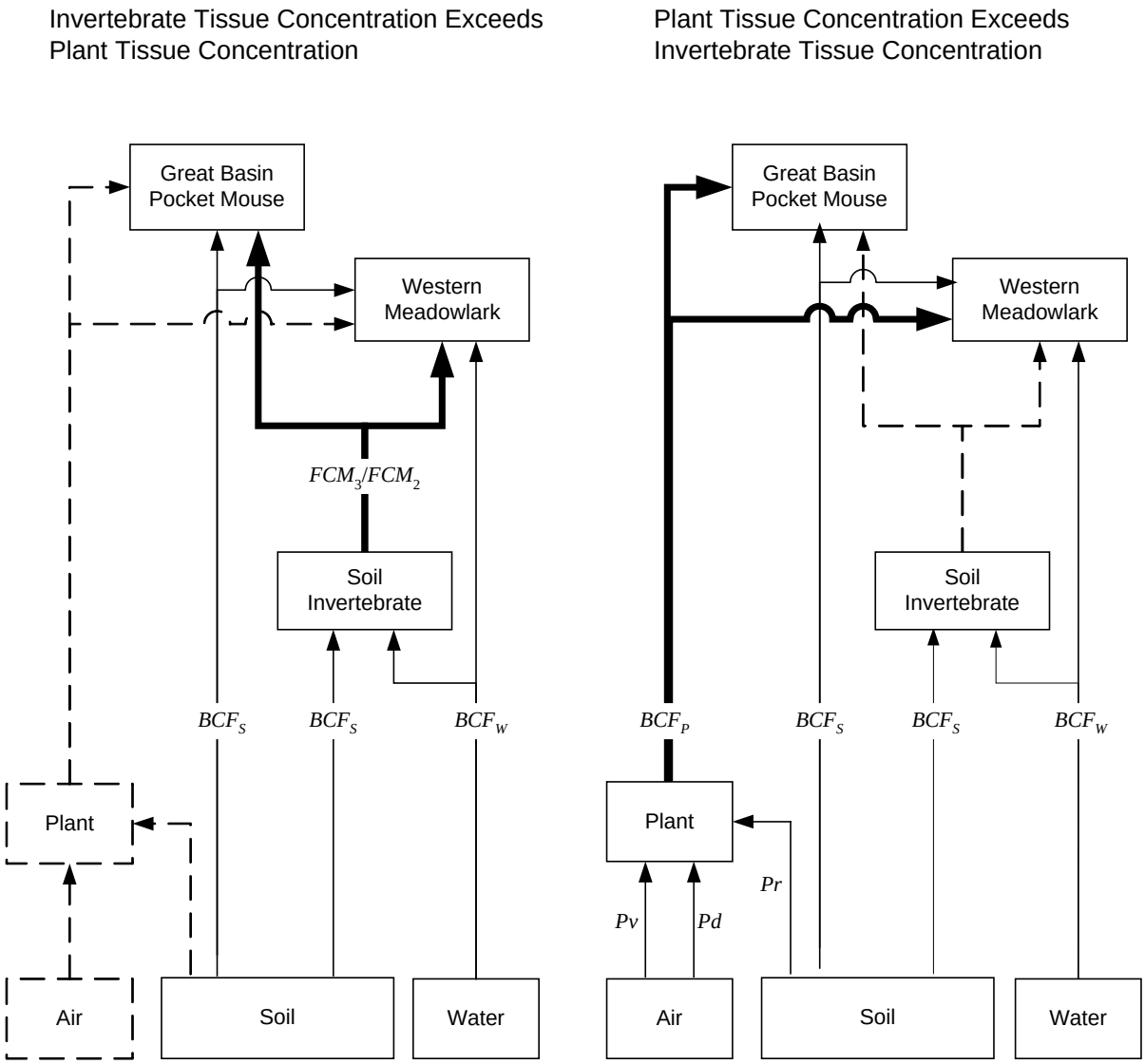
**Figure 8-9 Trophic Levels and Measurement Receptor Species Evaluated in the Columbia River Aquatic Conceptual Exposure Model**




Bold-faced type indicates measurement receptors for which exposure will be evaluated quantitatively.
Heavy lines indicate exposure pathways that will be evaluated quantitatively.
Thin lines indicate exposure pathways that will not be evaluated quantitatively.
Line color indicates ingestion pathway (brown=soil, blue=water, green=plant, black=prey).

* Species specific measurement receptors not identified because the group is evaluated on a community level.

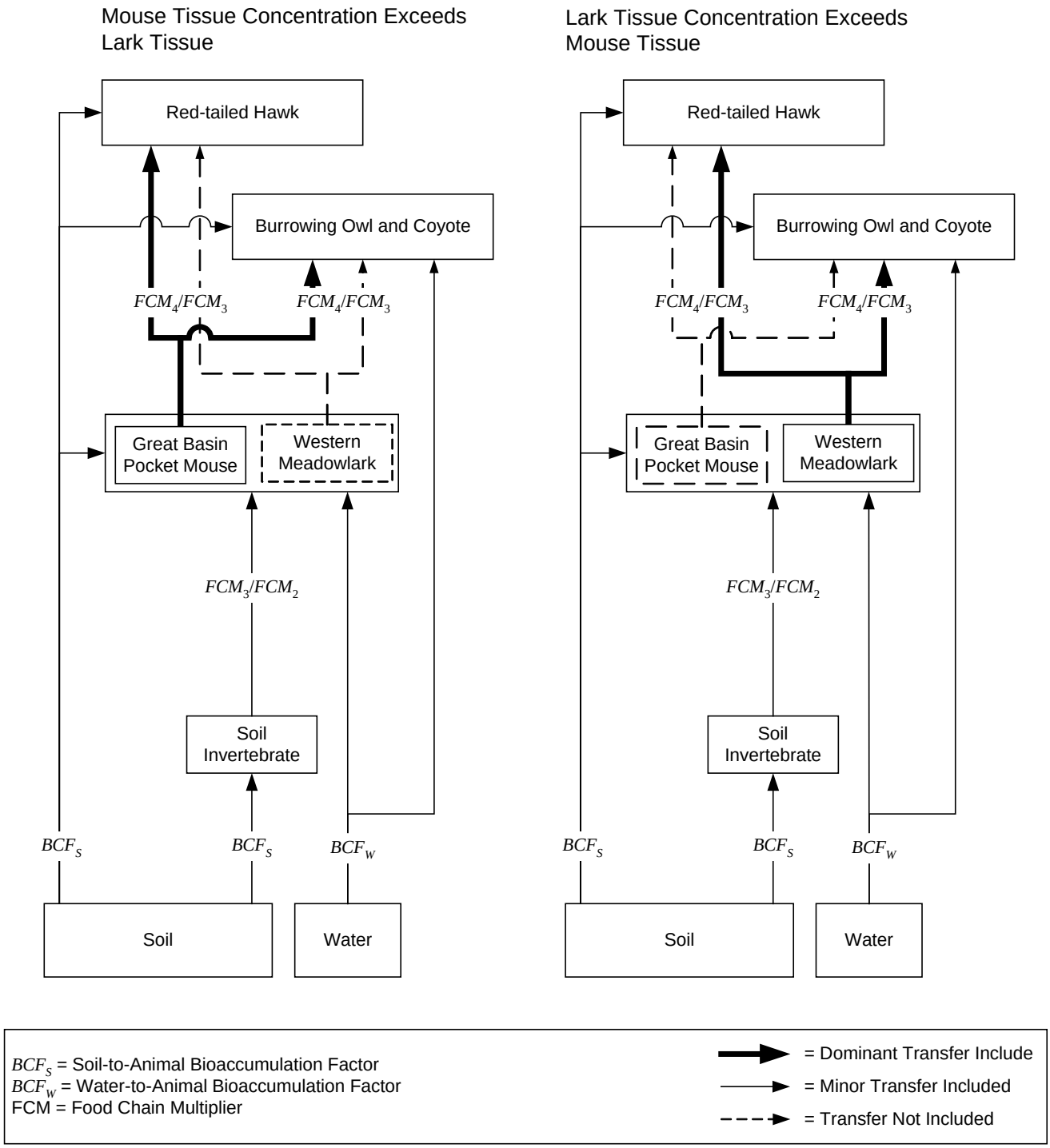
**Figure 8-10 Exclusive Diets for Omnivores**




$BCF_{INV}$ = Invertebrate-to-Animal Bioconcentration Factor
$BCF_S$ = Soil-to-Animal Bioconcentration Factor
$BCF_P$ = Plant-to-Animal Bioconcentration Factor
$BCF_W$ = Water-to-Animal Bioconcentration Factor
$FCM$ = Food Chain Multiplier
$Pd$ = Air-to-Plant Uptake Factor from Particles Deposited on Leaf Surface
$Pv$ = Air-to-Plant Uptake Factor of Vapors
$Pr$ = Soil-to-Plant Uptake Factor

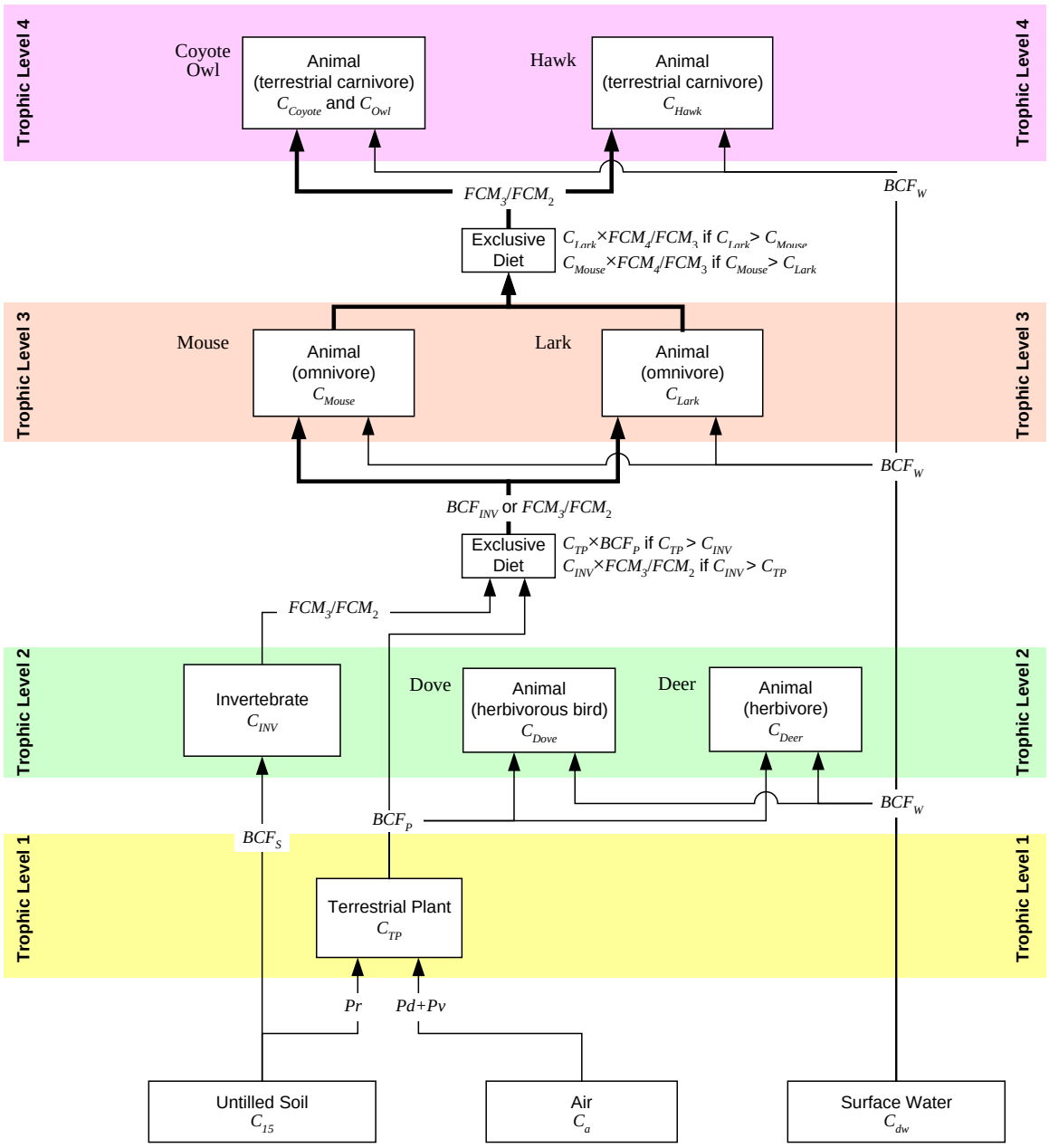
= Dominant Transfer Included
= Minor Transfer Included
= Transfer Not Included

**Figure 8-11 Exclusive Diets for Carnivores**



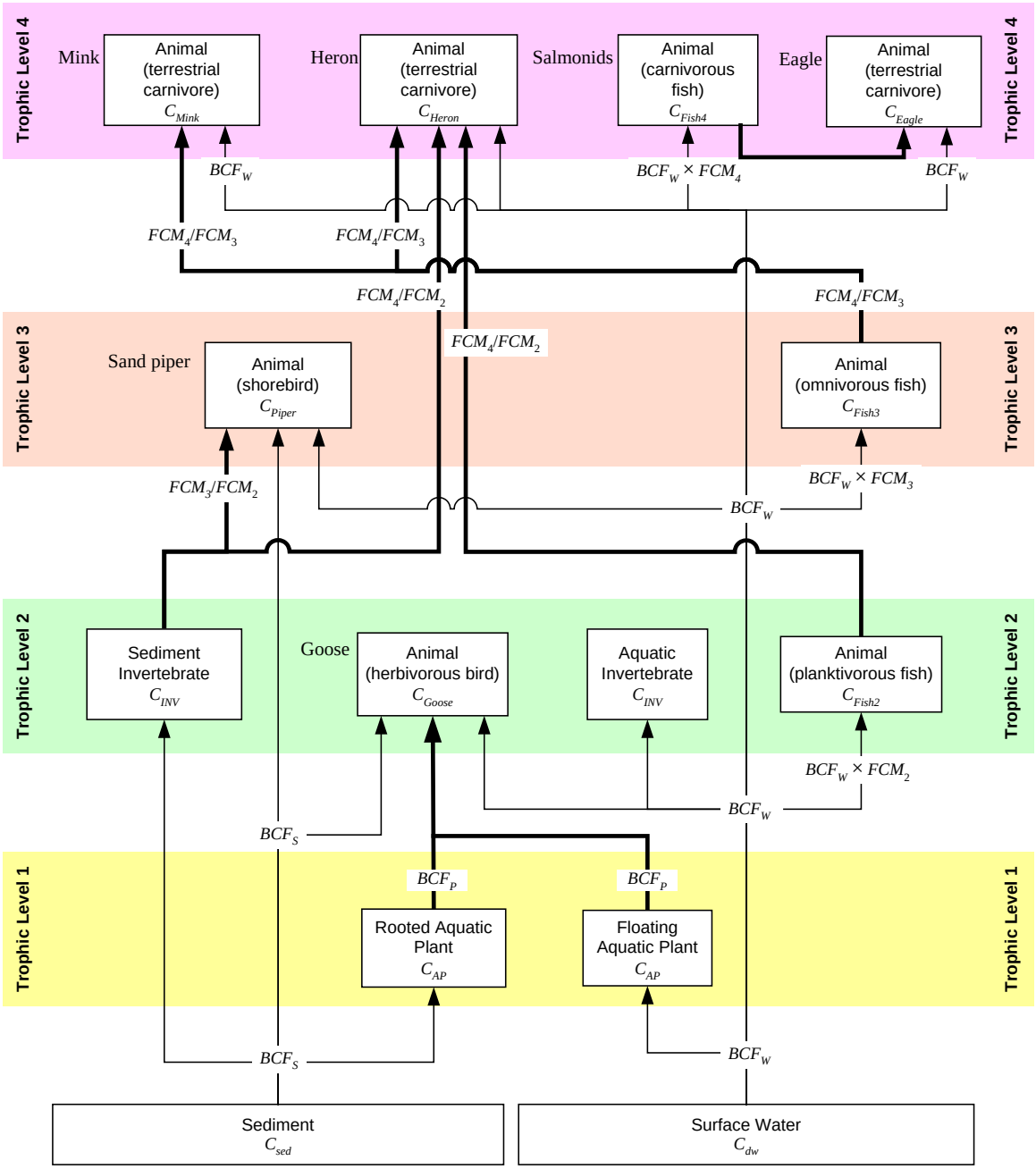

* FCMs for Top Predators Are Used for ROPCs Only

**Figure 8-12 Relationship Between Sources and Biotransfer Factors for Calculating Terrestrial Exposures**




| | | |
|---|---|---|
| $C_i$ | = Concentration in media, food source, or receptor $i$ | → = Dominant Transfer |
| $BCF_P$ | = Plant-to-Animal Bioconcentration Factor | → = Minor Transfer |
| $BCF_S$ | = Soil-to-Animal Bioconcentration Factor | |
| $BCF_W$ | = Water-to-Animal Bioconcentration Factor | |
| $FCM$ | = Food Chain Multiplier | |

**Figure 8-13 Relationship Between Sources and Biotransfer Factors for Calculating Aquatic Exposures**




| | |
|---|---|
| $C_i$ | = Concentration in media, food source, or receptor $i$ |
| $BCF_P$ | = Plant-to-Animal Bioconcentration Factor |
| $BCF_S$ | = Sediment-to-Animal Bioconcentration Factor |
| $BCF_W$ | = Water-to-Animal Bioconcentration Factor |
| $FCM$ | = Food Chain Multiplier |

→ = Dominant Transfer
→ = Minor Transfer

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# 9 Relationship of Risk Assessment to WTP

The intent of the SLRA is to provide information to help assess the impact of potential airborne emissions from the WTP to people who live near or work on or near the Hanford site, to American Indians who use resources on or near the Hanford site, and to plants and animals on or near the Hanford site. It is important that people and the environment are not harmed because potential exposures are overlooked or underestimated. It is also important to maximize the ability of the WTP to treat and immobilize tank wastes and, in doing so, minimize potential release of tank contents into the environment through leaks or spills. A balance of these goals will result through the iterative process of reviewing the RAWP, reviewing and updating environmental parameters for the SLRA and WTP engineering design, and calculating risk-based emission limits, as needed. This iterative process is shown on Figure 2-1.

During the PRA and FRA, any COPCs or ROPCs that exceed risk or hazard thresholds will be evaluated further to determine the driving factors behind the risk and the potential uncertainty associated with them. When the uncertainty associated with exposure parameters and toxicity values becomes reduced as much as possible and when there are exceedances of the thresholds, risk-based emission criteria for COPCs will be evaluated. Engineering design specifications, including changes to feed rate and acceptance criteria, may be revised based on risk-based emission limits. Each major step of the process will include review from regulatory agencies, American Indian tribes, and the public.

To better understand this iterative process, it is important to identify the relationship of the PRA and FRA (Section 9.1), the sources of potential changes that could affect the risk assessments (Section 9.2), and risk-based emission limits (Section 9.3). Each is briefly described below.

## 9.1 Relationship of the PRA and FRA

The PRA will be reviewed by regulatory agencies, by American Indian tribes, and by the public. Input from all these reviewers will be included in decisions about succeeding steps in the SLRA process, including refinement of the risk assessment assumptions for the FRA.

Both the PRA and FRA are designed to overestimate exposures to human and ecological receptors. To help make risk management decisions, predicted risks and hazards are compared to thresholds. There are thresholds for both human and ecological receptors. If the PRA indicates that total human health risks or hazards to plausible receptors are below the thresholds of 1E-05 (excess cancer risk expressed as ILCR) or 0.25 (HQ and HI), or if ESQs are less than the threshold of 0.25, the process will move on to the FRA following the environmental performance demonstration tests. Additionally, for acute exposure, the human HQ is set at 1.0.

These threshold values are summarized in Table 9-1 and described in Sections 7.4.1 and 7.4.2 (Human Health) and Section 8.4.3 (Ecological). If the PRA indicates that human risks to plausible receptors are greater than 1E-05 (ILCR), or if human noncancer HQs and HIs are greater than 0.25, or if ESQs are greater than 0.25, or if human acute HQs are greater than 1.0, a number of actions will be considered. Potential actions will include reevaluation of conservative exposure parameters for the risk assessment and reevaluation of operating conditions.

## 9.2 Sources of Potential Changes in the Risk Assessment

A risk assessment represents the status of receptors, facility, and toxicity knowledge at a point in time. If land use changes or if new site-specific data becomes available to replace default exposure assumptions, the assumptions used in the PRA may change. If there are changes in engineering design of the WTP that result in changes in emissions estimates, exposures may change; if site-specific uptake factors for the food chain become available, exposures may change; if there are revisions to toxicity data for some COPCs or ROPCs, ILCR risks and HQs and HIs for those COPCs and ROPCs may change. If any of these changes occur, the SLRA could be revisited to assess potential impacts to public welfare and the environment.

The PRA will evaluate the risks posed by the projected WTP emissions. Assuming these emissions do not pose unacceptable risks they will be incorporated into the DWP. The FRA will evaluate the risks posed by the projected emissions and for the HLW and LAW vitrification systems will evaluate emissions from environmental performance demonstration tests. Assuming these emissions do not pose unacceptable risks they will be incorporated into the WTP Permit replacing the PRA data. If either the PRA or the FRA exceed the thresholds listed in Table 9-1 additional site-specific data will be evaluated including evaluation of anticipated feed composition and projecting operating conditions.

## 9.3 Risk-Based Emissions Limits

Risk-based emission limits will be developed if risk and hazard thresholds are exceeded and if modification of overly conservative assumptions do not resolve any exceedances. These emission limits will be established following the PRA and the FRA. Risk-based emission limits will be provided for plausible exposure scenarios. Risk-based emission limits will be calculated for any COPC that exceeds risk thresholds in the PRA and FRA. If no individual COPC risks or hazards exceed these thresholds but the total risk or hazard exceeds thresholds, risk-based emission limits will be calculated for the COPCs having the largest contribution to this total risk or hazard. Additional site-specific information, and the results of the environmental performance demonstration test, will be available for the FRA and considered in development of risk-based emissions limits. Risk thresholds that are exceeded will be addressed to the satisfaction of Ecology and EPA and submitted for public comment prior to approval of the PRA and FRA.

### 9.3.1 Human Health Risk-Based Emission Limits

Risk-based emission limits will be calculated for COPCs with human health risks or hazards to plausible receptors greater than 1E-05 (ILCR) or 0.25 (HQ and HI). If no individual COPC risks or hazards exceed these thresholds but the total risk or hazard exceeds thresholds, risk-based emission limits will be calculated for the COPCs having the largest contribution to this total risk or hazard. Using the same exposure scenarios, pathways, toxicity values, and equations used to calculate plausible risk estimates, acceptable COPC concentrations in various media will be determined such that the corresponding total risk or hazard (across all media) is below the threshold values of 1E-05 (ILCR) and 0.25 (HQ and HI) for each plausible receptor. For acute exposure, the HQ threshold is 1.0. Air dispersion modeling results will then be used to convert these media concentrations to risk-based emission limits.

### 9.3.2 Ecological Risk-Based Emission Limits

Risk-based emission limits will be calculated for COPCs with ESQs greater than 0.25, or for driver chemicals if the total ESQ is greater than 0.25. Risk-based emission limits will be calculated for a given ecological receptor using the same exposure and food-web assumptions, toxicity values, and equations

used to calculate plausible ecological risk estimates. The risk equations will be used to back-calculate acceptable COPC concentrations in various media starting with an ESQ of 0.25. Air dispersion modeling results will then be used to convert these media concentrations to risk-based emissions limits.

## 9.4 Summary

In summary, the PRA will be submitted for review by regulatory agencies, American Indian tribes, and the public. If the PRA shows risks and hazards below the thresholds, the FRA will be performed and submitted following the environmental performance demonstration tests. If hazards or risks predicted in the PRA are above the thresholds, regulatory authorities will be consulted and the next course of action will be decided. Examples of potential actions are re-evaluating exposure parameters to determine whether the risk assessment was overly conservative and revising the operating plans to reduce emissions. If thresholds are still exceeded in the PRA, then risk-based criteria will be developed. All of these steps will help ensure that WTP operations will be conducted in a manner safe to human and ecological receptors on and near the Hanford site.

**Table 9-1 Overview of Risk Thresholds for COPCs and ROPCs in the PRA for the WTP**

| Receptor | Chronic Exposures | | | Acute Exposures | |
|---|---|---|---|---|---|
| | Carcinogens | Noncarcinogens | | | |
| | Incremental Lifetime Cancer Risk (ILCR) | Hazard Quotient or Ecological Screening Quotient (HQ or ESQ) | Hazard Index or Total Ecological Screening Quotient (HI or ESQ) | Acute Hazard Quotient (AHQ) | Acute Hazard Index (AHI) |
| **Chemicals of Potential Concern** | | | | | |
| Human | 1E-05 or 1 in 100,000 | 0.25 | 0.25 | 1.0 | 1.0 |
| Plants and Animals | NA | 0.25 | 0.25 | NA | NA |
| **Radionuclides of Potential Concern** | | | | | |
| Human | 1E-05 or 1 in 100,000 | NA | NA | 1.0 | 1.0 |
| Plants and Animals | NA | 0.25 | 0.25 | NA | NA |

NA = Not applicable.

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# 10 Uncertainty Assessment

Uncertainty or technical doubt is introduced into the human health and ecological risk assessments at every step of the process. As noted by EPA (2005), uncertainty occurs because risk assessment is a complex process, requiring integration of source information, fate and transport in various environments, exposure assessment, and effects assessment. Uncertainty is inherent in the risk assessment process even when the most accurate, up-to-date data and the most sophisticated models are used. Four types of uncertainty are addressed here:

- General (that is, non-effects) parameter uncertainty and variability
- Effects parameter uncertainty and variability
- Model uncertainty
- Decision-rule uncertainty

General parameter uncertainty occurs when variables used in equations cannot be measured precisely or accurately or have not been measured (such as lack of data). Other parameters are measurable and are represented by single fixed values, but actually have variability (such as body weight).

Effects parameter uncertainty and variability are associated with toxicity values (cancer slope factors [*CSF*s] and unit risk factors [*URF*s], reference doses [RfDs] and reference concentrations [*RfC*s] for human receptors, and toxicity reference values [*TRV*s] and benchmarks for ecological receptors), ecological measurement endpoints, and ecological assessment endpoints. Uncertainty occurs as a result of deficiencies in experimental design, extrapolation from experimental conditions to environmental conditions, or complete lack of effects information. Variability occurs as a result of variations in receptor sensitivity due to age, genetics, pre-existing conditions, presence of predators, or other environmental stressors.

Model uncertainty is associated with all models used in all phases of the risk assessments, including air dispersion and other environmental models, animal models used as surrogates for testing human health effects, and dose response models used in extrapolation of laboratory data to human health or ecological effects. All models are simplifications of reality, and therefore exclude some variables to reduce complexity and/or to compensate for missing data. The models identified in this environmental RAWP were selected on the basis of scientific policy because they provide the information needed to conduct the risk assessments and are considered by Ecology and EPA to be state-of-the-science models.

Decision-rule uncertainty arises out of the need to balance different social concerns when determining an acceptable level of risk. Decision-rule uncertainty is associated with the choice of models used, the selection of constituents to be included in the analysis, the default parameter values used, the dependence on single-point estimates of toxicity (human *RfD*s/*RfC*s and *CSF*s/*URF*s and ecological *TRV*s), and the selection of risk and hazard thresholds for evaluating the results of the SLRA.

An overview of the potential sources of uncertainty in the SLRA is provided in Section 10.1. A discussion of how uncertainty will be addressed in the PRA is provided in Section 10.2.

## 10.1 Sources of Uncertainty in the SLRA

A brief summary of the sources of uncertainty in each step of the risk assessment is provided below. Additional discussion is provided in Sections 4.2, 5.5, 6.8, 7.5, and 8.6 of this RAWP. One or more of the four types of uncertainty described above impact each of these steps.

### 10.1.1 Identification of Constituents of Potential Concern

The identification of COPCs and ROPCs discussed in Section 4 is uncertain because these constituents are identified before operation of the WTP and must rely on assumptions regarding what may be in the waste feed and what may be produced as products of incomplete combustion (PICs). Test data collected for the FRA during the environmental performance demonstration will reduce, but not eliminate, this uncertainty because this test data will include uncertainty due to tentatively identified compounds (TICs), detection limits, and variations in actual waste feed.

### 10.1.2 Estimation of Emissions

The primary sources of uncertainty in the emissions estimate are as follows:

- Characterization data that describes the waste feed streams to the WTP PT Facility
- Decontamination efficiency of the air pollution control equipment
- Creation of PICs by the WTP
- Potential impact of upset conditions and abated fugitive emissions on the overall emission rates

### 10.1.3 Environmental Modeling

Uncertainties are associated with each aspect of the environmental modeling (air-dispersion modeling, soil accumulation modeling, surface water accumulation modeling, sediment accumulation modeling, and plant accumulation modeling). Uncertainties are associated with both the models themselves, because models are simplifications of reality, and with the parameters and data used in the models.

### 10.1.4 Human Health Risk Assessment

Uncertainties associated with the COPC and ROPC selection, emission rates, and environmental modeling all contribute to the uncertainty in the HHRA. Sources of uncertainty unique to the HHRA are associated with each step of the HHRA: data evaluation, exposure assessment, toxicity assessment, and risk characterization.

Sources of uncertainty in the data evaluation are described above in Sections 10.1.1 and 10.1.2. Sources of uncertainty in the exposure assessment include contaminant concentrations in exposure media, exposure parameter uncertainty and variability in land-use assumptions, and selection of representative receptor populations and exposure parameter values. Sources of uncertainty in the toxicity assessment include effects uncertainty and variability in toxicity values (*RfD*s/*RfC*s and *CSF*s/*URF*s) and toxicity value data gaps, and surrogates to fill some toxicity data gaps. The risk characterization combines the results of the exposure assessment and toxicity assessment. Therefore, all of the uncertainty in these two steps, as well as the steps prior to the exposure assessment (such as environmental modeling), contributes to the uncertainty in the risk characterization. Additional uncertainty in the risk characterization step surrounds the practice of summing risks and hazard results across all chemicals and exposure pathways,

regardless of the mode of action. Also, uncertainty is associated with the eventual human health risk and hazard outcomes and their interpretation.

### 10.1.5 Ecological Risk Assessment

Uncertainties associated with the COPC and ROPC selection, emission rates, and environmental modeling also contribute to the uncertainty in the ERA. Sources of uncertainty unique to the ERA are associated with each of the four inter-related steps of the ERA: problem formulation, exposure assessment, effects assessment, and risk characterization.

Sources of uncertainty in the problem formulation include identification of representative receptor populations and exposure media. Sources of uncertainty in the exposure assessment include exposure parameter uncertainty and variability included in selection of representative exposure parameter values and contaminant concentrations in exposure media. Sources of uncertainty in the effects assessment include effects uncertainty and variability in toxicity values (TRVs and benchmark values) and toxicity value data gaps. The risk characterization combines the results of the exposure assessment and effects assessment. Therefore, all of the uncertainty in these two steps, as well as the steps prior to the exposure assessment (such as environmental modeling) contributes to the uncertainty in the ecological risk characterization. Additional uncertainty in the risk characterization step surrounds the practice of summing hazard results across all chemicals regardless of the mode of action. Also, uncertainty is associated with the eventual ecological risk outcomes and their interpretation.

## 10.2 Uncertainty Assessment in the PRA

The purpose of the uncertainty assessment is to identify and discuss uncertainty associated with the quantitative estimates of human health and ecological risk for the WTP. This discussion serves to place the risk estimates in proper perspective to allow fully informed risk management decisions.

The EPA (2005) notes that: “The science of risk assessment is evolving; where the science base is incomplete and uncertainties exist, science policy assumptions must be made.” Therefore, it is important for risk assessments of treatment facilities such as the WTP to identify uncertainties in the assessment. To meet this obligation, the PRA report will provide an uncertainty analysis that will include:

- Tables listing the general assumptions in each step of the assessments, the rationale for these assumptions, their potential effect on estimates of risk, and the direction and approximate magnitude of the effect
- An analysis of the key assumptions impacting the COPCs and ROPCs, receptors, and exposure pathways that are risk drivers (such as result in risks above or slightly below the established threshold values)
- An evaluation of several other specific sources of uncertainty associated with gaps in our scientific knowledge, or scientific debates over the most appropriate approaches

Each of these items is addressed in more detail below.

### 10.2.1 Uncertainty Tables

Tables listing the general assumptions in each step of the assessments, the rationale for these assumptions and their potential effect on estimates of risk (overestimation or underestimation), and the approximate

magnitude of the effect (minor or major) will be included in the uncertainty assessment. These tables will focus on categories of assumptions rather than specific assumptions. For example, residential exposure parameters may be included, whereas details of each exposure parameter (such as soil ingestion rate or body weight) will not be included. Examples of the planned table formats and contents are provided as Table 10-1 through Table 10-5.

### 10.2.2 Uncertainty Analysis of Key Assumptions

In addition to the tables described above, a more detailed analysis of the key assumptions impacting the COPCs and ROPCs, human and ecological receptors, and exposure pathways that are risk drivers (such as a result in risks above or slightly below the established threshold values) will be included in the PRA. Examples of possible scenarios resulting in an analysis of key assumptions for the HHRA and ERA are provided below.

- If the total estimated incremental lifetime cancer risk to a resident at the Hanford offsite maximum is 9E-06 (that is, 9 excess cancers in 1,000,000 people) and slightly below the risk threshold of 1E-05 (that is, 10 excess cancers in 1,000,000 people), the uncertainty analysis will focus on the specific constituents and exposure pathways that result in this risk and any assumptions that could result in the actual risk being higher or lower. For example, if the risk due to ingestion of one COPC in homegrown produce is 8E-06 and the total risk from all other COPCs and pathways is 1E-06, the uncertainty analysis would focus on the models and assumptions used to estimate the concentration of that chemical in plants, the residential produce ingestion assumptions, and the toxicity data for the one chemical of interest. This analysis will serve to evaluate whether this risk estimate is likely to be an overestimate or underestimate of reality, and if so, to what extent.
- If the total ESQ to a Great Basin pocket mouse at the onsite ground maximum is 0.35 (slightly above the hazard threshold of 0.25), the uncertainty analysis will focus on the specific chemicals and specific exposure pathways that result in this hazard and any assumptions that could result in the actual hazard being higher or lower. For example, if the hazard due to ingestion of one COPC in soil invertebrates is 0.20 and the hazard due to ingestion of another COPC in soil invertebrates is 0.10, the uncertainty analysis will focus on whether or not it is appropriate to add the ESQs for these two chemicals, the models and assumptions used to estimate the concentrations of these two chemicals in soil invertebrates, the assumption that the mouse has an exclusive diet of soil invertebrates, and the toxicity data for these two chemicals. This analysis will serve to evaluate whether this hazard estimate is likely to be an over- or underestimate of reality, and if so, to what extent.

These are just two examples of the type of specific uncertainty assessment that may be triggered by the findings of the PRA.

### 10.2.3 Alternate Exposure Scenarios

#### 10.2.3.1 Future Exposure at the Onsite Ground Maximum Location

The *Record of Decision (ROD): Hanford Comprehensive Land-Use Plan Environmental Impact Statement* (64 FR 61615) indicates that DOE has chosen to implement the DOE Preferred Alternative land-use map which designates the Central Plateau (including the ground-maximum location) geographic area Industrial-Exclusive. This land-use designation is consistent with DOE's current management and operation and allows DOE to continue waste management operations in this area of the site and to expand existing facilities or develop new facilities to meet future mission needs. Although this land-use

designation precludes the potential exposure to contaminants from residential occupation at the ground maximum, DOE acknowledges that the *Comprehensive Land-Use Plan* (CLUP) is an ongoing process (DOE 2008). If the land-use designation is modified and potential residential occupation at alternate locations (such as the onsite ground maximum or other parts of the site interior) becomes a possibility, the uncertainty assessment in the PRA will include estimated risks to selected receptors as a result of living at the alternate locations in the future timeframe. This assessment will be performed by incorporating future deposition values into the respective exposure scenarios for the appropriate pathways (incidental soil inhalation/ingestion, homegrown produce and livestock) at the location of interest. Note that future exposure at the onsite ground maximum is considered a worst-case scenario because future development at this location is unlikely due to the presence of other industrial and hazardous waste operations in the 200 Areas.

#### 10.2.3.2 Alternate American Indian Exposure Scenarios

Currently, the only American Indian scenario endorsed by the DOE is the American Indian hunter-gatherer exposure scenario developed for the *Final Tank Closure and Waste Management Environmental Impact Statement for the Hanford Site* (TC&WM EIS, DOE 2012). This exposure scenario has been adapted for use in this SLRA and associated risk assessment results will be reported along with other receptors of interest in the PRA. However, the American Indian scenario described in the TC&MW EIS does not necessarily have the full endorsement of regional American Indian tribes whose treaty rights grant them access to the Hanford site. Accordingly, two alternative American Indian scenarios have been developed for this risk assessment and will be fully evaluated and reported as part of the uncertainty assessment of the PRA. The lifestyle and exposure parameters of the first alternate resident subsistence American Indian are primarily based on data from *Exposure Scenario for CTUIR Traditional Subsistence Lifeways* (Harris and Harper 2004), *Application of the CTUIR Traditional Lifeways Exposure Scenario in Hanford Risk Assessments* (Harris 2008), and "A Native American Exposure Scenario" (Harris and Harper 1997). The lifestyle and exposure parameters of the second alternate resident subsistence American Indian are primarily based on data from *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment* (RUDOLFI Inc. 2007). Where these guidance documents omitted necessary information, exposure parameters were established using information published in the EFH (EPA 1997a). Children's exposure parameters were developed by proportioning the child caloric intake reported in the CSEFH (EPA 2008) according to the various proportions of meat, vegetable, roots, etc., in the diet of the adult tribal member as reported in the guidance documents cited above. The specific exposure parameters associated with these two alternate resident subsistent American Indian exposure scenarios are presented in Section 7.1.3.9. Incorporation of these alternate scenarios into the uncertainty assessment of the PRA will provide data of interest for regional tribes without contradicting the American Indian scenario established in the TC&MW EIS.

### 10.2.4 Other Specific Uncertainty Issues

Several sources of uncertainty associated with gaps in our scientific knowledge or scientific debates over the most appropriate approaches to use are identified throughout this RAWP. These issues, as detailed below, will be discussed in the PRA uncertainty assessment:

- **Offsite Exposure Point Concentrations** – In order to help quantify the degree of conservatism associated with using the 90th percentile of air concentration and deposition values from the offsite grid, the location and species values associated with the point of highest annual total air concentration and deposition will be determined in the uncertainty assessment. Total air concentration ($Conc_{Total}$) and deposition ($Dep_{Total}$) values for each year and offsite exposure grid node will be computed according to:

$$Conc_{Total} = Cyv_{PT} + Cyv_{LAW} + Cyv_{HLW} + Cyp_{PT_1} + Cyp_{LAW_1} + Cyp_{HLW_1} + Cyp_{PT_1} + Cyp_{LAW_1} + Cyp_{HLW_1}$$

$$\begin{aligned} Dep_{Total} = {} & Dydv_{PT} + Dydv_{LAW} + Dydv_{HLW} + Dywv_{PT} + Dywv_{LAW} + Dywv_{HLW} + \\ & Dydp_{PT_1} + Dydp_{LAW_1} + Dydp_{HLW_1} + Dydp_{PT_{2.5}} + Dydp_{LAW_{2.5}} + Dydp_{HLW_{2.5}} + \\ & Dywp_{PT_1} + Dywp_{LAW_1} + Dywp_{HLW_1} + Dywp_{PT_{2.5}} + Dywp_{LAW_{2.5}} + Dywp_{HLW_{2.5}} \end{aligned}$$

Where

$Conc_{Total}$ = the total air concentration (in µg·s/g·m$^3$)

$Dep_{Total}$ = the total deposition (in s/m$^2$·yr)

other variables as defined in Section 6.1.4.3

A comparison of all years and grid node values will be used to determine the maximum values of $Conc_{Total}$ and $Dep_{Total}$ along with their corresponding grid node coordinates (easting and northing, as represented here by the notation $X_{CONC}$, $Y_{CONC}$ and $X_{DEP}$, $Y_{DEP}$, corresponding to $Conc_{Total}$ and $Dep_{Total}$, respectively). High values for $Cyv$, $Cyp_1$, $Cyp_{2.5}$, $Dydv$, $Dydp_1$, $Dydp_{2.5}$, $Dywv$, $Dywp_1$, and $Dywp_{2.5}$ will derived from the offsite grid points associated with $X_{CONC}$, $Y_{CONC}$ and $X_{DEP}$, $Y_{DEP}$ and corresponding EPCs will be computed for comparison to those EPCs computed using 90$^{th}$ percentiles. Implications of this comparison will be presented in the uncertainty assessment of the PRA report.

- **Nursing infant assessment** – Potential risks to nursing infants from dioxin-like compounds will be evaluated by comparing the estimated infant dose of dioxins, furans, and coplanar PCBs from the WTP to the background infant dose of these chemicals throughout the United States. The background infant dose referenced in this RAWP may overestimate current exposures because dioxin exposures in the United States have been decreasing for many years. The source of this value and potential range of background infant doses will be discussed further in the uncertainty assessment of the PRA report. There is currently no consensus regarding the most appropriate single approach to quantitatively evaluate potential risks associated with exposure to dioxin-like compounds by nursing infants. Alternative approaches to the two preferred methods to be used in the PRA (that is, comparison to background and lifetime risk) include calculating infant risks using (1) the estimated infant ADD calculated with a exposure duration equal to the period of breast feeding and an equivalent averaging time, and (2) the estimated infant LADD calculated with a exposure duration equal to the period of breast feeding and a 70-year averaging time. These alternative methods will be presented in the uncertainty assessment of the PRA report.
- **Partial exclusion of dermal pathway from the HHRA** – Dermal exposure pathways (to soil, surface water, or air) will not be included in the PRA, with the exception of the sweat lodge exposure pathway, because dermal exposure pathways have been identified as insignificant contributors to risk in numerous risk assessments prepared or reviewed by EPA for airborne emissions from thermal treatment facilities. If initial PRA results indicate that the soil or surface water ingestion or inhalation pathways result in risks that are borderline (that is, close to the risk or hazard threshold) for any plausible receptor, then dermal exposure to that medium may be included in the PRA. A discussion of the potential impact associated with exclusion of this minor pathway from the quantitative risk assessment will be included in the uncertainty assessment of the PRA.
- **Evaluation of PAHs** – Potential human cancer risks associated with 7 polycyclic aromatic hydrocarbons (PAHs) considered to be carcinogenic by EPA (1993) will be evaluated using a RPF approach. The RPFs for an additional 15 PAHs are available from the California (Cal EPA 1999). If the total estimated risk from PAHs is near 1E-05, these additional 15 PAHs will be considered in the uncertainty analysis.
- **Dioxin slope factor** – Potential human cancer risks associated with dioxins and coplanar PCBs will be evaluated using the cancer *CSF* of 1.0E+06 (mg/kg-day)$^{-1}$ proposed in the *Exposure and Human*

*Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds* (EPA 2003), and as suggested by Ecology and EPA Region 10 (CCN 063809). While this proposed *CSF* has not yet been approved by EPA, it is more conservative than the current *CSF* published in the HEAST (EPA 1997b). A discussion of comparative risk results will appear in the uncertainty section of the PRA.

- **Toxicity data gaps** – The COPCs without toxicity values (*RfD*, *RfC*, *CSF*, *URF*, *TRV*, ecological benchmarks) cannot be included in the quantitative risk assessments. The potential impact of these COPCs on the risk results will be discussed in the uncertainty assessment.
- **Route-To-Route Extrapolations** – Uncertainties are associated with the estimation of dermal toxicity values from oral values. The *URF* and $RfC_{inh}$ derived from the $CSF_{inh}$ and $RfD_{inh}$ using the conversion in WAC 173-340-708(7)b will have uncertainty if the respiratory deposition and absorption characteristics of the gases and inhaled particles is unknown. Constituents for which a route-to-route extrapolated toxicity was used will be identified and a qualitative discussion of the impact will be included.
- **Radiation benchmarks** – The whole-organism radiation benchmarks for ecological receptors identified in this RAWP have uncertainty associated with them, because they do not take into account effects on sensitive tissues, critical organ effects, relative biological effectiveness, and microdosimetry issues. These issues are currently being investigated by the scientific community and will be mentioned in the uncertainty assessment.
- **Microdosimetry of radionuclides** – Possible synergistic effects of multiple radionuclides and microdosimetry to root hairs, eggs, embryos, and so forth for ecological receptors are currently being investigated and developed by researchers. The current status of this research will be mentioned in the uncertainty discussion in the PRA.
- **Exclusion of external alpha radiation** – The possible effects of external alpha radiation on ecological receptors will be included in the uncertainty assessment of the PRA because external alpha radiation should add only insignificantly to the whole-body dose for organisms (Blaylock and others 1993). The potential impact of omitting alpha radiation will be identified in the uncertainty assessment.
- **Summations of risks** – The PRA will include summations of the total COPC and ROPC risks and hazards as listed below:
  - Total cancer risk to human receptors from all COPCs
  - Total cancer risk to human receptors from all ROPCs
  - Total HI for human receptors from all COPCs
  - Total ESQ for ecological receptors from all COPCs
  - Total ESQ for ecological receptors from all ROPCs

These total risk and hazard calculations will be based on the assumption that the effects of all COPCs or ROPCs to a given receptor are summed. If risk or hazard thresholds are exceeded, a segregation of the constituents by toxicological mode of action and endpoint will be considered. If segregation by toxicological mode of action or endpoint is used, chemical groupings by endpoint will be assigned with approval by Ecology and EPA.

These issues, associated with gaps in our scientific knowledge or with scientific debates over the most appropriate approaches, and any other issues identified while conducting the PRA, will be included in the PRA uncertainty assessment.

## 10.3 Summary of Uncertainty Assessment

Uncertainty is inherent in every step of the risk assessment process. An uncertainty assessment will be included in the PRA to (1) identify sources of uncertainty associated with the quantitative estimates of human health and ecological risk from the WTP, (2) estimate the potential magnitude of key uncertainties that could influence the results of the PRA, and (3) show other analyses associated with data gaps and scientific discussion. The uncertainty assessment will be used to place the risk estimates in proper perspective to allow fully informed risk management decisions.

## 10.4 References

### 10.4.1 Project Documents

CCN 063809, *Ecology/EPA to WTP Regarding Dioxin Slope Factor and Acute Hazard Threshold*, Personal communication between SAIC, US Environmental Protection Agency, Region 10, and Washington Department of Ecology, at a meeting held on 23 and 24 April 2003 in Seattle, Washington.

### 10.4.2 Codes and Standards

None.

### 10.4.3 Other Documents

Blaylock BG, Frank ML, and O'Neal BR. 1993. *Methodology for Estimating Radiation Dose Rates to Freshwater Biota Exposed to Radionuclides in the Environment*, ES/ER/TM-78. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

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Harris SG. 2008. *Application of the CTUIR Traditional Lifeways Exposure Scenario in Hanford Risk Assessments*, Department of Science & Engineering, Confederated Tribes of the Umatilla Indian Reservation, P.O. Box 638, Pendleton, Oregon 97801.

RIDOLFI Inc. 2007. *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment*, Yakama Nation ERWM Program, September 2007.

**Table 10-1 Example of Sources of Uncertainty in Identification of COPCs and ROPCs**

| Examples of Sources of Uncertainty | Description | Potential Direction and Magnitude of Risk[b] | | |
|---|---|---|---|---|
| | | Over-estimation | Not defined | Under-estimation |
| Tank characterization data | Constituents identified in tank waste are included as COPCs and ROPCs. | | | |
| Nondetected constituents | Constituents not detected in tank waste but which may have been used at Hanford are included as COPCs and ROPCs. | | | |
| PICs identified in bench-scale testing | Constituents identified in bench-scale testing are included as COPCs and ROPCs. | | | |
| PICs identified at hazardous waste combustion facilities | Constituents identified in emissions from hazardous waste combustion facilities are included as COPCs and ROPCs. | | | |

[a] This is an example of the information to be included in the uncertainty table in the PRA report and is not intended to be inclusive of all sources of uncertainty.

[b] These columns will indicate whether the assumption used to compensate for the uncertainty is likely to overestimate or underestimate the actual risk, or whether the direction cannot be identified. The potential magnitude of this impact will be identified as minor (+, o, -) or major (++, oo, --).

**Table 10-2 Example of Sources of Uncertainty in Emissions Estimate**

| Examples of Sources of Uncertainty | Description | Potential Direction and Magnitude of Risk[b] | | |
|---|---|---|---|---|
| | | Over-estimation | Not defined | Under-estimation |
| Starting concentration of constituents in tank waste | Identified organics are multiplied by a scaling factor to adjust for unidentified organics. | | | |
| | Tanks assumed to have highest organic concentration were used for analysis. | | | |
| Throughput of treatment system | System is assumed to run at full capacity for 40 years. | | | |
| Efficacy of pollution control equipment | Removal is based on engineering design and assumptions rather than measured values | | | |
| Assignment of phase | Each COPC and ROPC is assumed to be present as either vapor, particulate, or particulate-bound. Some constituents may be present as a combination of phases. | | | |
| Default upset factors for vapor-phase emissions | Default upset factors are based on recorded operating conditions at hazardous waste combustion units. | | | |

[a] This is an example of the information to be included in the uncertainty table in the PRA report and is not intended to be inclusive of all sources of uncertainty.

[b] These columns will indicate whether the assumption used to compensate for the uncertainty is likely to overestimate or underestimate the actual risk, or whether the direction cannot be identified. The potential magnitude of this impact will be identified as minor (+, o, -) or major (++, oo, --).

**Table 10-3 Example of Sources of Uncertainty in Environmental Modeling**

| Examples of Sources of Uncertainty | Description | Potential Direction and Magnitude of Risk[b] | | |
|---|---|---|---|---|
| | | Over-estimation | Not defined | Under-estimation |
| **Air Dispersion Modeling** | | | | |
| COPC and ROPC lists | All COPCs and ROPCs are modeled; therefore, uncertainty in this list will be carried into the air dispersion modeling. | | | |
| Emission Rates of COPCs and ROPCs | Estimated emission rates are the starting point for predicting airborne dispersion; therefore, uncertainty in these estimates will be carried into the air dispersion modeling. | | | |
| Use of CALPUFF air dispersion model | Simulation of the atmospheric dispersion of emissions is limited by data limitations and simplifications inherent in the model. | | | |
| Surface meteorological data for 01 January 2002 through 31 December 2006 | This is considered representative of long-term conditions. | | | |
| Particle size distribution | Particle size influences deposition. Particle sizes of 1 µm and 2.5 µm are assumed. | | | |
| Land use and terrain data | Data represents land uses at a point in time, with terrain resolution that varies from 70 m to 90 m, with an absolute accuracy of 130 m in the horizontal and 30 m in the vertical. | | | |
| **Soil Accumulation Modeling** | | | | |
| COPC and ROPC list | All COPCs and ROPCs are modeled; therefore, uncertainty in this list will be carried into the soil accumulation modeling. | | | |
| Emission rates of COPCs and ROPCs | Estimated emission rates are the starting point for predicting airborne dispersion and air dispersion is the starting point for predicting soil concentrations; therefore, uncertainty in these estimates will be carried into the soil accumulation modeling. | | | |
| Constituent deposition rates | Air dispersion is the starting point for predicting soil concentrations; therefore, uncertainty in these estimates will be carried into the soil accumulation modeling. | | | |

**Table 10-3 Example of Sources of Uncertainty in Environmental Modeling**

| Examples of Sources of Uncertainty | Description | Potential Direction and Magnitude of Risk[b] | | |
|---|---|---|---|---|
| | | Over-estimation | Not defined | Under-estimation |
| Descriptive soil parameters | Parameters such as mixing depth, bulk density, and volumetric water content, which are assigned a single value, may vary widely over a relatively small area. | | | |
| Soil loss mechanisms – degradation | COPCs in soil are subject to loss due to biotic and abiotic degradation; however, transformation and subsequent increase of secondary COPCs are not considered in the assessment. | | | |
| | Degradation rates, which are assigned a single value, generally from laboratory testing, may vary widely under environmental conditions. | | | |
| **Surface Water and Sediment Accumulation Modeling** | | | | |
| COPC and ROPC list | All COPCs and ROPCs are modeled; therefore, uncertainty in this list will be carried into the surface water and sediment modeling. | | | |
| Emission rates of COPCs and ROPCs | Estimated emission rates are the starting point for predicting airborne dispersion and air dispersion is the starting point for predicting surface water concentrations; therefore, uncertainty in these estimates will be carried into the surface water and sediment modeling. | | | |
| Constituent deposition rates | Air dispersion is the starting point for predicting surface water concentrations; therefore, uncertainty in these estimates will be carried into the surface water and sediment modeling. | | | |
| Surface water and sediment model | Equations used to model the fate of COPCs and ROPCs deposited into the water body greatly simplify the mechanisms occurring within such a dynamic system. | | | |
| Deposition area | The maximum deposition of COPCs and ROPCs is assumed over the entire depositional area of the water body. | | | |
| Descriptive surface water and sediment parameters | Parameters such as depth of water column and depth of upper benthic sediment layer, which are assigned a single value, may vary widely. | | | |

**Table 10-3 Example of Sources of Uncertainty in Environmental Modeling**

| Examples of Sources of Uncertainty | Description | Potential Direction and Magnitude of Risk[b] | | |
|---|---|---|---|---|
| | | Over-estimation | Not defined | Under-estimation |
| **Plant Accumulation Modeling** | | | | |
| COPC and ROPC list | All COPCs and ROPCs are modeled; therefore, uncertainty in this list will be carried into the plant modeling. | | | |
| Emission rates of COPCs and ROPCs | Estimated emission rates are the starting point for predicting environmental concentrations; therefore, uncertainty in these estimates will be carried into the plant modeling. | | | |
| Air dispersion modeling | Airborne concentrations are the starting point for predicting direct uptake from air; therefore, uncertainty in these estimates will be carried into the plant modeling. | | | |
| Constituent deposition rates | Deposition is the starting point for predicting plant concentrations from direct deposition; therefore, uncertainty in these estimates will be carried into the plant modeling. | | | |
| Soil accumulation modeling | Soil concentration is the starting point for predicting uptake into plants; therefore, uncertainty in these estimates will be carried into the plant modeling. | | | |
| Plant uptake factors | Air-to-plant and soil-to-plant uptake factors, which are assigned a single value, generally from laboratory testing of a limited number of chemicals, may vary widely depending on constituent, plant species, and environmental conditions. | | | |
| Descriptive plant parameters | Parameters such as length of growing season and yield, which are assigned a single value, may vary widely among plant species and agricultural practices. | | | |

[a] This is an example of the information to be included in the uncertainty table in the PRA report and is not intended to be inclusive of all sources of uncertainty.

[b] These columns will indicate whether the assumption used to compensate for the uncertainty is likely to overestimate or underestimate the actual risk, or whether the direction cannot be identified. The potential magnitude of this impact will be identified as minor (+, o, -) or major (++, oo, --).

**Table 10-4 Example of Sources of Uncertainty in Human Health Risk Assessment** [a]

| Examples of Sources of Uncertainty | Description | Potential Direction and Magnitude of Risk[b] | | |
|---|---|---|---|---|
| | | Over-estimation | Not defined | Under-estimation |
| **Exposure Assessment** | | | | |
| COPC and ROPC list | All COPCs and ROPCs are modeled; therefore, uncertainty in this list will be carried into the HHRA. | | | |
| Emission Rates of COPCs and ROPCs | Estimated emission rates are the starting point for predicting environmental concentrations; therefore, uncertainty in these estimates will be carried into the HHRA. | | | |
| Air dispersion modeling | Airborne concentrations are the starting point for predicting inhalation exposures; therefore, uncertainty in these estimates will be carried into the HHRA | | | |
| Soil accumulation modeling | Soil concentration is the starting point for predicting soil ingestion exposures and uptake into foodstuffs; therefore, uncertainty in these estimates will be carried into the HHRA. | | | |
| Surface water accumulation modeling | Surface water concentration is the starting point for predicting drinking water, fish ingestion, and sweat lodge exposures; therefore, uncertainty in these estimates will be carried into the HHRA. | | | |
| Plant accumulation modeling | Plant concentration is the starting point for predicting produce ingestion exposures and concentrations in animal products; therefore, uncertainty in these estimates will be carried into the HHRA. | | | |
| Exposure parameters | Exposure parameters are a combination of average (such as body weight) and upper-bound (such as soil ingestion) point estimates of parameters that vary widely among individuals. | | | |
| **Toxicity Assessment** | | | | |
| Cancer slope factors (*CSF*s) for COPCs | *CSF*s are a plausible upper-bound estimate of the probability of a cancer, per unit intake of a chemical, over a lifetime. Most chemical *CSF*s are based on animal data. | | | |
| Cancer slope factors for ROPCs | *CSF*s are central estimates of the age-averaged, lifetime radiation cancer incidence risk and are based on human data. | | | |

**Table 10-4 Example of Sources of Uncertainty in Human Health Risk Assessment** [a]

| Examples of Sources of Uncertainty | Description | Potential Direction and Magnitude of Risk[b] | | |
|---|---|---|---|---|
| | | Over-estimation | Not defined | Under-estimation |
| **Risk Characterization** | | | | |
| Exposure assessment | All uncertainties in the exposure assessment are carried into the risk characterization. | | | |
| Toxicity assessment | All uncertainties in the toxicity assessment are carried into the risk characterization. | | | |
| Additivity of COPC cancer risk | The assumption of additivity of COPC cancer risk assumes intakes of individual chemicals are small, and there is no interaction among chemicals. | | | |
| Additivity of ROPC cancer risk | The assumption of additivity of ROPC cancer risk is much less uncertain than for COPCs because the mode of action is the same for all radionuclides. | | | |
| Additivity of COPC hazard quotients | The assumption of additivity is likely to overestimate risk since many chemicals act on different target organs. | | | |

[a] This is an example of the information to be included in the uncertainty table in the PRA report and is not intended to be inclusive of all sources of uncertainty.

[b] These columns will indicate whether the assumption used to compensate for the uncertainty is likely to overestimate or underestimate the actual risk, or whether the direction cannot be identified. The potential magnitude of this impact will be identified as minor (+, o, -) or major (++, oo, --).

[c] In this context, residential receptors include resident (adult and child), resident subsistence farmer (adult and child), resident fisher (adult and child), Native American subsistence resident (adult and child), and the residential portion of the Hanford Site industrial worker exposure.

**Table 10-5 Example of Sources of Uncertainty in Ecological Risk Assessment** [a]

| Examples of Sources of Uncertainty | Description | Potential Direction and Magnitude of Risk[b] | | |
|---|---|---|---|---|
| | | Over-estimation | Not defined | Under-estimation |
| **Problem Formulation** | | | | |
| Identification of ecological receptors | Receptors are identified to represent various feeding guilds and trophic levels. | | | |
| Choice of assessment endpoints | Endpoints are chosen to represent key species in the Hanford Site ecosystem. | | | |
| Choice of measurement endpoints | Endpoints are chosen to represent significant deleterious effects to ecological receptors. | | | |
| **Exposure Assessment** | | | | |
| COPC and ROPC list | All COPCs and ROPCs are modeled; therefore, uncertainty in this list will be carried into the ERA. | | | |
| Emission Rates of COPCs and ROPCs | Estimated emission rates are the starting point for predicting environmental concentrations; therefore, uncertainty in these estimates will be carried into the ERA. | | | |
| CALPUFF air dispersion modeling | Airborne concentrations are used to predict environmental concentrations; therefore, uncertainty in these estimates will be carried into the ERA. | | | |
| Soil accumulation modeling | Soil concentration is the starting point for predicting soil ingestion exposures and uptake into food; therefore, uncertainty in these estimates will be carried into the ERA. | | | |
| Surface water and sediment accumulation modeling | Surface water and sediment concentrations are the starting point for predicting exposure to aquatic biota; therefore, uncertainty in these estimates will be carried into the ERA. | | | |
| Plant accumulation modeling | Plant concentration is the starting point for predicting plant ingestion exposures and concentrations in higher trophic levels; therefore, uncertainty in these estimates will be carried into the ERA. | | | |
| Food chain multiplier (FCM) approach for aquatic receptors | The challenge of extrapolating from one aquatic species to another will be identified. | | | |
| FCM approach for terrestrial receptors | The challenge of extrapolating from aquatic species (which make up the database for FCMs) to terrestrial food chains will be identified. | | | |

**Table 10-5 Example of Sources of Uncertainty in Ecological Risk Assessment** [a]

| Examples of Sources of Uncertainty | Description | Potential Direction and Magnitude of Risk[b] | | |
|---|---|---|---|---|
| | | Over-estimation | Not defined | Under-estimation |
| Exclusive diets | Exclusive diets mathematically make the animal too dependant on one food source (whether plants or animals). This represents a large departure from realistic real diets for desert omnivores. | | | |
| Exposure parameters | Exposure parameters are a combination of average and upper-bound point estimates of parameters that vary widely among individuals. | | | |
| **Effects Assessment** | | | | |
| Toxicity reference values for terrestrial receptors | Toxicity thresholds are based on concentrations reported to have no, or little, effect on the test organism or are estimated conservatively from published toxicity data. | | | |
| Toxicity reference values for aquatic receptors | Toxicity thresholds are based on concentrations reported to have no, or little, effect on the test organism or are estimated conservatively from published toxicity data. | | | |
| **Risk Characterization** | | | | |
| Exposure assessment | All uncertainties in the exposure assessment are carried into the risk characterization. | | | |
| Effects assessment | All uncertainties in the effects assessment are carried into the risk characterization. | | | |
| Additivity of COPC hazard quotients | The assumption of additivity is likely to overestimate risk since many chemicals act on different target organs. | | | |

[a] This is an example of the information to be included in the uncertainty table in the PRA report and is not intended to be inclusive of all sources of uncertainty.

[b] These columns will indicate whether the assumption used to compensate for the uncertainty is likely to overestimate or underestimate the actual risk, or whether the direction cannot be identified. The potential magnitude of this impact will be identified as minor (+, o, -) or major (++, oo, --).

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## 11.10 Section 10

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# Appendix A

# Supporting Equations for Accumulation Modeling and Derivation of Selected Parameters

# Appendix A
# Supporting Equations for Accumulation Modeling and Derivation of Selected Parameters

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# A.1 Introduction

This appendix provides supplemental equations to the equations provided in Section 6 (Environmental Modeling) of this work plan. The equations and parameters in this appendix are from the US Environmenal Protection Agency's (EPA) *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (HHRAP) (EPA 2005). Equations that support the soil, surface water, and sediment accumulation modeling and data are provided in this appendix. Equations in Section 6 refer to the immediate supporting equations within this appendix. Parameters that are functions of other parameters are presented only in this appendix (e.g., the equation for the soil loss constant due to biotic and abiotic degradation, presented below, is referenced in the definition of parameters used to estimate the total soil loss constant, which is referenced in subsequent equations). Section 6 presents only the "high-level" equations; all supporting equations (including supporting equations for parameters that appear in other supporting equations) are presented in this appendix. A description of how the parameters shown in this appendix link to the equations in Section 6 is provided for each equation in this appendix.

Because many of the equations used in the soil modeling are functions of other equations, the intermediary calculations necessary to calculate the chemical of potential concern (COPC) or radionuclide of potential concern (ROPC) concentrations in soil should be performed in a logical order. The equations for these intermediary calculations can be found in this appendix; values for the contaminant-specific parameters are presented in Supplement 4. The order for these intermediary calculations is as follows:

1. Estimate individual COPC and ROPC soil-loss mechanisms. These include soil loss constant due to biotic and abiotic degradation (Eq. A-2-13 in HHRAP), soil loss constant due to radiological decay (Eq. A-2-13 in HHRAP), soil loss constant due to leaching (Eq. 5-5A in HHRAP), soil loss constant due to surface runoff (Eq. 5-4 in HHRAP), soil loss constant due to volatilization (Eq. 5-6 in HHRAP), and soil loss constant due to soil erosion (Eq. 5-3 in HHRAP). These soil loss mechanisms are estimated using methods provided in EPA (2005), along with Hanford-specific parameter values (a site-specific parameter value unique to the Hanford Site), site-specific parameter values (a parameter unique to a site and independent of the constituent being evaluated; the actual value may be a default value and not specific to the Hanford Site), and contaminant-specific parameter values (a parameter unique to a contaminant and independent of the site being evaluated) where appropriate (see Table 6-3 for Hanford-specific and site-specific parameter values, and Supplement 4 for contaminant-specific parameter values).
2. Compute the total soil loss (summing across all available soil loss mechanisms) for each soil depth (untilled soil, root zone soil, and tilled soil) (Eq. 5-2A in HHRAP).
3. Calculate the deposition term (denoted by *Ds*) used to estimate the soil concentration (Eq. 5-11 or Table B-1-1 in HHRAP). Note that for mercury, the deposition term to soil is modeled slightly differently from all other COPCs (as specified in HHRAP). Table B-1-1 in HHRAP provides supplemental equations used to estimate *Ds* for total mercury, divalent mercury, and methyl mercury. The deposition term to soil is estimated using methods provided in EPA (2005), along with site-specific parameter values where appropriate (see Supplement 4).

4 Calculate soil concentrations (Eqs. 5-1C through 5-1E in HHRAP, and Section 6.2). The soil concentrations are estimated using methods provided in EPA (2005), along with site-specific parameter values where appropriate (see Table 6-3 for a list of site-specific parameter values used in soil modeling).

The following sections present specific equations to support the soil, surface water, and sediment accumulation modeling.

# A.2 Soil Losses

## A.2.1 Soil Loss Due to Leaching (*ksl*)

Soil loss due to leaching (*ksl*) is a function of the amount of water available to generate leachate and soil properties, such as bulk density, soil moisture, soil porosity, and soil sorption properties. Equation 5-5A (Table B-1-5) in the HHRAP is used to calculate the soil loss constant due to leaching for COPCs and ROPCs. The *ksl* is used in the estimation of the total soil loss constant (see Eq. 5-2A in the HHRAP), which is used in the estimation of soil concentrations (Eqs. 5-1C, 5-1D, and 5-1E in the HHRAP). The equation to estimate $ks_l$ is as follows:

$$ksl = \frac{P + I - RO - E_v}{\theta_{sw} \cdot Z_s \cdot [1 + (Kd_s \cdot BD / \theta_{sw})]} \qquad \text{(Eq. 5-5A in HHRAP)}$$

where:

| | | |
|---|---|---|
| *ksl* | = | COPC or ROPC soil loss constant due to leaching ($yr^{-1}$). *ksl* is constituent-specific and depth-specific. If no *ksl* value exists for a constituent, the model uses *ksl* = 0 $yr^{-1}$. |
| *P* | = | average annual precipitation (cm/yr). A value of 18.19 cm/yr (7.16 in./yr for Richland, Washington [Western Regional Climate Center 2002]) is used. |
| *I* | = | average annual irrigation (cm/yr). A value of 0 cm/yr is used (assumed value). |
| *RO* | = | average annual surface runoff from pervious areas (cm/yr). *RO* is site-specific. A value of 2.5 cm/yr (estimated value, assuming that the majority of rainfall recharges or evaporates) is used. |
| $E_v$ | = | average annual evapotranspiration (cm/yr). $E_v$ is site-specific. A value of 16.8 cm/yr is used (Wisiol 1984, Table 2). |
| $\theta_{sw}$ | = | soil volumetric water content (mL water/$cm^3$ soil). $\theta_{sw}$ is site-specific. The recommended default value of 0.2 mL/$cm^3$ is used (Eq. 5-5A of the HHRAP). |
| $Z_s$ | = | soil mixing zone depth (cm). Three different values (depths) are used for $Z_s$: untilled soil (2 cm), root-zone soil (15 cm), and tilled soil (20 cm). |
| $Kd_s$ | = | soil-water partition coefficient (mL water/g soil). $Kd_s$ is constituent-specific. If no $Kd_s$ value exists for an organic constituent, then $Kd_s$ is estimated using Eq. A-2-10 in the HHRAP and a $f_{oc}$ = 0.0044 (fraction of organic carbon in soil, site-specific value from average organic carbon measurements [CCN 150854]), provided the constituent $K_{oc}$ value (soil organic carbon-water partition coefficient) is known. If $Kd_s$ is not available and cannot be estimated, the model uses $Kd_s$ = 0 mL/g. |

*BD* = soil bulk density (g soil/cm$^3$ soil). A site-specific value of 1.3 g/cm$^3$ is used (Halvorson et al. 1998).

### A.2.2 Loss Constant Due to Runoff (*ksr*)

Equation 5-4 (Table B-1-4) in EPA (2005) can be used to calculate the soil-loss constant due to surface runoff (*ksr*) for COPCs and ROPCs. The *ksr* is used in the estimation of the total soil loss constant (see Eq. 5-2A in the HHRAP), which is used in the estimation of soil concentrations (Eqs. 5-1C, 5-1D, and 5-1E in the HHRAP). The equation to estimate *ksr* is as follows:

$$ksr = \left(\frac{RO}{\theta_{sw} \cdot Z_s}\right) \cdot \left(\frac{1}{1 + (Kd_s \cdot BD / \theta_{sw})}\right) \qquad \text{(Eq. 5-4 in HHRAP)}$$

where:

*ksr* = COPC or ROPC soil loss constant due to surface runoff (yr$^{-1}$). *ksr* is constituent and depth-specific.

*RO* = average annual surface runoff from pervious areas (cm/yr). *RO* is site-specific. A value of 2.5 cm/yr is used (estimated value, assuming that the majority of rainfall recharges or evaporates).

$\theta_{sw}$ = soil volumetric water content (mL water/cm$^3$ soil). $\theta_{sw}$ is site-specific. The recommended default value of 0.2 mL/cm$^3$ is used (Eq. 5-5A of the HHRAP).

$Z_s$ = soil mixing zone depth (cm).

$Kd_s$ = soil-water partition coefficient (mL water/g soil). $Kd_s$ is constituent-specific. If no $Kd_s$ value exists for an organic constituent, then $Kd_s$ is estimated using Eq. A-2-10 in the HHRAP and a $f_{oc}$ = 0.0044 (fraction of organic carbon in soil, site-specific value from average organic carbon measurements [CCN 150854]), provided the constituent $K_{oc}$ value (soil organic carbon-water partition coefficient) is known. If $Kd_s$ is not available and cannot be estimated, the model uses $Kd_s$ = 0 mL/g.

*BD* = soil bulk density (g soil/cm$^3$ soil). A site-specific value of 1.3 g/cm$^3$ is used (Halvorson et al.1998).

Since neither natural precipitation nor irrigation provide adequate water to generate surface runoff (refer to Section 6.2), the soil loss constant due to surface runoff is set to zero.

### A.2.3 Loss Constant Due To Volatilization (*ksv*)

Volatile and semivolatile organic COPCs, as well as mercury, emitted in high concentrations may become adsorbed to soil particles and exhibit volatilization losses from soil (*ksv*). This soil loss is a function of the rate of movement of the constituents to the soil surface, the chemical vapor concentration at the soil surface, and the rate at which vapor is carried away by the atmosphere. Equation 5-7A (Table B-1-6) in the HHRAP is used to calculate the soil loss constant due to volatilization for organic COPCs and mercury (*ksv* is assumed to be zero for ROPCs and inorganic COPCs (except for mercury) since these constituents are not considered to be volatile). The *ksv* is used in the estimation of the total soil loss constant, *ks* (see Eq. 5-2A in the HHRAP), which is used in the estimation of soil concentrations (Eqs. 5-1C, 5-1D, and 5-1E in the HHRAP).

The equation to estimate *ksv* is as follows:

$$ksv = \left(\frac{CF \cdot H}{Z_s \cdot Kd_s \cdot R \cdot T_a \cdot BD}\right) \cdot \left(\frac{D_a}{Z_s}\right) \cdot \left[1 - \frac{BD}{\rho_{soil}} - \theta_{sw}\right] \qquad \text{(Eq. 5-7A in HHRAP)}$$

where:

*ksv* = COPC soil loss constant due to volatilization ($yr^{-1}$). *ksv* is constituent-specific. If no *ksv* value can be calculated for a constituent, then the soil loss due to volatilization (*ksv*) is assigned a value of 0 $yr^{-1}$.

*CF* = units conversion factor of 3.1536E+07 (s/yr).

*H* = Henry's Law Constant ($atm \cdot m^3/mol$). If no value is available for *H*, then it is calculated using Eq. A-2-3 in the HHRAP if constituent vapor pressure and water solubility data are available (see Supplement 4). If *H* cannot be determined, then the soil loss due to volatilization (*ksv*) is assigned a value of 0 $yr^{-1}$.

$Z_s$ = soil mixing zone depth (cm). Three different values (depths) are used for $Z_s$: untilled soil (2 cm), root-zone soil (15 cm), and tilled soil (20 cm).

$Kd_s$ = soil-water partition coefficient (mL water/g soil). $Kd_s$ is constituent-specific. If no $Kd_s$ value exists for an organic constituent, then $Kd_s$ is estimated using Eq. A-2-10 in the HHRAP and a $f_{oc}$ = 0.0044 (fraction of organic carbon in soil, site-specific value from average organic carbon measurements [CCN 150854]), provided the constituent $K_{oc}$ value (soil organic carbon-water partition coefficient) is known. If $Kd_s$ is not available and cannot be estimated, the model uses $Kd_s$ = 0 mL/g.

*R* = universal gas constant ($atm \cdot m^3/mol \cdot °K$). A value of $R = 8.205 \times 10^{-5}$ $atm \cdot m^3/mol \cdot °K$ is used.

$T_a$ = ambient air temperature (°K). $T_a$ is site-specific and an average value of 286 °K is used (PNNL 2003. 2004, 2005a, 2006, 2007).

*BD* = soil bulk density (g soil/$cm^3$ soil). A site-specific value of 1.3 g/$cm^3$ is used (Halvorson et al.1998).

$D_a$ = diffusion coefficient of contaminant in air ($cm^2/s$). $D_a$ is constituent-specific (see Supplement 4). If $D_a$ is not available, it can be estimated using Eqs. A-2-4 and A-2-6 in the HHRAP. If no value is available for $D_a$, and if it cannot be estimated, then *ksv* is not calculated and the soil loss due to volatilization is assigned a value of 0 $yr^{-1}$.

$\rho_{soil}$ = Solids particle density (g/$cm^3$). The recommended default value of 2.7 g/$cm^3$ is used.

$\theta_{sw}$ = Soil volumetric water content (mL/$cm^3$ soil). $\theta_{sw}$ is the volumetric fraction of water retained in soil. The recommended default value of 0.2 mL/$cm^3$ is used.

All default values are from Eq. 5-7A in the HHRAP, unless otherwise specified.

### A.2.4 Loss Constant Due to Soil Erosion (*kse*)

Equation 5-3 (HHRAP, Table B-1-3) in the HHRAP is used to calculate the soil loss constant due to soil erosion (*kse*) for COPCs. The *kse* is used in the estimation of the total soil loss constant, *ks* (see Eq. 5-2A in the HHRAP), which is used in the estimation of soil concentrations (Eqs. 5-1C, 5-1D, and 5-1E in the HHRAP). The equation to estimate *kse* is as follows:

$$ks_e = \left(\frac{CF \cdot X_e \cdot SD \cdot ER}{BD \cdot Z_s}\right) \cdot \left(\frac{Kd_s \cdot BD}{\theta_{sw} + (Kd_s \cdot BD)}\right) \qquad \text{(Eq. 5-3 in HHRAP)}$$

where:

| | | |
|---|---|---|
| *kse* | = | COPC soil loss constant due to soil erosion ($yr^{-1}$). *kse* is constituent-specific and depth-specific. |
| *CF* | = | units conversion factor of 0.1 ($g \cdot m^2/kg \cdot cm^2$). |
| $X_e$ | = | unit soil loss ($kg/m^2 \cdot yr$). $X_e$ is site-specific and calculated in Eq. 5-33A in the HHRAP. |
| *SD* | = | watershed sediment delivery ratio (unitless). *SD* is site-specific and is calculated in Eq. 5-34 in the HHRAP. |
| *ER* | = | soil enrichment ratio (unitless). *ER* is site-specific. The following recommended values (EPA 2005) are used: 3 for organic COPCs and 1 for inorganic COPCs. |
| *BD* | = | soil bulk density (g soil/$cm^3$ soil). A site-specific value of 1.3 g/$cm^3$ is used (Halvorson et al. 1998). |
| $Z_s$ | = | soil mixing zone depth (cm). |
| $Kd_s$ | = | soil-water partition coefficient (mL water/g soil). $Kd_s$ is constituent-specific. If no $Kd_s$ value exists for an organic constituent, then $Kd_s$ is estimated using Eq. A-2-10 in the HHRAP and a $f_{oc}$ = 0.0044 (fraction of organic carbon in soil, site-specific value from average organic carbon measurements [CCN 150854]), provided the constituent $K_{oc}$ value (soil organic carbon-water partition coefficient) is known. If $Kd_s$ is not available and cannot be estimated, the model uses $Kd_s$ = 0 mL/g. |
| $\theta_{sw}$ | = | soil volumetric water content (mL water/$cm^3$ soil). $\theta_{sw}$ is site-specific. |

Since neither natural precipitation nor irrigation provide adequate water to cause surface erosion (refer to Section 6.3), the soil loss constant due to erosion is set to zero.

### A.2.5 Soil Loss Constant (*ks*)

Equation 5-2A (HHRAP Table B-1-2) in the HHRAP calculates the total soil loss constant (*ks*) due to biotic and abiotic degradation, radiological decay, leaching, surface runoff, volatilization, and erosion. The *ks* is used in the estimation of soil concentrations (Eqs. 5-1C, 5-1D, and 5-1E in the HHRAP). The site-specific equation to estimate *ks* for all constituents (modified from Eq. 5-2A the HHRAP to include soil loss from radiological decay) is as follows:

$$ks = ksg + kse + ksr + ksl + ksv + k_{decay} \qquad \text{(modified Eq. 5-2A in HHRAP)}$$

where:

| | | |
|---|---|---|
| *ks* | = | total constituent soil loss constant due to biotic and abiotic degradation, radiological decay, leaching, surface runoff, volatilization, and erosion ($yr^{-1}$). *ks* is constituent-specific, site-specific, and depth-specific. If no *ks* value exists for a constituent, the model uses *ks* = 0 $yr^{-1}$. |

*ksg* = COPC soil loss constant due to biotic and abiotic degradation ($yr^{-1}$). *ksg* is COPC-specific, site-specific, and calculated in Eq. A-2-13 in the HHRAP for COPCs (but not for ROPCs). If no *ksg* value exists for a constituent, the model uses *ksg* = 0 $yr^{-1}$.

*kse* = Since neither natural precipitation nor irrigation provide adequate water to generate erosion (refer to Section 6.2), the model uses *kse* = 0 $yr^{-1}$.

*ksr* = Since neither natural precipitation nor irrigation provide adequate water to generate surface runoff (refer to Section 6.2), the model uses *ksr* = 0 $yr^{-1}$.

*ksl* = constituent soil loss constant due to leaching ($yr^{-1}$). *ksl* is constituent-specific, site-specific, depth-specific, and is calculated in Eq. 5-5A in the HHRAP. If no *ksl* value exists for a constituent, the model uses *ksl* = 0 $yr^{-1}$.

*ksv* = constituent soil loss constant due to volatilization ($yr^{-1}$). *ksv* is constituent-specific, site-specific, depth-specific, and is calculated in Eq. 5-7A in the HHRAP. If no *ksv* value exists for a constituent, the model uses *ksv* = 0 $yr^{-1}$.

$k_{decay}$ = ROPC radiological decay constant ($yr^{-1}$). $k_{decay}$ is ROPC-specific, site-specific, and calculated by using the decay half-life in Eq. A-2-13 in the HHRAP for ROPCs (but not for COPCs). If no $k_{decay}$ value exists for a constituent, the model uses $k_{decay}$ = 0 $yr^{-1}$.

# A.3 Soil Terms and Concentration

## A.3.1 Deposition Term (*Ds*)

Equations in Table B-1-1 in the HHRAP are used to calculate the soil deposition term used in soil modeling (*Ds*) for all COPCs. *Ds* is used in the estimation of soil concentrations (Eqs. 5-1C, 5-1D, and 5-1E in the HHRAP). The equation to calculate *Ds* is as follows:

$$Ds = \frac{Q \cdot CF}{Z_s \cdot BD} \cdot \left[F_v \cdot (Dydv + Dywv) + (Dydp + Dywp) \cdot (1 - F_v)\right] \qquad \text{(Table B-1-1 in HHRAP)}$$

where:

*Ds* = deposition term to soil (mg/kg·yr). *Ds* is constituent-specific, site-specific, and depth-specific.

*Q* = constituent-specific emission rate (g/s for COPCs, Ci/s for ROPCs). *Q*, obtained from calculations after the air dispersion modeling, is constituent-specific, site-specific, and stack-specific. If no *Q* value exists for a constituent, the model uses *Q* = 0 g/s or Ci/s.

*CF* = units conversion factor of 100 ($mg \cdot m^2/kg \cdot cm^2$) for COPCs. For ROPCs, the conversion factor is $1 \times 10^8$ ($pCi \cdot m^2/Ci \cdot cm^2$)

$F_v$ = fraction of constituent air concentration in vapor phase (unitless). $F_v$ is constituent-specific, ranges from 0 to 1. Constituents with a vapor fraction less than 0.05 are modeled as entirely particulate with an $F_v$ value of 0 (CCN 097844). When $F_v$ is not available, it is empirically derived for most constituents (except metals and some mercury compounds) using Eqs. A-2-1 and A-2-2 (when appropriate) in the HHRAP.

*Dydv* = unitized yearly average dry deposition from vapor phase (s/m$^2$·yr). *Dydv*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dydv* value exists for a constituent, the model uses *Dydv* = 0 s/m$^2$·yr.

*Dywv* = unitized yearly average wet deposition from vapor phase (s/m$^2$·yr). *Dywv*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dywv* value exists for a constituent, the model uses *Dywv* = 0 s/m$^2$·yr.

*Dydp* = unitized yearly average dry deposition from particle phase (s/m$^2$·yr). *Dydp*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dydp* value exists for a constituent, the model uses *Dydp* = 0 s/m$^2$·yr.

*Dywp* = unitized yearly average wet deposition from particle phase (s/m$^2$·yr). *Dywp*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dywp* value exists for a constituent, the model uses *Dywp* = 0 s/m$^2$·yr.

$Z_s$ = soil mixing zone depth (cm). $Z_s$ is site-specific. Three different values (depths) are used for $Z_s$: untilled soil (2 cm), root-zone soil (15 cm), and tilled soil (20 cm).

*BD* = soil bulk density (g/cm$^3$). A site-specific value of 1.3 g/cm$^3$ is used (Halvorson et al. 1998).

Equations in Table B-1-1 in the HHRAP are used to calculate the soil deposition term used in soil modeling for total mercury [$Ds_{(Hg)}$], divalent mercury [$Ds_{(Hg^{2+})}$], methyl mercury [$Ds_{(MHg)}$], and elemental mercury [$Ds_{(Hg^0)}$]. $Ds_{(Hg)}$ is used in the estimation of soil concentrations (Eqs. 5-1C, 5-1D, and 5-1E in the HHRAP). The equation to estimate $Ds_{(Hg)}$ is as follows:

$$Ds_{(Hg)} = \frac{0.48 \cdot Q \cdot CF}{Z_s \cdot BD} \cdot \left[ F_{v_{(Hg^{2+})}} \cdot (Dydv + Dywv) + (Dydp + Dywp) \cdot \left(1 - F_{v_{(Hg^{2+})}}\right) \right] \quad \text{(Table B-1-1 in HHRAP)}$$

where:

$Ds_{(Hg)}$ = deposition term to soil for total mercury (mg/kg·yr). $Ds_{(Hg)}$ is constituent-specific, site-specific, and depth-specific.

*Q* = total mercury emission rate (g/s). *Q*, obtained from calculations after the air dispersion modeling constituent-specific, site-specific and stack-specific.

*CF* = units conversion factor of 100 (mg·m$^2$/kg·cm$^2$).

$F_{v_{(Hg^{2+})}}$ = fraction of mercury air concentration in vapor phase (unitless). The model uses $F_{v_{(Hg^{2+})}}$ = 0.85 (EPA 2005) for total mercury.

*Dydv* = unitized yearly average dry deposition from vapor phase (s/m$^2$·yr). *Dydv*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dydv* value exists for a constituent, the model uses *Dydv* = 0 s/m$^2$·yr.

*Dywv* = unitized yearly average wet deposition from vapor phase (s/m$^2$·yr). *Dywv*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dywv* value exists for a constituent, the model uses *Dywv* = 0 s/m$^2$·yr.

*Dydp* = unitized yearly average dry deposition from particle phase (s/m$^2$·yr). *Dydp*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dydp* value exists for a constituent, the model uses *Dydp* = 0 s/m$^2$·yr.

*Dywp* = unitized yearly average wet deposition from particle phase (s/m$^2$·yr). *Dywp*, from the air dispersion modeling, is site-specific and stack-specific. If no *Dywp* value exists for a constituent, the model uses *Dywp* = 0 s/m$^2$·yr.

$Z_s$ = soil mixing zone depth (cm). $Z_s$ is site-specific. Three different values (depths) are used for $Z_s$: untilled soil (2 cm), root-zone soil (15 cm), and tilled soil (20 cm).

*BD* = soil bulk density (g/cm$^3$). A site-specific value of 1.3 g/cm$^3$ is used (Halvorson et al. 1998).

A supplemental equation in Table B-1-1 in the HHRAP calculates the soil deposition term used in soil modeling for divalent mercury [$Ds_{(Hg2+)}$]. $Ds_{(Hg2+)}$ is used in the estimation of soil concentrations (Eqs. 5-1C, 5-1D, and 5-1E in the HHRAP). The equation to estimate $Ds_{(Hg2+)}$ is as follows:

$$Ds_{(Hg^{2+})} = 0.98 \cdot Ds_{(Hg)} \qquad \text{(Table B-1-1 in HHRAP)}$$

where:

$Ds_{(Hg^{2+})}$ = deposition term to soil for divalent mercury (mg/kg·yr). $Ds_{(Hg2+)}$ is constituent-specific, site-specific, and depth-specific.

$Ds_{(Hg)}$ = deposition term to soil for total mercury (mg/kg·yr). $Ds_{(Hg)}$ is constituent-specific, site-specific, and depth-specific.

A supplemental equation in Table B-1-1 in the HHRAP calculates the soil deposition term used in soil modeling for methyl mercury [$Ds_{(MHg)}$]. $Ds_{(MHg)}$ is used in the estimation of soil concentrations (Eqs. 5-1C, 5-1D, and 5-1E in the HHRAP). The equation to estimate $Ds_{(MHg)}$ is as follows:

$$Ds_{(MHg)} = 0.02 \cdot Ds_{(Hg)} \qquad \text{(Table B-1-1 in HHRAP)}$$

where:

$Ds_{(MHg)}$ = deposition term to soil for methyl mercury (mg/kg·yr). $Ds_{(MHg)}$ is constituent-specific, site-specific, and depth-specific.

$Ds_{(Hg)}$ = deposition term to soil for total mercury (mg/kg·yr). $Ds_{(Hg)}$ is constituent-specific, site-specific, and depth-specific.

The soil term equation combines the unitized stack deposition rate with the mass flow rate of constituents from the stack and the quantity of soil to arrive at a time-dependent soil concentration.

The time period during which emissions and deposition occur is assumed to start at year zero and cease at year *tD*. Receptor exposures are assumed to occur from year $T_1$ (when the receptor arrives at the exposure location) to $T_2$ (when the receptor departs from the exposure location). Receptors that arrive at the exposure location before the cessation of emissions and deposition ($T_1 < tD$) are considered part of the current exposure scenario. Receptors that arrive at the exposure location at the time of, or subsequent to, cessation of emissions and deposition ($T_1 \geq tD$) are considered part of the future exposure scenario.

### A.3.2 Soil Concentration

Because the hazard quotient associated with noncarcinogenic constituents is based on a threshold dose rather than a lifetime exposure, the highest annual soil concentration ($Cs_{tD}$) occurring during the exposure duration period is used for dose assessment for noncarcinogenic constituents. $Cs_{tD}$ typically occurs at the end of the operating life of the emission source. Because carcinogenic risk is averaged over the lifetime of an individual, the soil concentration averaged over the exposure duration (represented by $Cs$) is used for dose assessment for carcinogenic compounds (EPA 2005).

For constituents that undergo soil loss ($ks > 0$), the concentration is increasing due to continued stack deposition during WTP operations, while simultaneously decreasing due to soil loss. After WTP shutdown, constituent accumulation in the soil stops and the loss continues. Since the soil loss is not zero, the soil concentration is a first-order function of the soil deposition term. In instances where there is no soil loss ($ks = 0$), soil concentration is directly proportional to the rate of deposition and time, and reaches a maximum when deposition ceases (at time $tD$). Figure A-1 of this appendix presents the graphical relationship between instantaneous soil concentration, the corresponding running average soil concentration (for a receptor exposed from the time at which emissions and deposition begin), and time. The figure also shows the running soil concentration average for a receptor that arrives at the time at which emissions and deposition end. The figure is based on an emissions/deposition period ($tD$) of 40 yr. Table A-1 of this appendix summarizes the applicable soil concentration equations for the various combinations of carcinogenicity, soil loss, and exposure timing.

Because soil concentrations may require many years to reach steady state, the equations used to calculate the average soil concentration over the period of receptor exposure are derived by integrating the instantaneous soil concentration equation over the period of receptor exposure and dividing by the exposure period (Section 5.2 of HHRAP, EPA 2005). Furthermore, during the time period following the cessation of WTP emissions, soil concentrations decline gradually because of various soil loss mechanisms and may require many more years to reach steady state. Again, integrating the instantaneous soil concentration equation over the period of exposure and dividing by the exposure period will yield an average exposure concentration for the receptor. Because the function for soil concentration changes from accumulation to degradation when emissions cease, exposures before and after WTP shutdown must be distinguished.

The following discussion presents the formulae for the instantaneous soil concentration followed by the derivation of the formulae used to compute average soil concentrations (for use as soil exposure point concentrations [EPCs]).

#### A.3.2.1 Noncarcinogen Soil Concentration, Current Scenario, No Soil Loss

As stated above, the instantaneous soil concentration for a constituent that does not undergo soil loss is directly proportional to the rate of deposition and time and reaches a maximum when deposition ceases. The instantaneous soil concentration when $ks = 0$ is as follows:

$$Cs_{T2} = Ds \cdot T_2 \qquad \text{(Equation A-1)}$$

where:

$Cs_{T2}$ = instantaneous soil concentration at time $T_2$ (mg/kg soil).

| | | |
|---|---|---|
| *Ds* | = | deposition term to soil (mg/kg·yr). *Ds* is constituent-specific, site-specific, and depth-specific. |
| $T_2$ | = | the time at the end of exposure (yr), usually set equal to *tD*. |

Since the HHRAP stipulates that the highest annual soil concentration ($Cs_{tD}$) occurring during the exposure duration period is used for dose assessment for noncarcinogenic constituents, the following equation is used to estimate the maximum instantaneous soil concentration for constituents where *ks* = 0 and the receptor is exposed in the current scenario ($T_2 \leq tD$):

$$Cs_{tD} = Ds \cdot tD \qquad \text{(Equation A-2)}$$

where:

| | | |
|---|---|---|
| $Cs_{tD}$ | = | instantaneous soil concentration at time *tD* (mg/kg soil) |
| *Ds* | = | deposition term to soil (mg/kg·yr). *Ds* is constituent-specific, site-specific, and depth-specific. |
| $T_2$ | = | the time at the end of exposure (yr), usually set equal to *tD* |
| *tD* | = | the time at the end of emissions/deposition (yr) |

Note that the maximum concentration occurs at time *tD* (at the end of the emissions/deposition period), so the receptor will receive the maximum potential dose when $T_2 \geq tD$, thus, $Cs_{tD}$ is used for the soil EPC of noncarcinogens in the current scenario when there is no soil loss.

### A.3.2.2 Carcinogen Soil Concentration, Current Scenario, No Soil Loss

For receptors that are exposed to carcinogens that do not undergo soil losses (*ks* = 0), the average soil concentration is used as the soil EPC. This quantity is determined by integrating the instantaneous soil concentration over the exposure period, and then dividing by the exposure period. For the current scenario where $T_2 \leq tD$ the instantaneous concentration is described by the following equation:

$$Cs_t = Ds \cdot t \qquad \text{(Equation A-3)}$$

where *t* is the time of exposure. Integrating the instantaneous soil concentration over the exposure period and dividing by the exposure period yields the following:

$$\overline{Cs}_{T_1 \to T_2} = \frac{1}{T_2 - T_1} \cdot \int_{T_1}^{T_2} Cs_t \, dt \to \int_{T_1}^{T_2} Ds \cdot t \;\; dt \qquad \text{(Equation A-4)}$$

which equates to:

$$\frac{1}{T_2 - T_1} \cdot \left( \frac{Ds \cdot T_2^2}{2} - \frac{Ds \cdot T_1^2}{2} \right)$$

$$\rightarrow \quad \left( \frac{Ds}{2} \right) \cdot \left( \frac{1}{T_2 - T_1} \right) \cdot \left[ T_2^2 - T_1^2 \right]$$

$$\rightarrow \quad \left(\frac{Ds}{2}\right)\cdot\left(\frac{1}{T_2 - T_1}\right)\cdot\left(T_2 + T_1\right)\cdot\left(T_2 - T_1\right)$$

which simplifies to:

$$\overline{Cs}_{T_1 \rightarrow T_2} = \frac{Ds}{2}\cdot\left(T_2 + T_1\right)$$

where:

$\overline{Cs}_{T_1 \rightarrow T_2}$ = average soil concentration from time $T_1$ to $T_2$ (mg/kg soil).

$Ds$ = deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific.

$T_2$ = the time at the end of exposure (yr), where $T_2 \leq tD$.

$T_1$ = the time at the start of exposure (yr).

### A.3.2.3 Noncarcinogen Soil Concentration, Current Scenario, With Soil Loss

When a constituent is subject to soil loss, for time $t \leq tD$, there is accumulation of the constituent in the soil from the continued deposition of the constituent; however, there is a simultaneous loss of the constituent at a rate proportional to its value. When a constituent is subject to soil loss, the soil deposition term ($Ds$) is divided by the soil loss term ($ks$) to yield the soil concentration. Soil loss is assumed to follow first order kinetics (EPA 2005).

For noncarcinogens evaluated in the current scenario that undergo soil loss, the instantaneous soil concentration is equal to the soil deposition term ($Ds$) over the soil loss ($ks$) less any soil loss. The equation describing this is as follows:

$$Cs_t = \frac{Ds}{ks} - \frac{Ds \cdot e^{-ks \cdot t}}{ks} \qquad \text{(Equation A-5)}$$

which is commonly written as:

$$Cs_t = \frac{Ds \cdot \left(1 - e^{-ks \cdot t}\right)}{ks}$$

where:

$Cs_t$ = instantaneous soil concentration at time $t$ (mg/kg soil).

$Ds$ = deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific.

$ks$ = overall soil loss constant due to all processes ($yr^{-1}$).

The maximum concentration occurs at time *tD*, and declines afterwards because of soil loss. Substituting *tD* for *t* in the equation above yields:

$$Cs_{tD} = \frac{Ds \cdot \left(1 - e^{-ks \cdot tD}\right)}{ks} \qquad \text{(HHRAP Equation 5-1E)}$$

where:

| | | |
|---|---|---|
| $Cs_{tD}$ | = | instantaneous soil concentration at time *tD* (mg/kg soil). |
| *Ds* | = | deposition term to soil (mg/kg·yr). *Ds* is constituent-specific, site-specific, and depth-specific. |
| *ks* | = | overall soil loss constant due to all processes ($yr^{-1}$). |
| *tD* | = | the time at the end of emissions/deposition (yr). |

### A.3.2.4 Carcinogen Soil Concentration, Current Scenario, With Soil Loss

Integrating the above equation for $Cs_t$ over the period of $T_1$ to $T_2$ (with the constraint $T_1 \le T_2 \le tD$), and dividing by the time period of $T_1$ to $T_2$ will yield the average soil exposure concentration for use in estimating carcinogen dose:

$$\overline{Cs}_{T_1 \to T_2} = \frac{1}{T_2 - T_1} \cdot \int_{T_1}^{T_2} Cs_t \, dt \;\rightarrow\; \frac{1}{T_2 - T_1} \cdot \int_{T_1}^{T_2} \frac{Ds \cdot \left(1 - e^{-ks \cdot t}\right)}{ks} \, dt$$

The solution to this integral is:

$$\overline{Cs}_{T_1 \to T_2} = \frac{Ds}{T_2 - T_1} \cdot \left[ \left(T_2 \cdot ks \cdot e^{ks \cdot T_2} + 1\right) \cdot \frac{e^{-ks \cdot T_2}}{ks^2} + \left(-T_1 \cdot ks \cdot e^{ks \cdot T_1} - 1\right) \cdot \frac{e^{-ks \cdot T_1}}{ks^2} \right]$$

This equation is simplified by combining terms:

$$\frac{Ds}{T_2 - T_1} \cdot \left[ \left( T_2 \cdot ks \cdot e^{ks \cdot T_2} \cdot \frac{e^{-ks \cdot T_2}}{ks^2} + \frac{e^{-ks \cdot T_2}}{ks^2} \right) + \left( -T_1 \cdot ks \cdot e^{ks \cdot T_1} \cdot \frac{e^{-ks \cdot T_1}}{ks^2} - \frac{e^{-ks \cdot T_1}}{ks^2} \right) \right]$$

$$\rightarrow \quad \frac{Ds}{T_2 - T_1} \cdot \left[ \left( \frac{T_2}{ks} + \frac{e^{-ks \cdot T_2}}{ks^2} \right) - \left( \frac{T_1}{ks} + \frac{e^{-ks \cdot T_1}}{ks^2} \right) \right]$$

$$\rightarrow \quad \frac{Ds}{ks \cdot \left(T_2 - T_1\right)} \cdot \left[ \left( T_2 + \frac{e^{-ks \cdot T_2}}{ks} \right) - \left( T_1 + \frac{e^{-ks \cdot T_1}}{ks} \right) \right]$$

where:

$\overline{Cs}_{T_1 \to T_2}$ = average soil concentration from time $T_1$ to $T_2$ (mg/kg soil).

| | | |
|---|---|---|
| $Ds$ | = | deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific. |
| $ks$ | = | overall soil loss constant due to all processes ($yr^{-1}$). |
| $T_2$ | = | the time at the end of exposure (yr), where $T_2 \leq tD$. |
| $T_1$ | = | the time at the start of exposure (yr) , where $T_1 < T_2$. |

If $T_2$ is set to $tD$, then the solution becomes:

$$\overline{Cs}_{tD} = \frac{Ds}{ks \cdot (tD - T_1)} \cdot \left[ \left( tD + \frac{e^{-ks \cdot tD}}{ks} \right) - \left( T_1 + \frac{e^{-ks \cdot T_1}}{ks} \right) \right] \qquad \text{(HHRAP Equation 5-1C)}$$

where:

| | | |
|---|---|---|
| $\overline{Cs}_{tD}$ | = | average soil concentration at time $tD$ (mg/kg soil) |
| $Cs_{T2}$ | = | instantaneous soil concentration at time $T_2$ (mg/kg soil) |
| $Ds$ | = | deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific |
| $ks$ | = | overall soil loss constant due to all processes ($yr^{-1}$) |
| $tD$ | = | the time at the end of emissions/deposition (yr) |

### A.3.2.5 Noncarcinogen Soil Concentration, Future Scenario, No Soil Loss

Since there is no loss, the contaminant level is assumed to remain constant subsequent to cessation of emissions, therefore, $Cs_{tD}$ is used for the soil EPC of noncarcinogens in the future scenario when there is no soil loss.

$$Cs_{T2} = Cs_{tD} = Ds \cdot tD \qquad \text{(Equation A-6)}$$

where:

| | | |
|---|---|---|
| $Cs_{T2}$ | = | instantaneous soil concentration at time $T_2$ (mg/kg soil). |
| $Cs_{tD}$ | = | instantaneous soil concentration at time $tD$, or subsequent to time $tD$ (mg COPC/kg soil). |
| $Ds$ | = | deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific. |
| $tD$ | = | the time at the end of emissions/deposition (yr). |

### A.3.2.6 Carcinogen Soil Concentration, Future Scenario, No Soil Loss

The maximum concentration occurs at time $tD$, and since there is no soil loss or continued emissions, the corresponding average concentration cannot exceed the level it reaches at time $tD$, thus, $Cs_{tD}$ is used for future scenarios:

$$\overline{Cs}_{tD \to T_2} = \frac{1}{T_2 - tD} \cdot \int_{tD}^{T_2} Cs_{tD} \; dt \qquad \text{(Equation A-7)}$$

$$\rightarrow \quad \overline{Cs}_{tD \to T_2} = \frac{1}{T_2 - tD} \cdot \int_{tD}^{T_2} Ds \cdot t \; dt \qquad \text{(Equation A-8)}$$

the solution to this integral is:

$$\frac{Ds}{T_2 - tD} \cdot \left( \frac{T_2^{\,2} - tD^2}{2} \right) \qquad \text{(Equation A-9)}$$

$$\rightarrow \quad \frac{Ds}{2} \cdot \left( T_2 + tD \right) \qquad \text{(Equation A-10)}$$

where:

$\overline{Cs}_{tD \to T_2}$ = average soil concentration from time $tD$ to $T_2$ (mg/kg soil).

$Ds$ = deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific.

$T_2$ = the time at the end of exposure (yr), where $tD \leq T_2$.

$tD$ = the time at the end of emissions/deposition (yr).

### A.3.2.7 Noncarcinogen Soil Concentration, Future Scenario, With Soil Loss

For noncarcinogens evaluated in the future scenario that undergo soil loss, the instantaneous soil concentration is equal to the soil concentration at the time emissions/deposition ceased, less any decline in soil concentration due to losses. At time $tD$ (time emissions/deposition ceased), the soil concentration is as follows:

$$Cs_{tD} = \frac{Ds \cdot \left(1 - e^{-ks \cdot tD}\right)}{ks} \qquad \text{(HHRAP Equation 5-1E)}$$

and at some time in the future, $T_2$, the remaining concentration (with the constraint of $tD \leq T_2$) is:

$$Cs_{T2} = Cs_{tD} \cdot e^{-ks \cdot (T_2 - tD)} \qquad \text{(Equation A-11)}$$

However, since noncarcinogenic risk is based on a threshold value (the reference dose), HHRAP (Section 5.2.1) recommends that the maximum instantaneous concentration should be used for risk assessment. Therefore, the following equation applies:

$$Cs_{T2} = Cs_{tD} = \frac{Ds \cdot \left(1 - e^{-ks \cdot tD}\right)}{ks} \qquad \text{(Equation A-12)}$$

where:

| | | |
|---|---|---|
| $Cs_{T2}$ | = | instantaneous soil concentration at time $T_2$ (mg/kg soil). |
| $Cs_{tD}$ | = | soil concentration at time $tD$ (mg COPC/kg soil). |
| $Ds$ | = | deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific. |
| $ks$ | = | overall soil loss constant due to all processes ($yr^{-1}$). |
| $tD$ | = | the time at the end of emissions/deposition (yr). |
| $T_2$ | = | the time at the end of exposure (yr), where $tD \leq T_2$. |

The maximum concentration occurs at time $tD$ and declines afterwards because of soil loss.

### A.3.2.8 Carcinogen Soil Concentration, Future Scenario, With Soil Loss

Integrating the above equation for $Cs$ over the period of $tD$ to $T_2$ (with the constraint $tD \leq T_2$), and dividing by the time period of $tD$ to $T_2$ will yield the average soil exposure concentration for use in estimating carcinogen dose.

$$\overline{Cs}_{tD \to T_2} = \frac{1}{T_2 - tD} \cdot \int_{tD}^{T_2} Cs_{tD} \cdot e^{-ks \cdot (t - tD)} dt \qquad \text{(Equation A-13)}$$

$$\rightarrow \quad \frac{Cs_{tD}}{T_2 - tD} \cdot \int_{tD}^{T_2} e^{-ks \cdot (t - tD)} dt$$

the solution to this integral is:

$$\frac{Cs_{tD}}{T_2 - tD} \cdot \left( \frac{1}{ks} + \frac{-e^{-ks \cdot (T_2 - tD)}}{ks} \right)$$

this equation is simplified by combining terms:

$$\frac{Cs_{tD}}{ks \cdot (T_2 - tD)} \cdot \left(1 - e^{-ks \cdot (T_2 - tD)}\right)$$

where:

| | | |
|---|---|---|
| $\overline{Cs}_{tD \to T_2}$ | = | average soil concentration from time $tD$ to $T_2$ (mg/kg soil) |
| $Cs_{tD}$ | = | instantaneous soil concentration at time $tD$ ($Cs_{tD} = Ds \cdot (1 - e^{-ks \cdot tD})/ks$), (mg/kg soil) |
| $tD$ | = | the time at the end of emissions/deposition (yr) |
| $T_2$ | = | the time at the end of exposure (yr), where $tD \leq T_2$. |
| $Ds$ | = | deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific. |
| $ks$ | = | overall soil loss constant due to all processes ($yr^{-1}$) |

To compute the average soil concentration for exposure over a distinct time interval (from time $T_1$ to $T_2$), integrate of the timeframe of interest, and divide by the time.

$$\overline{Cs}_{T_1 \to T_2} = \frac{1}{T_2 - T_1} \cdot \int_{T_1}^{T_2} Cs_{tD} \cdot e^{-ks\cdot(t-tD)} dt \qquad \text{(Equation A-14)}$$

this equation equates to:

$$\frac{Cs_{tD}}{T_2 - T_1} \cdot \left( \int_{tD}^{T_2} e^{-ks\cdot(t-tD)} dt - \int_{tD}^{T_1} e^{-ks\cdot(t-tD)} dt \right)$$

which yields:

$$\frac{Cs_{tD}}{ks \cdot (T_2 - T_1)} \cdot \left( e^{-ks\cdot(T_1 - tD)} - e^{-ks\cdot(T_2 - tD)} \right)$$

where:

$\overline{Cs}_{T_1 \to T_2}$ = average soil concentration from time $T_1$ to $T_2$ (mg/kg soil)

$Cs_{tD}$ = instantaneous soil concentration at time $tD$ ($Cs_{tD} = Ds\cdot(1-e^{-ks\cdot tD})/ks$), (mg/kg soil)

$Ds$ = deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific

$ks$ = overall soil loss constant due to all processes ($yr^{-1}$)

$tD$ = the time at the end of emissions/deposition (yr)

$T_2$ = the time at the end of exposure (yr), where $tD < T_2$

$T_1$ = the time at the start of exposure (yr), where $tD \leq T_1 < T_2$

### A.3.2.9 Noncarcinogen Soil Concentration, Spanning Current and Future Scenarios, With No Soil Loss

Some receptors may have an exposure duration that exceeds the period on emissions/deposition. In such cases, both current and future scenario equations must be combined. For constituents that do not undergo soil loss, the contaminant level increases throughout the emissions period, and is assumed to remain constant subsequent to cessation of emissions. Accordingly, instantaneous soil concentration is computed as follows:

$$Cs_{T2} = Ds \cdot tD \qquad \text{(Equation A-15)}$$

where:

$Cs_{T2}$ = instantaneous soil concentration at time $T_2$ (mg/kg soil)

$Ds$ = deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific

$tD$ = the time at the end of emissions/deposition (yr)

### A.3.2.10 Carcinogen Soil Concentration, Spanning Current and Future Scenarios, With No Soil Loss

For a receptor whose exposure time spans the period of emissions and a time period after cessation of emissions, the equation for the average soil concentration must also include the contribution of post-emissions/deposition exposure with exposure during emissions/deposition:

$$\overline{Cs}_{T_2 \to T_1} = \frac{1}{T_2 - T_1} \cdot \left( \int_{T_1}^{tD} Cs_t \, dt + \int_{tD}^{T_2} Cs_t \, dt \right) \qquad \text{(Equation A-16)}$$

which equates to:

$$\frac{1}{T_2 - T_1} \cdot \left( \int_{T_1}^{tD} Ds \cdot t \; dt + \int_{tD}^{T_2} Cs_{tD} \, dt \right)$$

which yields:

$$\frac{1}{T_2 - T_1} \cdot \left[ \frac{Ds \cdot tD}{2} \cdot (T_2 - tD) + \frac{Ds}{2} \cdot \left( tD^2 - T_1^2 \right) \right]$$

$$\rightarrow \quad \frac{Ds}{2 \cdot (T_2 - T_1)} \cdot \left( T_2 \cdot tD - T_1^2 \right)$$

where:

| | | |
|---|---|---|
| $\overline{Cs}_{T_1 \to T_2}$ | = | average soil concentration from time $T_1$ to $T_2$ (mg/kg soil) |
| $Ds$ | = | deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific |
| $T_2$ | = | the time at the end of exposure (yr), where $T_2 \geq tD$. |
| $T_1$ | = | the time at the start of exposure (yr), where $T_1 < tD$ |
| $tD$ | = | the time at the end of emissions/deposition (yr) |

### A.3.2.11 Noncarcinogen Soil Concentration, Spanning Current and Future Scenarios, With Soil Loss

For noncarcinogens that span the current and future scenario and undergo soil loss, the instantaneous soil concentration is equal to the soil concentration at the time emissions/deposition ceased, less any decline in soil concentration due to losses. At time $tD$ (time emissions/deposition ceased), the soil concentration is as follows:

$$Cs_{tD} = \frac{Ds \cdot \left(1 - e^{-ks \cdot tD}\right)}{ks} \qquad \text{(HHRAP Equation 5-1E)}$$

and at some time in the future, $T_2$, the remaining concentration (with the constraint of $tD \leq T_2$) is:

$$Cs_{T2} = Cs_{tD} \cdot e^{-ks \cdot (T_2 - tD)} \qquad \text{(Equation A-17)}$$

However, since noncarcinogenic risk is based on a threshold value (the reference dose), HHRAP (Section 5.2.1) recommends that the maximum instantaneous concentration should be used for risk assessment. Therefore, the following equation applies:

$$Cs_{T2} = Cs_{tD} = \frac{Ds \cdot \left(1 - e^{-ks \cdot tD}\right)}{ks} \qquad \text{(Equation A-18)}$$

where:

| | | |
|---|---|---|
| $Cs_{T2}$ | = | instantaneous soil concentration at time $T_2$ (mg/kg soil). |
| $Cs_{tD}$ | = | soil concentration at time $tD$ (mg COPC/kg soil). |
| $Ds$ | = | deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific. |
| $ks$ | = | overall soil loss constant due to all processes ($yr^{-1}$). |
| $tD$ | = | the time at the end of emissions/deposition (yr). |
| $T_2$ | = | the time at the end of exposure (yr), where $tD \leq T_2$. |

### A.3.2.12 Carcinogen Soil Concentration, Spanning Current and Future Scenarios, With Soil Loss

Accounting for the contribution of post-emissions/deposition exposure (Section A.3.2.4 ) with exposure during emissions/deposition (Section A.3.2.8 ), the following integral is used to derive the average soil concentration:

$$\overline{Cs}_{T_1 \to T_2} = \frac{1}{T_2 - T_1} \cdot \left( \int_{T_1}^{tD} Cs_t \, dt + \int_{tD}^{T_2} Cs_{tD} \cdot e^{-ks \cdot (t - tD)} dt \right) \qquad \text{(Equation A-19)}$$

which equates to:

$$\frac{1}{T_2 - T_1} \cdot \left( \int_{T_1}^{tD} \frac{Ds \cdot \left(1 - e^{-ks \cdot t}\right)}{ks} dt + Cs_{tD} \cdot \int_{tD}^{T_2} e^{-ks \cdot (t - tD)} dt \right)$$

the solution to this integral is:

$$\frac{1}{T_2 - T_1} \cdot \left[ \frac{Ds}{ks} \cdot \left[ \left( tD + \frac{e^{-ks \cdot tD}}{ks} \right) - \left( T_1 + \frac{e^{-ks \cdot T_1}}{ks} \right) \right] + \frac{Cs_{tD}}{ks} \cdot \left(1 - e^{-ks \cdot (T_2 - tD)}\right) \right]$$

this equation is simplified by combining and canceling terms:

$$\frac{1}{T_2 - T_1} \cdot \left[ \left( \frac{Ds \cdot tD}{ks} + \frac{Ds \cdot e^{-ks \cdot tD}}{ks^2} \right) - \left( \frac{Ds \cdot T_1}{ks} + \frac{Ds \cdot e^{-ks \cdot T_1}}{ks^2} \right) + \frac{Cs_{tD}}{ks} \cdot \left( 1 - e^{-ks \cdot (T_2 - tD)} \right) \right]$$

$$\rightarrow \quad \frac{1}{T_2 - T_1} \cdot \left[ \frac{Ds \cdot tD - Ds \cdot T_1}{ks} + \frac{Ds \cdot e^{-ks \cdot tD} - Ds \cdot e^{-ks \cdot T_1}}{ks^2} + \frac{Cs_{tD}}{ks} \cdot \left( 1 - e^{-ks \cdot (T_2 - tD)} \right) \right]$$

$$\rightarrow \quad \frac{Ds}{ks \cdot (T_2 - T_1)} \cdot \left[ tD - T_1 + \frac{e^{-ks \cdot tD} - e^{-ks \cdot T_1}}{ks} + \frac{Cs_{tD}}{Ds} \cdot \left( 1 - e^{-ks \cdot (T_2 - tD)} \right) \right]$$

where:

| | | |
|---|---|---|
| $\overline{Cs}_{T_1 \rightarrow T_2}$ | = | average soil concentration from time $T_1$ to $T_2$ (mg/kg soil) |
| $Cs_{tD}$ | = | instantaneous soil concentration at time $tD$ ($Cs_{tD} = Ds \cdot (1 - e^{-ks \cdot tD})/ks$), (mg/kg soil) |
| $Ds$ | = | deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific |
| $T_2$ | = | the time at the end of exposure (yr), where $T_2 > tD$ |
| $T_1$ | = | the time at the start of exposure (yr), where $T_1 < tD$ |
| $tD$ | = | the time at the end of emissions/deposition (yr) |

Note that if $T_1$ is set to coincide with the start of emissions/deposition ($T_1 = 0$) as assumed in the HHRAP, then the equation above can be simplified as follows:

$$\frac{Ds}{ks \cdot (T_2 - T_1)} \cdot \left[ tD - 0 + \frac{e^{-ks \cdot tD} - 1}{ks} + \frac{Cs_{tD}}{Ds} \cdot \left( 1 - e^{-ks \cdot (T_2 - tD)} \right) \right]$$

$$\rightarrow \quad \frac{1}{T_2 - T_1} \cdot \left[ \frac{Ds \cdot tD}{ks} + \frac{Ds \cdot \left( e^{-ks \cdot tD} - 1 \right)}{ks^2} + \frac{Cs_{tD}}{ks} \cdot \left( 1 - e^{-ks \cdot (T_2 - tD)} \right) \right]$$

$$\rightarrow \quad \frac{1}{T_2 - T_1} \cdot \left[ \frac{Ds \cdot tD}{ks} + \frac{-Ds \cdot \left( 1 - e^{-ks \cdot tD} \right)}{ks} \cdot \frac{1}{ks} + \frac{Cs_{tD}}{ks} \cdot \left( 1 - e^{-ks \cdot (T_2 - tD)} \right) \right]$$

and substituting $Cs_{tD} = \dfrac{Ds \cdot \left( 1 - e^{-ks \cdot tD} \right)}{ks}$

$$\rightarrow \quad \frac{1}{T_2 - T_1} \cdot \left[ \frac{Ds \cdot tD}{ks} + \frac{-Cs_{tD}}{ks} + \frac{Cs_{tD}}{ks} \cdot \left( 1 - e^{-ks \cdot (T_2 - tD)} \right) \right]$$

$$\rightarrow \quad \frac{\dfrac{Ds \cdot tD - Cs_{tD}}{ks} + \left( \dfrac{Cs_{tD}}{ks} \right) \cdot \left( 1 - e^{-ks \cdot (T_2 - tD)} \right)}{T_2 - T_1} \qquad \text{(HHRAP Equation 5-1D)}$$

Figure A-1 shows a plot of soil concentration (*Cs*) with time (*T*) for the various equations above. The plot lines represent the change in exposure point concentration as it might correspond with a given exposure duration as represented by the light blue bars. The bars represent the receptor's exposure on the timeline of WTP operations and post-operations (current and future exposure scenarios). The bars help illustrate how receptor exposure scenarios have been developed to conservatively coincide maximum exposures.

The blue line shows the instantaneous soil concentration with time for cases where there is no known soil loss ($Cs = Ds \cdot T$, $ks = 0$). The magenta line below shows an average soil concentration, without soil loss, that is, it represents the area under the blue line, divided by time ($Cs = Ds \cdot T \div 2$, $ks = 0$ $Cs = (Ds \cdot T) \div 2, ks = 0$). In both cases, at the end of operations (*tD*), the soil concentration has reached a maximum, $Cs_{tD}$ ($Cs_{tD} = Ds \cdot tD$ and $Cs_{tD} = Ds \cdot tD \div 2$ $Cs_{tD} = (Ds \cdot tD) \div 2$, respectively) since this is when deposition of emitted particles ceases.

The green line shows the instantaneous soil concentration with soil loss ($Cs = (Ds/ks) \cdot [1-e^{-ks \cdot T}]$, $ks \neq 0$). At time *tD* soil concentration has reached its maximum, $Cs_{tD}$ ($Cs_{tD} = (Ds/ks) \cdot [1-e^{-ks \cdot tD}]$ $Cs_{tD} = \frac{Ds}{ks} \cdot [1 - e^{-ks \cdot tD}]$), when operations cease and deposition is no longer occurring, at which point the soil concentration begins decreasing due to losses ($Cs = [Cs_{tD}/(ks \cdot (T-tD)] \cdot [1-e^{-ks \cdot (T-tD)}]$).

The red line represents average soil concentration, with time and soil loss ($Cs = [Ds/(ks \cdot T)] \cdot (T+[e^{-ks \cdot T}-1]/ks)$, $ks \neq 0$). Because the line plots the average concentration, the line represents the area under the green instantaneous soil concentration line, divided by time. As with other plots of soil concentration, the maximum, $Cs_{tD}$ ($Cs_{tD} = Ds \cdot [1-e^{-ks \cdot tD}]/ks$) $Cs_{tD} = Ds \cdot [1 - e^{-ks \cdot tD}]/ks$, is reached at the cessation of WTP operations. The plot shows that as a receptor remains exposed, the average EPC declines as soil loss occurs ($Cs = [(Ds \cdot tD - Cs_{tD})/(ks \cdot T)] + [Cs_{tD}/(ks \cdot T)] \cdot [1-e^{-ks \cdot (T-tD)}]$).

The figure shows that for current scenarios, the EPC is conservatively computed assuming exposure concludes at time = *tD* for any case where the exposure duration (*ED*) is less than 40 years, thus the corresponding soil EPC is $Cs_{tD}$. For current exposure scenarios where the exposure duration (*ED*) is greater than 40 years, the figure shows the value of $Cs_{tD}$ is used as the EPC for assessing exposures where there is no known soil loss. In cases where there is soil loss, noncarcinogen exposures are bounded at EPC = $Cs_{tD}$, however, carcinogen exposures (where an average soil concentration is used), the EPC declines after the cessation of emissions due to the effect of soil loss. The figure also shows the start of receptor exposure is set to coincide with time = *tD* in the future scenarios. Future exposures are also bounding because the EPC is *CstD* when the soil loss is zero or unknown. When there are soil losses, receptor exposure occurs over the period of highest concentration as opposed to the tail end of the assessment period when soil concentrations are tapering off. As an example, the figure shows where (on the plotted lines) the exposure concentrations correspond with an *ED* = 70 yr for a current exposure scenario spanning plant operations, and *ED* = 30 yr and *ED* = 40 yr for a future exposure scenario.

# A.4 Water Body Load

## A.4.1 Direct Deposition Load to Water Body ($L_{DEP}$)

Equation 5-29 (Table B-4-8) in the HHRAP calculates the average load to the water body from direct deposition of wet and dry particles and wet and dry vapors onto the surface of the water body ($L_{DEP}$) for

all constituents (ROPCs and COPCs), except divalent mercury and methyl mercury. $L_{DEP}$ is used in the estimation of the total load to the surface water body (see Eq. 5-28 in the HHRAP). The equation to estimate $L_{DEP}$ is:

$$L_{DEP} = Q \cdot \left[F_v \cdot Dytwv + \left(1 - F_v\right) \cdot Dytwp\right] \cdot A_w \qquad \text{(Eq. 5-29 in HHRAP)}$$

where:

$L_{DEP}$ = total (wet and dry) particle-phase and total (wet and dry) vapor-phase direct deposition load to water body (g/yr for COPCs and Ci/yr for ROPCs). $L_{DEP}$ is constituent-specific and site-specific.

$Q$ = constituent-specific emission rate (g/s for COPCs and Ci/s for ROPCs). $Q$, obtained from calculations after the air dispersion modeling, is constituent-specific, site-specific, and stack-specific. If no value exists for $Q$, a value of 0 g/s (for COPCs) or 0 Ci/s (for ROPCs) is used.

$F_v$ = fraction of constituent air concentration in vapor phase (unitless). $F_v$ is constituent-specific, ranges from 0 to 1. Constituents with a vapor fraction less than 0.05 are modeled as entirely particulate with an $F_v$ value of 0 (CCN 097844). When $F_v$ is not available, it is empirically derived for most constituents (except metals and some mercury compounds) using (when appropriate) Eqs. A-2-1 and A-2-2 in the HHRAP. In accordance with the HHRAP, the $F_v$ of metals is assumed to be zero.

*Dytwv* = unitized yearly average wet deposition from vapor phase over water body (s/m$^2$·yr). *Dytwv*, from the air dispersion modeling, is site-specific and stack-specific, and is the sum of *Dywv* and *Dydv*.

*Dytwv* = *Dywv* + *Dydv* *Dywv* and *Dydv* are defined in A.3.1

If no *Dytwv* value exists for a constituent, the model uses *Dytwv* = 0 s/m$^2$·yr.

*Dytwp* = unitized yearly average total (wet and dry) deposition from particle phase over water body (s/m$^2$·yr). *Dytwp*, from the air dispersion modeling, is site-specific and stack-specific, and is the sum of *Dywp* and *Dydp*.

*Dytwp* = *Dywp* + *Dydp* *Dywp* and *Dydp* are defined in A.3.1

If no *Dytwp* value exists for a constituent, the model uses *Dytwp* = 0 s/m$^2$·yr.

$A_w$ = average annual water body surface area (m$^2$). $A_w$ is site-specific, a value of $A_w$ = 3.652E+07 m$^2$ is used (PNNL 2005b).

Table B-4-8 in the HHRAP also contains the equation to calculate the average load to the water body from direct deposition of wet and dry particles and wet and dry vapors onto the surface of the water body for total mercury [$L_{DEP(Hg)}$]. $L_{DEP(Hg)}$ is used in the estimation of the total load to the surface water body (see Eq. 5-28 in the HHRAP). The equation to estimate $L_{DEP(Hg)}$ is:

$$L_{DEP(Hg)} = 0.48 \cdot Q_{(Hg)} \cdot \left\lfloor F_{v(Hg^{2+})} \cdot Dytwv + (1 - F_{v(Hg^{2+})}) \cdot Dytwp \right\rfloor \cdot A_w \qquad \text{(Table B-4-8 in HHRAP)}$$

where:

$L_{DEP(Hg)}$ = total (wet and dry) particle phase and total (wet and dry) vapor phase direct deposition load to water body for total mercury (g/yr). $L_{DEP(Hg)}$ is constituent-specific and site-specific.

$Q_{(Hg)}$ = COPC-specific emission rate for total mercury (g/s). $Q_{(Hg)}$, obtained from calculations after the air dispersion modeling, is constituent-specific, site-specific and stack-specific.

$F_{v_{(Hg^{2+})}}$ = fraction of mercury air concentration in vapor phase (unitless). The model uses $F_{v_{(Hg^{2+})}}$ = 0.85 (refer to Table B-4-8 in the HHRAP) for total mercury.

*Dytwv* = unitized yearly average wet deposition from vapor phase over water body (s/m$^2$·yr). *Dytwv*, from the air dispersion modeling, is site-specific and stack-specific, and is the sum of *Dywv* and *Dydv*.

*Dytwv* = *Dywv* + *Dydv* *Dywv* and *Dydv* are defined in A.3.1

If no *Dytwv* value exists for a constituent, the model uses *Dytwv* = 0 s/m$^2$·yr.

*Dytwp* = unitized yearly average total (wet and dry) deposition from particle phase over water body (s/m$^2$·yr). *Dytwp*, from the air dispersion modeling, is site-specific and stack-specific, and is the sum of *Dywp* and *Dydp*.

*Dytwp* = *Dywp* + *Dydp* *Dywp* and *Dydp* are defined in A.3.1

If no *Dytwp* value exists for a constituent, the model uses *Dytwp* = 0 s/m$^2$·yr.

$A_w$ = average annual water body surface area (m$^2$). $A_w$ is site-specific, a value of $A_w$ = 3.652E+07 m$^2$ is used (PNNL 2005b).

### A.4.2 Diffusion Load to Water Body ($L_{dif}$)

Equation 5-30 (Table B-4-12) in the HHRAP calculates the load to the water body due to dry vapor diffusion ($L_{dif}$) for all constituents except divalent mercury and methyl mercury. $L_{dif}$ is used in the estimation of the total load to the surface water body (see Eq. 5-28 in the HHRAP). The equation to estimate $L_{dif}$ is:

$$L_{dif} = \frac{K_v \cdot Q \cdot F_v \cdot Cywv \cdot A_w \cdot CF}{\frac{H}{R \cdot T_{wk}}} \qquad \text{(Eq. 5-30 in HHRAP)}$$

where:

$L_{dif}$ = vapor phase COPC or ROPC dry deposition diffusion load to water body (g/yr for COPCs and Ci/yr for ROPCs). $L_{dif}$ is constituent-specific and site-specific.

$K_v$ = overall transfer rate coefficient (m/yr). $K_v$ is constituent-specific, site-specific, and is calculated in Eq. 5-40 in the HHRAP.

$Q$ = constituent-specific emission rate (g/s for COPCs and Ci/s for ROPCs). $Q$, obtained from calculations after the air dispersion modeling, is constituent-specific, site-specific, and stack-specific. If no value exists for $Q$, a value of 0 g/s (for COPCs) or 0 Ci/s (for ROPCs) is used.

$F_v$ = fraction of constituent air concentration in vapor phase (unitless). $F_v$ is constituent-specific, ranges from 0 to 1. Constituents with a vapor fraction less than 0.05 are modeled as entirely particulate with an $F_v$ value of 0 (CCN 097844). When $F_v$ is not

available, it is empirically derived for most constituents (except metals and some mercury compounds) using (when appropriate) Eqs. A-2-1 and A-2-2 in the HHRAP. In accordance with the HHRAP, the $F_v$ of metals is assumed to be zero.

*Cywv* = unitized yearly average air concentration from vapor phase over the water body (μg·s/g·m$^3$ for COPCs and μCi·s/Ci·m$^3$ for ROPCs). *Cywv*, from the air dispersion modeling, is site-specific and stack-specific. *Cywv* is the same as *Cyv* , the unitized yearly air concentration from vapor phase. If no value exists for *Cywv*, the model uses *Cywv* = 0 μg·s/g·m$^3$ for COPCs and *Cywv* = 0 μCi·s/Ci·m$^3$ for ROPCs.

$A_w$ = average annual water body surface area (m$^2$). $A_w$ is site-specific, a value of $A_w$ = 3.652E+07 m$^2$ is used (PNNL 2005b).

*CF* = units conversion factor of $1 \times 10^{-6}$ (g/μg for COPCs and Ci/μCi for ROPCs)

*H* = Henry's Law Constant (atm·m$^3$/mol). *H* is constituent-specific. If no value is available for *H*, then it is estimated using Eq. A-2-3 in the HHRAP if constituent vapor pressure and water solubility data are available, otherwise, the model uses $L_{dif}$ = 0 g/yr for COPCs and Ci/yr for ROPCs.

*R* = universal gas constant (atm·m$^3$/mol-°K). A value of $R = 8.205 \times 10^{-5}$ atm·m$^3$/mol·°K is used.

$T_{wk}$ = water body temperature (°K). $T_{wk}$ is site-specific and an average value of 285 °K is used (PNNL 2003. 2004, 2005a, 2006, 2007).

Table B-4-12 in the HHRAP also provides an equation to calculate the load to the water body due to dry vapor diffusion for total mercury [$L_{dif(Hg)}$]. $L_{dif(Hg)}$ is used in the estimation of the total load to the surface water body (see Eq. 5-28 in the HHRAP). The equation to estimate $L_{dif(Hg)}$ is:

$$L_{dif(Hg)} = \frac{K_v \cdot 0.48 \cdot Q_{(Hg)} \cdot F_{v_{(Hg^{2+})}} \cdot Cywv \cdot A_w \cdot CF}{\frac{H}{R \cdot T_{wk}}} \qquad \text{(Table B-4-12 in HHRAP)}$$

where:

$L_{dif(Hg)}$ = vapor phase COPC dry deposition diffusion load to water body for total mercury (g/yr). $L_{dif(Hg)}$ is constituent-specific and site-specific.

$K_v$ = overall transfer rate coefficient for total mercury (m/yr). $K_v$ is constituent-specific, site-specific, and is calculated in Eq. 5-40 in the HHRAP.

$Q_{(Hg)}$ = constituent-specific emission rate for total mercury (g/s). $Q_{(Hg)}$, obtained from calculations after the air dispersion modeling, is constituent-specific, site-specific, and stack-specific.

$F_{v_{(Hg^{2+})}}$ = fraction of mercury air concentration in vapor phase (unitless). The model uses $F_{v_{(Hg^{2+})}}$ = 0.85 (refer to Table B-4-12 in the HHRAP) for total mercury.

*Cywv* = unitized yearly average air concentration from vapor phase over the water body (μgs/g·m$^3$). *Cywv*, from the air dispersion modeling, is site-specific and stack-specific. *Cywv* is the same as *Cyv* , the unitized yearly air concentration from vapor phase.

$A_w$ = average annual water body surface area (m$^2$). $A_w$ is site-specific, a value of $A_w$ = 3.652E+07 m$^2$ is used (PNNL 2005b).

*CF* = units conversion factor of $1 \times 10^{-6}$ (g/μg)

*H* = Henry's Law Constant for total mercury (atm·m$^3$/mol).

*R* = universal gas constant (atm·m$^3$/mol-°K). A value of $R = 8.205 \times 10^{-5}$ atm·m$^3$/mol-°K is used.

$T_{wk}$ = water body temperature (°K). $T_{wk}$ is site-specific and an average value of 285 °K is used (PNNL 2003. 2004, 2005a, 2006, 2007).

### A.4.3 Runoff Load From Impervious Surfaces ($L_{RI}$)

Equation 5-31 (Table B-4-9) in the HHRAP calculates the average runoff load to the water body from impervious surfaces in the watershed from which runoff is conveyed directly to the water body ($L_{RI}$), for all constituents except total mercury. $L_{RI}$ is used in the estimation of the total load to the surface water body (see Eq. 5-28 in the HHRAP). The equation to estimate $L_{RI}$ is:

$$L_{RI} = Q \cdot \left[F_v \cdot Dytwv + \left(1 - F_v\right) \cdot Dytwp\right] \cdot A_I \qquad \text{(Eq. 5-31 in HHRAP)}$$

where:

$L_{RI}$ = runoff load from impervious surfaces (g/yr for COPCs and Ci/yr for ROPCs). $L_{RI}$ is constituent-specific and site-specific.

*Q* = COPC or ROPC-specific emission rate (g/s for COPCs and Ci/s for ROPCs). *Q*, obtained from calculations after the air dispersion modeling, is constituent-specific, site-specific, and stack-specific. If no value exists for *Q*, a value of 0 g/s (for COPCs) or 0 Ci/s (for ROPCs) is used.

$F_v$ = fraction of constituent air concentration in vapor phase (unitless). $F_v$ is constituent-specific, ranges from 0 to 1. Constituents with a vapor fraction less than 0.05 are modeled as entirely particulate with an $F_v$ value of 0 (CCN 097844). When $F_v$ is not available, it is empirically derived for most constituents (except metals and some mercury compounds) using (when appropriate) Eqs. A-2-1 and A-2-2 in the HHRAP. In accordance with the HHRAP, the $F_v$ of metals is assumed to be zero.

*Dytwv* = unitized yearly average wet deposition from vapor phase over water body (s/m$^2$·yr). *Dytwv*, from the air dispersion modeling, is site-specific and stack-specific, and is the sum of *Dywv* and *Dydv*.

*Dytwv* = *Dywv* + *Dydv* *Dywv* and *Dydv* are defined in A.3.1

If no *Dytwv* value exists for a constituent, the model uses *Dytwv* = 0 s/m$^2$·yr.

*Dytwp* = unitized yearly average total (wet and dry) deposition from particle phase over water body (s/m$^2$·yr). *Dytwp*, from the air dispersion modeling, is site-specific and stack-specific, and is the sum of *Dywp* and *Dydp*.

*Dytwp* = *Dywp* + *Dydp* *Dywp* and *Dydp* are defined in A.3.1

If no *Dytwp* value exists for a constituent, the model uses *Dytwp* = 0 s/m$^2$·yr.

$A_I$ = impervious watershed area receiving COPC deposition (m$^2$). $A_I$ is site-specific. The model uses $A_I$ = 0 m$^2$.

Table B-4-9 in the HHRAP also calculates the average runoff load to the water body from impervious surfaces in the watershed from which runoff is conveyed directly to the water body, for total mercury

[$L_{RI(Hg)}$]. $L_{RI(Hg)}$ is used in the estimation of the total load to the surface water body (see Eq. 5-28 in the HHRAP). The equation to estimate $L_{RI(Hg)}$ is:

$$L_{RI(Hg)} = 0.48 \cdot Q_{(Hg)} \cdot \left[ F_{v(Hg^{2+})} \cdot Dywwv + (1 - F_{v(Hg^{2+})}) \cdot Dytwp \right] \cdot A_I \qquad \text{(Table B-4-9 in HHRAP)}$$

where:

$L_{RI(Hg)}$ = runoff load from impervious surfaces for total mercury (g/yr). $L_{RI(Hg)}$ is constituent-specific and site-specific.

$Q_{(Hg)}$ = constituent-specific emission rate for total mercury (g/s). $Q_{(Hg)}$, obtained from calculations after the air dispersion modeling, is constituent-specific, site-specific and stack-specific.

$F_{v(Hg^{2+})}$ = fraction of mercury air concentration in vapor phase (unitless). The model uses $F_{v(Hg^{2+})}$ = 0.85 (refer to Table B-4-9 in the HHRAP) for total mercury.

*Dytwv* = unitized yearly average wet deposition from vapor phase over water body (s/m$^2$·yr). *Dytwv*, from the air dispersion modeling, is site-specific and stack-specific, and is the sum of *Dywv* and *Dydv*.

*Dytwv* = *Dywv* + *Dydv* *Dywv* and *Dydv* are defined in A.3.1

If no *Dytwv* value exists for a constituent, the model uses *Dytwv* = 0 s/m$^2$·yr.

*Dytwp* = unitized yearly average total (wet and dry) deposition from particle phase over water body (s/m$^2$·yr). *Dytwp*, from the air dispersion modeling, is site-specific and stack-specific, and is the sum of *Dywp* and *Dydp*.

*Dytwp* = *Dywp* + *Dydp* *Dywp* and *Dydp* are defined in A.3.1

If no *Dytwp* value exists for a constituent, the model uses *Dytwp* = 0 s/m$^2$·yr.

$A_I$ = impervious watershed area receiving COPC or ROPC deposition (m$^2$). $A_I$ is site-specific. The model uses $A_I$ = 0 m$^2$.

Since evapotranspiration exceeds precipitation in the Hanford Site area (e.g., there is no water to run off; the water goes back up into the air) there is insufficient water available to cause significant runoff of COPCs and ROPCs, thus surface runoff is expected to be an insignificant soil loss mechanism and an insignificant surface water loading mechanism (refer to Section 6.3). The model therefore sets $L_{RI}$ equal to zero for all constituents.

### A.4.4 Pervious Runoff Load to Water Body ($L_R$)

Equation 5-32 (Table B-4-10) in the HHRAP calculates the average runoff load to the water body from pervious soil surfaces in the watershed ($L_R$) for all COPCs. Note that the untilled soil concentration is used in this equation. $L_R$ is used in the estimation of the total load to the surface water body (see Eq. 5-28 in the HHRAP). The equation to estimate $L_R$ for COPCs (Eq. 5-32 in EPA 2005) is:

$$L_{RP} = RO \cdot \left( A_L - A_I \right) \cdot \left( \frac{Cs_2 \cdot BD}{\theta_{sw} + Kd_s \cdot BD} \right) \cdot CF \qquad \text{(Equation 5-32 in HHRAP)}$$

where:

$L_{RP}$ = runoff load from pervious surfaces (g/yr). $L_{RP}$ is constituent-specific and site-specific.

$RO$ = average annual surface runoff from pervious areas (cm/yr). $RO$ is site-specific. A value of 2.5 cm/yr is used (estimated value, assuming that the majority of rainfall recharges or evaporates).

$A_L$ = total watershed area receiving COPC deposition ($m^2$). $A_L$ is site-specific. The model uses $A_L = 3.927 \times 10^9$ $m^2$ (estimated as half of the study area).

$A_I$ = impervious watershed area receiving COPC deposition ($m^2$). $A_I$ is site-specific. The model uses $A_I = 0$ $m^2$.

$Cs_2$ = constituent concentration over the exposure duration in untilled soil (mg/kg). $Cs_2$ is constituent-specific, site-specific, and is calculated using Eqs. 5-1C, 5-1D, and 5-1E of the HHRAP.

$BD$ = soil bulk density (g soil/$cm^3$ soil). A site-specific value of 1.3 g/$cm^3$ is used (Halvorson et al. 1998).

$CF$ = units conversion factor of 0.01 (kg·$cm^2$/mg·$m^2$) for COPCs, and $1 \times 10^{-8}$ (Ci·$cm^2$/pCi·$m^2$) for ROPCs.

$\theta_{sw}$ = soil volumetric water content (mL water/$cm^3$ soil). $\theta_{sw}$ is site-specific. The EPA-recommended default value of 0.2 mL/$cm^3$ is used (Eq. 5-32 in HHRAP).

$Kd_s$ = soil-water partition coefficient (mL/g). $Kd_s$ is constituent-specific. If no $Kd_s$ value exists for an organic constituent, then $Kd_s$ is estimated using Eq. A-2-10 in the HHRAP and a $f_{oc} = 0.0044$ (fraction of organic carbon in soil, site-specific value from average organic carbon measurements [CCN 150854]), provided the constituent $K_{oc}$ value (soil organic carbon-water partition coefficient) is known. If $Kd_s$ is not available and cannot be estimated, the model uses $Kd_s = 0$ mL/g.

Since evapotranspiration exceeds precipitation in the Hanford Site area (there is no water to run off; the water goes back up into the air) there is insufficient water available to cause significant runoff of COPCs and ROPCs, thus surface runoff is expected to be an insignificant soil loss mechanism and an insignificant surface water loading mechanism (refer to Section 6.3). The model therefore sets $L_{RP}$ equal to zero for all constituents.

### A.4.5 Soil Erosion Load ($L_E$)

Equation 5-33 (Table B-4-11) in the HHRAP calculates the average load to the water body from soil erosion ($L_E$). Since one of the parameters in the equation ($ER$) is not defined for ROPCs, $L_E$ is only quantified for COPCs. Note that the untilled soil concentration is used in this equation. $L_E$ is used in the estimation of the total load to the surface water body (see Eq. 5-28 in the HHRAP). The equation to estimate $L_E$ for all COPCs (Eq. 5-33 in the HHRAP) is:

$$L_E = X_e \cdot \left(A_L - A_I\right) \cdot SD \cdot ER \cdot \left(\frac{Cs_2 \cdot Kd_s \cdot BD}{\theta_{sw} + Kd_s \cdot BD}\right) \cdot CF \qquad \text{(Eq. 5-33 in HHRAP)}$$

where:

$L_E$ = soil erosion load to the water body (g/yr). $L_E$ is constituent-specific and site-specific.

$X_e$ = unit soil loss (kg/$m^2$·yr). $X_e$ is site-specific and calculated in Eq. 5-33A in the HHRAP.

$A_L$ = total watershed area receiving COPC deposition ($m^2$). $A_L$ is site-specific. The model uses $A_L = 3.927 \times 10^9$ $m^2$ (estimated as half of the study area).

$A_I$ = impervious watershed area receiving COPC deposition ($m^2$). $A_I$ is site-specific. The model uses $A_I = 0$ $m^2$.

$SD$ = watershed sediment delivery ratio (unitless). $SD$ is site-specific and is calculated using Eq. 5-34 in the HHRAP.

$ER$ = soil enrichment ratio (unitless). $ER$ is site-specific. The following recommended values (Table B-4-11 of the HHRAP) are used: 3 for organic COPCs and 1 for inorganic COPCs and ROPCs.

$Cs_2$ = constituent concentration in untilled soil (mg/kg). $Cs_2$ is constituent-specific, site-specific, and is calculated using Eqs. 5-1C, 5-1D, and 5-1E of the HHRAP.

$Kd_s$ = soil-water partition coefficient (L/kg or mL/g). $Kd_s$ is constituent-specific. If no $Kd_s$ value exists for an organic constituent, then $Kd_s$ is estimated using Eq. A-2-10 in the HHRAP and a $f_{oc} = 0.0044$ (fraction of organic carbon in soil, site-specific value from average organic carbon measurements [CCN 150854]), provided the constituent $K_{oc}$ value (soil organic carbon-water partition coefficient) is known. If $Kd_s$ is not available and cannot be estimated, the model uses $Kd_s = 0$ mL/g.

$BD$ = soil bulk density (g soil/$cm^3$ soil). A site-specific value of 1.3 g/$cm^3$ is used (Halvorson et al. 1998).

$\theta_{sw}$ = soil volumetric water content (mL water/$cm^3$ soil). $\theta_{sw}$ is site-specific. The EPA-recommended default value of 0.2 mL/$cm^3$ is used (Eq. 5-33 in the HHRAP).

$CF$ = units conversion factor of $1 \times 10^{-3}$ (g/mg).

Since evapotranspiration exceeds precipitation in the Hanford Site area (there is no water to cause erosion; the water goes back up into the air) there is insufficient water available to cause significant erosion of COPCs and ROPCs, thus erosion is expected to be an insignificant soil loss mechanism and an insignificant surface water loading mechanism (refer to Section 6.3). The model therefore sets $L_E$ equal to zero for all constituents.

### A.4.6 Fraction of Total Water Body Concentration in the Water Column ($f_{wc}$)

Equation 5-36A (Table B-4-16) in the HHRAP calculates the fraction of total water body COPC or ROPC concentration occurring in the water column ($f_{wc}$). $f_{wc}$ is used to estimate four other parameters: the fraction of the total water body concentration in the benthic sediment (Eq. 5-36B of the HHRAP), the overall total water body dissipation rate constant (Eq. 5-38 in the HHRAP), the total water body concentration (Eq. 5-35 in the HHRAP), and the water column concentration (Eq. 5-45 in the HHRAP). The equation to estimate $f_{wc}$ for all constituents is:

$$f_{wc} = \frac{\left(1 + Kd_{sw} \cdot TSS \cdot CF\right) \cdot \frac{d_{wc}}{d_z}}{\left(1 + Kd_{sw} \cdot TSS \cdot CF\right) \cdot \frac{d_{wc}}{d_z} + \left(\theta_{bs} + Kd_{bs} \cdot C_{BS}\right) \cdot \frac{d_{bs}}{d_z}} \qquad \text{(Eq. 5-36A in HHRAP)}$$

where:

$f_{wc}$ = fraction of total water body COPC or ROPC concentration in the water column (unitless). $f_{wc}$ is constituent-specific, site-specific, and ranges from 0 to 1.

$Kd_{sw}$ = suspended sediments/surface water partition coefficient (L/kg). $Kd_{sw}$ is constituent-specific. If no $Kd_{sw}$ value exists for an organic constituent, then $Kd_{sw}$ is estimated using Eq. A-2-11 in the HHRAP and a default $f_{oc,sw}$ = 0.075 (fraction of organic carbon in suspended sediments), provided the constituent $K_{oc}$ value (soil organic carbon-water partition coefficient) is known. If $Kd_{sw}$ is not available and cannot be estimated, the model uses $Kd_{sw}$ = 0 L/kg.

$TSS$ = total suspended solids concentration (mg/L). $TSS$ is site-specific and ranges from 2 to 300 mg/L. The recommended default value of 10 mg/L is used (see Section 5.7.4.1 and Table B-4-16 of the HHRAP).

$CF$ = units conversion factor of $1 \times 10^{-6}$ (kg/mg)

$d_{wc}$ = average annual depth of water column (m). $d_{wc}$ is site-specific, and varies dramatically for the Columbia River as a result of dams on either end of the Columbia Reach. The model uses an estimated value of $d_{wc}$ = 8.65632 m (Columbia Basin Research 2000).

$d_{bs}$ = depth of upper benthic sediment layer (m). $d_{bs}$ is site-specific. The recommended default value of 0.03 m is used (Section 5.7.4 and Table B-4-16 of the HHRAP).

$d_z$ = total water body depth (m), sum of $d_{wc}$ and $d_{bs}$ (refer to definitions in Table B-4-16 of the HHRAP). $d_z$ is site-specific.

$C_{BS}$ = bed sediment concentration (g/cm$^3$). $C_{BS}$ is site-specific and ranges from 0.5 to 1.5 g/cm$^3$. The recommended default value of 1 g/cm$^3$ is used (Section 5.7.4.1 of the HHRAP).

$\theta_{bs}$ = bed sediment porosity ($L_{pore\ water}/L_{sediment}$). $\theta_{bs}$ is site-specific and ranges from 0.4 to 0.8 $L_{pore\ water}/L_{sediment}$. The recommended default value of 0.6 $L_{pore\ water}/L_{sediment}$ is used (Section 5.7.4.1 of the HHRAP).

$Kd_{bs}$ = bed sediment/sediment pore water partition coefficient (L/kg). $Kd_{bs}$ is constituent-specific. If no $Kd_{bs}$ value exists for an organic constituent, then $Kd_{bs}$ is estimated using Eq. A-2-12 in the HHRAP and a default $f_{oc,bs}$ = 0.04 (fraction of organic carbon in bottom sediments), provided the constituent $K_{oc}$ value (soil organic carbon-water partition coefficient) is known. If no $Kd_{bs}$ value exists for a constituent, and if $Kd_{bs}$ cannot be estimated, the model uses $Kd_{bs}$ = 0 L/kg.

### A.4.7 Fraction of Total Water Body Concentration in the Benthic Sediment ($f_{bs}$)

Equation 5-36B (Table B-4-16) in the HHRAP calculates the fraction of total water body COPC or ROPC concentration occurring in the benthic sediment ($f_{bs}$). $f_{bs}$ is used to estimate three other parameters: the overall total water body dissipation rate constant, $k_{wt}$ (see Table B-4-17 in the HHRAP), the benthic burial rate constant, $k_b$ (Eq. 5-43 in the HHRAP), and the bed sediment concentration, $C_{BS}$ (see Eq. 5-47 in the HHRAP). The equation to estimate $f_{bs}$ for all constituents is:

$$f_{bs} = 1 - f_{wc} \qquad \text{(Eq. 5-36B in HHRAP)}$$

where:

$f_{wc}$ = fraction of total water body COPC or ROPC concentration in the water column (unitless). $f_{wc}$ is constituent-specific, site-specific, and ranges from 0 to 1.

### A.4.8 Overall Total Water Body Dissipation Rate Constant in Surface Water ($k_{wt}$)

Equation 5-38 (Table B-4-17) in the HHRAP calculates the overall total water body COPC or ROPC dissipation rate constant in surface water ($k_{wt}$). $k_{wt}$ is used to estimate the total water body concentration (Eq. 5-35 in the HHRAP). The equation to estimate $k_{wt}$ for all constituents (Eq. 5-38 in the HHRAP) is:

$$k_{wt} = f_{wc} \cdot k_v + f_{bs} \cdot k_b \qquad \text{(Eq. 5-38 in HHRAP)}$$

where:

$k_{wt}$ = overall total water body COPC or ROPC dissipation rate constant ($yr^{-1}$). $k_{wt}$ is constituent-specific, and site-specific.

$f_{wc}$ = fraction of total water body constituent concentration in the water column (unitless). $f_{wc}$ is COPC- and ROPC-specific, site-specific, ranges from 0 to 1, and is calculated in Eq. 5-36A in the HHRAP.

$k_v$ = water column volatilization rate constant ($yr^{-1}$). $k_v$ is constituent-specific, site-specific, and calculated in Eq. 5-39 in the HHRAP.

$f_{bs}$ = fraction of total water body constituent concentration in the benthic sediment (unitless). $f_{bs}$ is constituent-specific, site-specific, ranges from 0 to 1, and is calculated in Eq. 5-36B in the HHRAP.

$k_b$ = benthic burial rate constant ($yr^{-1}$). $k_b$ is site-specific and calculated in Eq. 5-43 in the HHRAP.

### A.4.9 Water Column Volatilization Rate Constant ($k_v$)

Equation 5-39 in the HHRAP calculates the water column volatilization rate constant ($k_v$). $k_v$ is used to estimate the overall total water body dissipation rate constant, $k_{wt}$ (Eq. 5-38 in the HHRAP), which is used to estimate the total water body concentration, $C_{wtot}$ (Section 6.3 of this RAWP and Eq. 5-35 in the HHRAP). The equation to estimate $k_v$ for all constituents (Eq. 5-39 in the HHRAP) is:

$$k_v = \frac{K_v}{d_z \cdot \left(1 + Kd_{sw} \cdot TSS \cdot CF\right)} \qquad \text{(Eq. 5-39 in HHRAP)}$$

where:

$k_v$ = water column volatilization rate constant ($yr^{-1}$). $k_v$ is constituent-specific and site-specific.

$K_v$ = overall transfer rate coefficient (m/yr). $K_v$ is constituent-specific, site-specific, and is calculated in Eq. 5-40 in the HHRAP.

$d_z$ = total water body depth (m), sum of $d_{wc}$ and $d_{bs}$ (refer to definitions in Table B-4-16 of the HHRAP). $d_z$ is site-specific.

$Kd_{sw}$ = suspended sediments/surface water partition coefficient (L/kg). $Kd_{sw}$ is constituent-specific. If no $Kd_{sw}$ value exists for an organic constituent, then $Kd_{sw}$ is estimated

using Eq. A-2-11 in the HHRAP and a default $f_{oc,sw}$ = 0.075 (fraction of organic carbon in suspended sediments), provided the constituent $K_{oc}$ value (soil organic carbon-water partition coefficient) is known. If $Kd_{sw}$ is not available and cannot be estimated, the model uses $Kd_{sw}$ = 0 L/kg.

*TSS* = total suspended solids concentration (mg/L). *TSS* is site-specific and ranges from 2 to 300 mg/L. The recommended default value of 10 mg/L is used (see Section 5.7.4.1 of the HHRAP).

*CF* = units conversion factor of $1 \times 10^{-6}$ (kg/mg).

### A.4.10 Benthic Burial Rate Constant ($k_b$)

Equation 5-43 (Table B-4-22) in the HHRAP calculates the water column loss constant due to burial in benthic sediment ($k_b$). $k_b$ is used to estimate the overall total water body dissipation rate constant (see Eq. 5-38 in the HHRAP), which is used to estimate the total water body concentration (Eq. 5-35 in the HHRAP). The equation to estimate $k_b$ for all constituents is:

$$k_b = \left( \frac{X_e \cdot A_L \cdot SD \cdot CF_1 - Vf_x \cdot TSS}{A_w \cdot TSS} \right) \cdot \left( \frac{TSS \cdot CF_2}{C_{BS} \cdot d_{bs}} \right) \qquad \text{(Eq. 5-43 in HHRAP)}$$

where:

$k_b$ = benthic burial rate constant (1/yr). $k_b$ is site-specific.

$X_e$ = unit soil loss (kg/m$^2$·yr). $X_e$ is site-specific and calculated in Eq. 5-33A in the HHRAP.

$A_L$ = total watershed area receiving COPC or ROPC deposition (m$^2$). $A_L$ is site-specific. The model uses $A_L = 3.927 \times 10^9$ m$^2$ (estimated as half of the study area).

*SD* = watershed sediment delivery ratio (unitless). *SD* is site-specific and is calculated in Eq. 5-34 of the HHRAP.

$CF_1$ = units conversion factor of $1 \times 10^3$ (g/kg)

$Vf_x$ = average annual volumetric flow rate through the water body (m$^3$/yr). $Vf_x$ is site-specific. The model uses $Vf_x = 1.06 \times 10^{11}$ m$^3$/yr (PNNL 2002).

*TSS* = total suspended solids concentration (mg/L). *TSS* is site-specific and ranges from 2 to 300 mg/L. The recommended default value of 10 mg/L is used (see Section 5.7.4.1 of the HHRAP).

$A_w$ = average annual water body surface area (m$^2$). $A_w$ is site-specific, a value of $A_w$ = 3.652E+07 m$^2$ is used (PNNL 2005b).

$CF_2$ = units conversion factor of $1 \times 10^{-6}$ (kg/mg)

$C_{BS}$ = bed sediment concentration (g/cm$^3$). $C_{BS}$ is site-specific and ranges from 0.5 to 1.5 g/cm$^3$. The recommended default value of 1 g/cm$^3$ is used.

$d_{bs}$ = depth of upper benthic sediment layer (m). $d_{bs}$ is site-specific. The recommended default value of 0.03 m is used.

All default values are from Table B-4-22 in the HHRAP, unless otherwise specified.

### A.4.11 Overall Transfer Rate Coefficient ($K_v$)

Equation 5-40 (Table B-4-19) in the HHRAP calculates the overall transfer rate of contaminants from the liquid and gas-phases in surface water ($K_v$). $K_v$ is used to estimate the load to the water body due to dry vapor diffusion (Eq. 5-30 in the HHRAP), which is used to estimate the total load to the water body (Eq. 5-28 in the HHRAP). $K_v$ is also used to estimate the water column volatilization rate constant (Eq. 5-39 in the HHRAP). The equation to estimate $K_v$ for all constituents is:

$$K_v = \frac{\theta^{(T_{wk}-293)}}{\frac{1}{K_L} + \frac{R \cdot T_{wk}}{H \cdot K_G}} \qquad \text{(Eq. 5-40 in HHRAP)}$$

where:

$K_v$ = overall transfer rate coefficient (m/yr). $K_v$ is constituent-specific and site-specific.

$\theta$ = temperature correction factor (unitless). $\theta$ is site-specific. The recommended default value of 1.026 is used (Section 5.7.4.4 and Table B-4-19 of the HHRAP).

$T_{wk}$ = water body temperature (°K). $T_{wk}$ is site-specific and an average value of 285 °K is used (PNNL 2003. 2004, 2005a, 2006, 2007).

$K_L$ = liquid phase transfer coefficient (m/yr). $K_L$ is constituent-specific, site-specific, and is calculated in Eq. 5-41 of the HHRAP.

$R$ = universal gas constant (atm·m$^3$/mol·°K). A value of $R = 8.205 \times 10^{-5}$ atm·m$^3$/mol·°K is used (Section 5.7.4.4 and Table B-4-19 of the HHRAP).

$H$ = Henry's Law Constant (atm·m$^3$/mol). $H$ is constituent-specific. If no value is available for $H$, then it is estimated using Eq. A-2-3 in the HHRAP if constituent vapor pressure and water solubility data are available. If no $H$ value exists for a constituent, the model sets the overall transfer rate coefficient ($K_v$) to 0 m/yr.

$K_G$ = gas-phase transfer coefficient (m/yr). $K_G$ is site-specific and the recommended default value of 36,500 m/yr for a flowing river is used (Eq. 5-42A of the HHRAP).

### A.4.12 Equation for Calculating Unit Soil Loss ($X_e$)

Equation 5-33A (Table B-4-13) in the HHRAP calculates the soil loss rate from the watershed ($X_e$) by using the universal soil loss equation (USLE). $X_e$ is used to estimate the soil loss due benthic burial rate constant (see Eq. 5-43 in the HHRAP). The benthic burial rate constant is used to estimate loss constants that feed into the overall total water body dissipation rate constant (see Eq. 5-2A in the HHRAP), which is used to estimate the total water body concentration, including the water column and bed sediment (Eqs. 5-35 and 5-47 of the HHRAP). The equation to estimate $X_e$ for all constituents is:

$$X_e = \frac{RF \cdot K \cdot LS \cdot C \cdot PF \cdot CF_1}{CF_2} \qquad \text{(Eq. 5-33A in HHRAP)}$$

where:

$X_e$ = unit soil loss (kg/m$^2$·yr). $X_e$ is site-specific.

*RF* = USLE rainfall (or erosivity) factor ($yr^{-1}$). *RF* is site-specific and ranges from 50 to 300 $yr^{-1}$. The recommended default value of 50 $yr^{-1}$ from EPA 1998 is used.

*K* = USLE erodibility factor (ton/acre). *K* is site-specific. The recommended default value of 0.39 ton/acre is used.

*LS* = USLE length-slope factor (unitless). *LS* is site-specific. The recommended default value of 1.5 is used.

*C* = USLE cover management factor (unitless). *C* is site-specific. The recommended default value of 0.1 is used.

*PF* = USLE supporting practice factor (unitless). *PF* is site-specific. The recommended default value of 1.0 is used.

$CF_1$ = units conversion factor of 907.18 (kg/ton).

$CF_2$ = units conversion factor of 4047 ($m^2$/acre).

All default values are from Table B-4-13 in the HHRAP.

### A.4.13 Sediment Delivery Ratio (*SD*)

Equation 5-34 (Table B-4-14) in the HHRAP calculates the sediment delivery ratio (*SD*) for the watershed. *SD* is used to estimate several parameters, including the benthic burial rate constant, $k_b$ (see Eq. 5-43 in the HHRAP). Note that the benthic burial rate constant is used to estimate the overall total water body dissipation rate constant, $K_{wt}$ (see Eq. 5-38 in the HHRAP), which is used to estimate the total water body concentration (5-35 in the HHRAP), including the water column and bed sediment (Eqs. 5-35 and 5-47 of the HHRAP). The equation to estimate *SD* for all constituents is:

$$SD = a \cdot \left(A_L\right)^{-b} \qquad \text{(Eq. 5-34 in HHRAP)}$$

where:

*SD* = watershed sediment delivery ratio (unitless). *SD* is site-specific.

*a* = empirical intercept coefficient (unitless). The parameter *a* is site-specific and is determined by the watershed area as follows (EPA 2005):

| Watershed Area ($mile^2$) | *a* (unitless) |
|---|---|
| area ≤ 0.1 | 2.1 |
| 0.1 < area ≤ 1 | 1.9 |
| 1 < area ≤ 10 | 1.4 |
| 10 < area ≤ 100 | 1.2 |
| 100 < area | 0.6 |

Since the watershed area is > 100 $mile^2$, a site-specific value of $a = 0.6$ is used.

$A_L$ = total watershed area receiving COPC or ROPC deposition ($m^2$). $A_L$ is site-specific. An estimated value of $3.927 \times 10^9$ $m^2$ (estimated as half of the study area) is used.

$b$ = empirical slope coefficient (unitless). The recommended default value of 0.125 is used (Table B-4-14 in the HHRAP).

### A.4.14 Liquid Phase Transfer Coefficient ($K_L$)

Equation 5-41 (Table B-4-20) in the HHRAP calculates the rate of contaminant transfer from the liquid phase ($K_L$). The Columbia River is assumed to be a flowing river (as opposed to a quiescent lake or pond). Therefore, the equation to estimate $K_L$ for flowing streams or rivers is used. $K_L$ is used to estimate the overall transfer rate coefficient (Eq. 5-40 in the HHRAP), which is used to estimate the water column volatilization rate constant (Eq. 5-39 in the HHRAP), as well as the load to the water body due to dry vapor diffusion (Eq. 5-20 in the HHRAP), which is used to estimate the total load to the water body (Eq. 5-28 in the HHRAP). The equation to estimate $K_L$ for flowing streams or rivers for all constituents is:

$$K_L = \sqrt{\frac{CF_1 \cdot D_w \cdot u}{d_z}} \cdot CF_2 \qquad \text{(Eq. 5-41A in HHRAP)}$$

where:

$K_L$ = liquid phase transfer coefficient (m/yr). $K_L$ is constituent-specific and site-specific.

$CF_1$ = units conversion factor of $1 \times 10^{-4}$ ($m^2/cm^2$).

$D_w$ = diffusivity of COPC or ROPC in water ($cm^2/s$). $D_w$ is constituent-specific. If $D_w$ is not available, it can be estimated using Eq. A-2-5 in the HHRAP. If no value is available for $D_w$, and if it cannot be estimated, then the model uses $D_w = 0$ $cm^2/s$.

$u$ = current velocity (m/s). $u$ is site-specific. The model uses a value of $u = 1.37$ m/s, based on modeling data from Columbia Basin Research, 2000 (John Day free flow rate of 4.5 ft/sec).

$d_z$ = total water body depth (m). $d_z$ is site-specific and calculated in Table B-4-16 of the HHRAP.

$CF_2$ = units conversion factor of $3.1536 \times 10^7$ (s/yr).

### A.4.15 Gas Phase Transfer Coefficient ($K_G$)

Equation 5-42A (Table B-4-21) defines the rate of contaminant transfer from the gas phase ($K_G$) for a flowing system (as opposed to a quiescent system). Since the Columbia River is considered a flowing river as opposed to a quiescent lake or pond, parameter values for flowing streams are used for all constituents to estimate $K_G$. $K_G$ is used to estimate the overall transfer rate coefficient, $K_v$ (Eq. 5-40 in the HHRAP). Note that the overall transfer rate coefficient is used to estimate the water column volatilization rate constant (Eq. 5-39 in the HHRAP), as well as the load to the water body due to dry vapor diffusion (Eq. 5-20 in the HHRAP), which is used to estimate the total load to the water body (Eq. 5-28 in the HHRAP). The equation for $K_G$ for all constituents is:

$$K_G = 36{,}500 \text{ m/yr} \qquad \text{(Eq. 5-42A in HHRAP)}$$

where:

$K_G$ = gas-phase transfer coefficient (m/yr). $K_G$ is constant for flowing streams. The recommended default value of 36,500 m/yr for a flowing river is used.

# A.5 Mass-Limited Uptake

## A.5.1 Mass-Limited Uptake Factors for Plants

For both aboveground and belowground plants, the concentrations of contaminants in plants due to root uptake are a function of the soil concentration (*Cs*) and soil-to-plant bioaccumulation uptake factor (*Br*). Uptake factors for organic chemicals are calculated using regression equations and can result in over-estimation of plant uptake, therefore, before computing plant uptake, mass-limited uptake factors will be compared to the uptake factors as calculated per the HHRAP guidance (Sections A2-2.12.2 and A2-2.12.3), and the lesser of the two uptake factors will be used to compute plant uptake when experimental or site-specific uptake factors are not available.

From Section 6.6.3.3, the initial soil-to-plant, mass-limited uptake factor is calculated as follows:

Initial Uptake Factor = Soil Density ÷ Plant Yield

and the reasonable maximum uptake factors can be calculated as follows:

Mass-limited Uptake Factor = Initial Uptake Factor × Modifying Factor

Combining the equations above and substituting variables in the equation yields a mass-limited uptake factor equation as follows:

$$Br_{(\text{mass-limited})} = \frac{BD \cdot Z_s}{Yp} \cdot MF \qquad \text{(Equation A-20)}$$

where:

$Br_{(\text{mass-limited})}$ = final mass-limited, soil-to-plant uptake factor (kg soil/m$^2$ per kg DW plant/m$^2$).

$BD$ = soil bulk density (g soil/cm$^3$ soil). A site-specific value of 1.3 g/cm$^3$ is used (Halvorson et al. 1998) (see Table 6-3).

$Z_s$ = soil mixing zone depth (cm). Two values (depths) are used for $Z_s$: root-zone soil (15 cm) for wild produce, forage, and wild grain, and tilled soil (20 cm) for domestic produce, silage, and domestic grain (Section 6.2).

$Yp$ = yield or standing crop biomass of the edible portion of the plant for aboveground produce (productivity) (kg/m$^2$). *Yp* is site-specific and plant-type-specific. The recommended default value of 2.24 kg/m$^2$ (representing a weighted average of fruits and vegetables; HHRAP, Section 5.3.1.4) is used for produce, while a value of 0.15 kg/m$^2$ for forage (site-specific value, see Wisiol [1984]), and a value of 0.8 kg/m$^2$ (HHRAP, Section 5.4.1.4) is used for silage. A yield value of 0.25 kg/m$^2$ was assumed for above ground grains (Baes et al. 1984, Figure 4.14), and 1.17 kg/m$^2$ for belowground produce[1] (USDA 2009; Baes et. al. 1984).

[1] A yield of 600 cwt (WW)/acre (6.72 kg/m$^2$) was assumed based on USDA 2009 data for potatoes and onions. A conversion factor of 0.173 kg(DW)/kg(WW) (Baes et. al. 1984, Table 2.3, potato and onion average) is applied resulting a dry weight yield of 1.17 kg/m$^2$.

*MF* = adjustments necessary for aboveground versus belowground portions of the plant and for operating duration of the facility that is producing emissions:

- 1/80 for aboveground produce due to root uptake (1/2 × 1/40)
- 1/80 for belowground produce due to root uptake (1/2 × 1/40)
- 1 for forage (no modifying factor applied)
- 1/40 for silage (1/2 modifying factor not applied)
- 1/40 for grain (1/2 modifying factor not applied)

Substituting the appropriate values for each variable and solving for the mass-limited uptake factor yields the following:

For aboveground wild produce:

$$Br_{ag(mass-limited)} = \frac{1.3\frac{g}{cm^3}\cdot 15cm}{2.24\frac{kg}{m^2}}\cdot\frac{1}{80} = 1.09$$

For aboveground domestic produce:

$$Br_{ag(mass-limited)} = \frac{1.3\frac{g}{cm^3}\cdot 20cm}{2.24\frac{kg}{m^2}}\cdot\frac{1}{80} = 1.45$$

For aboveground forage:

$$Br_{ag(mass-limited)} = \frac{1.3\frac{g}{cm^3}\cdot 15cm}{0.15\frac{kg}{m^2}}\cdot 1 = 1300$$

For aboveground silage:

$$Br_{ag(mass-limited)} = \frac{1.3\frac{g}{cm^3}\cdot 20cm}{0.8\frac{kg}{m^2}}\cdot\frac{1}{40} = 8.13$$

For aboveground wild grain:

$$Br_{ag(mass-limited)} = \frac{1.3\frac{g}{cm^3}\cdot 15cm}{0.25\frac{kg}{m^2}}\cdot\frac{1}{40} = 19.5$$

For aboveground domestic grain:

$$Br_{ag(mass-limited)} = \frac{1.3\frac{g}{cm^3}\cdot 20cm}{0.25\frac{kg}{m^2}}\cdot\frac{1}{40} = 26.0$$

For belowground wild produce:

$$Br_{rootveg(mass-limited)} = \frac{1.3\frac{g}{cm^3}\cdot 15cm}{1.17\frac{kg}{m^2}}\cdot\frac{1}{80} = 2.08$$

For belowground domestic produce:

$$Br_{rootveg(mass-limited)} = \frac{1.3\frac{g}{cm^3}\cdot 20cm}{1.17\frac{kg}{m^2}}\cdot\frac{1}{80} = 2.78$$

### A.5.2 Mass-Limited Uptake Factors for Livestock and Game

The HHRAP recommended sources for animal uptake factors (*Ba*) for organic chemicals sometimes result in animals predicted to take up more chemical into their tissues than is present in their food, therefore, before computing plant uptake, mass-limited uptake factors will be compared to the uptake factors as calculated per the HHRAP guidance (Sections A2-2.13), and the lesser of the two uptake factors will be used to compute plant uptake when experimental or site-specific uptake factors are not available. This mass-limited uptake factor is not chemical-specific but rather it is a function of exposure duration and body weight. The feed-to-animal tissue mass-limited uptake factor as described in Section 7.1.7.4 (and Section 8.2.5.3) is calculated as follows:

Feed-to-Animal Tissue Uptake Factor = (Exposure Duration) ÷ (Tissue Weight)

Substituting variables in the equation yields a mass-limited uptake factor equation of:

$$Ba_{(mass-limited)} = \frac{ED_{animal}}{FW_{animal}} \qquad \text{(Equation A-21)}$$

where:

$Ba_{(mass\text{-}limited)}$ = mass-limited feed-to-animal tissue uptake factor (days/kg)
$ED_{animal}$ = duration to bring animal to market weight (days)
$FW_{animal}$ = total mass of animal at market weight (kg)

Substituting values from the table below yields the following:

| Animal | Value | Reference/Assumptions |
|---|---|---|
| Exposure duration, time to market, $ED_{animal}$ (days) | | |
| Beef | 1260 | USDA (1996). The approximate maximum age limitation for the Prime, Choice, and Standard grades of steers, heifers, and cows is 42 months. |
| Pork | 180 | Oklahoma State University (2007). In outdoor lot systems of swine production, hogs should reach market weight (240 lb.) in 180 days or less. |
| Poultry | 150 | 9 CFR 381.170(a)(1)(iv). *Roaster or roasting chicken*. A bird of this class is a young chicken (usually 3 to 5 months of age). |
| Mass of Animal, $FW_{animal}$ (kg) | | |
| Beef | 515 | USDA (2011), other cattle, live weight of 1137 lbs |
| Pork | 114 | USDA (2011), other hogs, live weight of 250 lbs |
| Poultry | 1.50 | USDA (2009), pounds sold=1,267,000 lbs, sold for slaughter=384,000 (2008 data) |

For beef:
$$Ba_{beef(mass-limited)} = \frac{1260\ \text{days}}{515\ \text{kg FW}} = 2.45\frac{\text{days}}{\text{kg FW}}$$

For pork:
$$Ba_{pork(mass-limited)} = \frac{180\ \text{days}}{114\ \text{kg FW}} = 2.58\frac{\text{days}}{\text{kg FW}}$$

For poultry (domestic and wild): $$Ba_{poultry(\,mass-limited\,)} = \frac{150\ \text{days}}{1.5\ \text{kg FW}} = 100\frac{\text{days}}{\text{kg FW}}$$

The equation above is used to estimate mass-limited feed-to-animal tissue uptake factors for beef, pork, and poultry. Estimating a mass-limited feed-to-animal uptake factor for animal products (that is, milk and eggs) is slightly different. The mass limited feed-to-animal product uptake factor is a function of the daily product weight for the animal. The equation for the mass-limited feed-to-animal product uptake factor is:

Feed-to-Animal Product Uptake Factor = 1 ÷ (Daily Product Weight)

Substituting variables in the equation yields a mass-limited uptake factor equation of:

$$Ba_{(\text{mass-limited})} = \frac{1}{FW_{product}}$$

where:

$Ba_{(\text{mass-limited})}$ = mass-limited feed-to-animal product uptake factor (days/kg)
$FW_{product}$ = total expected weight of animal product each day (kg/day)

Substituting values from the table below yields the following:

| **Food Product** | **Value** (kg FW/day) | **Reference/Assumptions** |
|---|---|---|
| Milk | 29.0 | USDA (2009), annual milk production=23,344 lbs |
| Eggs | 0.0426 | USDA (2009), annual eggs produced=1533 million,<br>average layers producing=5,584,000 (2008 data),<br>USDA (2000), egg weight=2.0 oz. (Grade A egg, Table I of §56.218) |

For milk: $$Ba_{milk(\,mass-limited\,)} = \frac{1}{29.0\ \frac{\text{kg FW}}{\text{day}}} = 0.0345\frac{\text{days}}{\text{kg FW}}$$

For eggs (domestic and wild): $$Ba_{eggs(\,mass-limited\,)} = \frac{1}{0.0426\ \frac{\text{kg FW}}{\text{day}}} = 23.5\frac{\text{days}}{\text{kg FW}}$$

# A.6 Derivation of Selected Site-Specific Parameters

Where available, site-specific data is used as input for risk modeling. In some cases, where data for multiple years or conditions is available, arithmetic averages are used. The site-specific inputs for the Columbia River water temperature and flow, Hanford Site annual precipitation, humidity, ambient air temperature, and wind speed fall into this category. These data are available annually as published in the *Hanford Site Environmental Report* (PNNL 2003. 2004, 2005a, 2006, 2007). The *Hanford Site*

*Environmental Report* is prepared annually and provides an overview of activities at the site, and summarizes environmental data that characterize the Hanford Site. The table below presents the data from reports corresponding to the period for which air modeling was done (2002 - 2006), and provides the average values that are used as site-specific inputs (PNNL 2003. 2004, 2005a, 2006, 2007).

| Year | 2002 | 2003 | 2004 | 2005 | 2006 | Average Value |
|---|---|---|---|---|---|---|
| River Temperature (deg F) | 53.6 | 55.4 | 53.6 | 51.8 | 51.8 | 53.2 |
| River Flowrate ($m^3$/yr) (Priest Rapids) | 1.05E+11 | 9.04E+10 | 8.92E+10 | 9.38E+10 | 1.05E+11 | 9.68E+10 |
| Precipitation (inches) | 5.41 | 8.14 | 7.96 | 6.39 | 8.46 | 7.27 |
| Relative Humidity (percent) | 53.6 | 53.7 | 57.9 | 55.2 | 55.3 | 55.1 |
| Ambient Air Temperature (deg F) | 54.4 | 55.6 | 54.6 | 53.5 | 54.1 | 54.4 |
| Wind Speed (m/s) | 3.5 | 3.5 | 3.1 | 3.2 | 3.5 | 3.4 |

# A.7 Derivation of Particulate Emission Factor

The particulate emission factor (*PEF*) represents an annual average emission rate based on wind erosion that should be compared with chronic health criteria. The *PEF* equation is based on the "unlimited reservoir" model developed to estimate particulate emissions due to wind erosion (Cowherd et al. 1985). The *PEF* is computed according to the following equation (EPA 2000):

$$PEF = Q/C \cdot \frac{3600}{0.036 \cdot (1-V) \cdot \left(\frac{u_m}{u_t}\right)^3 \cdot \mathrm{F}(x)} \qquad \text{(Eq. 4 of EPA 2000)}$$

where:

| | | |
|---|---|---|
| *PEF* | = | particulate emission factor ($m^3$/kg) |
| *Q/C* | = | inverse of mean concentration at location of exposure ($g/m^2 \cdot s$ per $kg/m^3$) |
| *V* | = | fraction of vegetative cover (unitless) |
| $u_m$ | = | mean annual windspeed (m/s) |
| $u_t$ | = | critical wind speed at 7-m height (m/s) |
| F(*x*) | = | integration function dependent on $u_m/u_t$ derived using Cowherd et al. (1985) (unitless) |

define the steps for determining potential respirable particulate emission from wind erosion. The soil particle size distribution, apparent roughness of the site, vegetation cover, presence of a crust on the soil, and presence of non-erodible elements (e.g., large stones) are used to define the potential for suspension.

The potential for wind erosion is quantified in terms of a threshold friction velocity. The greater the value of the threshold friction velocity for a site, the lower the potential for particle suspension. The threshold friction velocity for the contaminated area is determined by knowing the mode of the aggregate particulate size distribution (which is derived from the soil composition) and using a formula derived from the graphical relationship given in equation 5-39 in Streile et al. (1996):

$$u_t^* = \frac{N \cdot e^{(0.412 \cdot \ln(X) + 4.17)}}{100}$$ (Equation 5-39 in Streile et al. [1996])

where:

$u^*_t$ = threshold friction velocity (m/s)
$X$ = aggregate size distribution (mm) = (0.0106) × (Percent Sand) + 0.05
$N$ = nonerodible elements correction factor (dimensionless)

The aggregate size distribution is estimated using $X$ = (0.0106) × (Percent Sand) + 0.05. From the viewpoint of increasing the potential for suspension, this relationship provides relatively realistic estimates for soils with greater than 75 % sand content. For other soils, the relationship provides relatively conservative estimates that are more typical of disturbed soils than undisturbed soils (Streile et al. 1996). The *Hanford Site National Environmental Policy Act (NEPA) Characterization* (Neitzel et al. 2005) describes 15 different surface soil types on the Hanford Site, varying from sand to silty and sandy loam. Burbank loamy sand, which characterizes much of the soil in the region of the Hanford Central plateau where the WTP is located, has a subsoil gravel content of 20 % to 80 %. Assuming the remaining subsoil is sand, a corresponding aggregate distribution ($X$) of 262 µm to 898 µm is computed. From Figure 3-4 of Cowherd et al. (1985), the corresponding uncorrected threshold friction velocity ($u^*_t$) ranges from 37.9 to 63.0.

The uncorrected threshold friction velocity ($u^*_t$) must be adjusted for the effects of any non-erodible elements in the contaminated area. This correction for the fraction of surface coverage is given by Eq. 4-3 in Cowherd et al. (1985). Once the threshold friction velocity has been determined, the critical wind speed at a given height above the surface can be determined using the following equation:

$$u_t = \frac{1}{k} \cdot u_t^* \cdot \ln\left(\frac{z}{z_0}\right)$$ (modified Eq. 4-3 in Cowherd et al. [1985])

where:

$u_t$ = corrected threshold values, or, critical wind speed at 7-m height (m/s)
$k$ = von Karman constant (0.4; dimensionless)
$z$ = reference height above the surface (7 m)
$z_0$ = surface roughness length (m)

The value of $z$ recommended by Cowherd et al. (1985) is 7 m. The surface roughness length of the site, $z_o$, is related to the size and spacing of the roughness elements in the area. Figure 2.1 in Cowherd et al. (1985) illustrates $z_o$ for various surfaces. For the land use scenarios of the risk assessment, $z_0$ will range from 1 (subsistence farming) to 4 (undisturbed grass steppe). Conservatively, a value of $z_0$ = 1 is used. The corresponding range of critical wind speed at 7 m height is from 6.21 m/s to 10.3 m/s.

The vertical flux of particles smaller than 10 µm in diameter is assumed to be proportional to the cube of the horizontal wind speed (Cowherd et al. 1985). The integration function, F($x$), comes from the cubic relationship of the vertical transport of particles and the wind speed. The F($x$) function is derived using the following equations from Cowherd et al. (1985):

$$x = \frac{\sqrt{\pi}}{2} \cdot \frac{u_t}{u_m}$$ (Appendix B in Cowherd et al. [1985])

$$\mathrm{F}(x) = 0.18 \cdot \left(8 \cdot x^3 + 12 \cdot x\right) \cdot e^{-x^2}$$ (Appendix B in Cowherd et al. [1985])

where:

$x$ = ratio of mean annual windspeed and critical wind speed at 7-m height (unitless)
F($x$) = integration function dependent on $u_m/u_t$ derived using Cowherd et al. (1985) (unitless)
$u_m$ = mean annual windspeed (m/s)
$u_t$ = critical wind speed at 7-m height (m/s)

The mean wind speed for the Hanford Site is available in *Hanford Site Environmental Report* (PNNL 2003, 2004, 2005a, 2006, 2007) (see Section A.6). The average windspeed for the period of 2002 through 2006 is 3.4 m/s.

With a critical windspeed in the range of 6.21 m/s to 10.3 m/s, and an average windspeed of 3.4 m/s, the value of $x$ ranges from 1.62 to 2.72, with a corresponding function (F($x$)) of to 0.674 to 0.0212.

The EPA default fraction of vegetative cover assumes 50 % vegetative cover and 50 % open soil. The Hanford Site is located in climatic zone 4 (Figure A-1, EPA 2000), so a value of 40.4 is used to describe the inverse mean concentration at center of a 30-acre-square source (average value of cities in climatic zone 4).

Using the parameter values above, the following range of *PEF* values applies to the Hanford Site:

| **Variable** | **Description** | **Value associated with 20 % sand** | **Value associated with 80 % sand** | **Reference** |
|---|---|---|---|---|
| $P_{sand}$ | percent sand (percent) | 20 | 80 | Neitzel et al. (2005). (Section 4.3.3. "Surface Soil" Burbank Loamy Sand) |
| $V$ | fraction of vegetative cover (unitless) | 0.5 | 0.5 | EPA (2000). (Eq. 3 default values) |
| $u_m$ | mean annual windspeed (m/s) | 3.4 | 3.4 | Site-specific (see Section A.6) |
| $Q/C$ | inverse of mean concentration at location of exposure (g/m$^2$·s per kg/m$^3$) | 40.4 | 40.4 | EPA (2000). (Exhibit 10 . Q/C Values by Source Area, City, and Climatic Zone, Seattle, 0.5 acre) |
| $N$ | nonerodible elements correction factor (unitless) | 101.8 | 101.8 | Gillette et al. (1980) |
| $X$ | aggregate size distribution (mm) | 0.262 | 0.898 | Streile et al. (1996) |
| $u^*_t$ | threshold friction velocity (m/s) | 37.9 | 63.0 | |

| Variable | Description | Value associated with 20 % sand | Value associated with 80 % sand | Reference |
|---|---|---|---|---|
| $r$ | von Karman constant (unitless) | 0.4 | 0.4 | Cowherd et al. (1985). (Figure 3-6 for plowed field, Eq. 4-3) |
| $z$ | reference height above the surface (m) | 7 | 7 | |
| $z_0$ | surface roughness length (m) | 1 | 1 | |
| $u_t$ | critical wind speed at 7-m height (m/s) | 6.21 | 10.3 | |
| $x$ | ratio of mean annual windspeed and critical wind speed at 7-m height (unitless) | 1.62 | 2.72 | Resulting computed values |
| F($x$) | integration function (unitless) | 0.674 | 0.0212 | |
| *PEF* | particulate emission factor ($m^3$/kg) | $7.58 \times 10^7$ | $1.10 \times 10^{10}$ | |

The actual *PEF* for the Hanford Site varies with the location, and most likely falls within the range predicted above ($7.58 \times 10^7$ $m^3$/kg to $1.10 \times 10^{10}$ $m^3$/kg). To estimate the exposure point concentration of resuspended dust particles, the soil concentration is divided by the *PEF* (refer to modified HHRAP Table C-2-1 and modified Eq. 3 of EPA (2000) as described in Section 7.1.5.2). Therefore, a lower *PEF* will yield a higher (or more conservative) estimate of the exposure point concentration of resuspended dust particles. The *PEF* of $7.58 \times 10^7$ $m^3$/kg will be used in the initial risk assessment for the WTP. If the initial assessment of risks indicates that inhalation of resuspended soil is a critical pathway (i.e., the pathway contributes an unacceptable amount of risk), then additional and more accurate site-specific information will be sought and a more accurate *PEF* will be determined for use in the final risk assessment.

# A.8 Derivation of Alternate American Indian Scenario Consumption Rates

## A.8.1 Alternate American Indian Scenario #1

The lifestyle and exposure parameters of the first alternate subsistence American Indian resident are primarily based on data from *Exposure Scenario for CTUIR Traditional Subsistence Lifeways* (Harris and Harper 2004) and *Application of the CTUIR Traditional Lifeways Exposure Scenario in Hanford Risk Assessments* (Harris 2008). Other parameters were taken from the "A Native American Exposure Scenario" (Harris and Harper 1997) or from EPA's *Exposure Factors Handbook* (EFH, EPA 1997). Children's exposure parameters were developed by proportioning the child caloric intake reported in the *Child-Specific Exposure Factors Handbook* (CSEFH, EPA 2008) according to the various proportions of meat, vegetable, roots, etc. in the diet of the adult American Indian member as reported in the guidance documents provided by the Confederated Tribes of the Umatilla Indian Reservation (CTUIR). The derivation of food consumption rates is shown in the following tables. Data from Figure 1 of Harris (2008) was used to derive child consumption rates by applying the adult diet caloric intake (as percent of calories for each food category) to a child caloric consumption rate of 1466 kcal/day. The child caloric

consumption rate of 1466 kcal/day is based on the average food-energy intake for children ages 3 to 5 shown in Table 6-35 of the CSEFH. Consumption rates are converted to units appropriate to RAWP equations by dividing the daily intake by the receptor weight[2].

**Consumption rates prorated for children**

| | Data for Adult Consumption (Harris 2008) | | | | Diet prorated for children | |
|---|---|---|---|---|---|---|
| **Food Category** | **g/day** | **kcal/100g** | **kcal/day** | **Percent of calories** | **g/day** | **kcal/day** |
| **Aboveground Produce** | | | | | | |
| Berries, Fruits | 125 | 100 | 125 | 6 % | 88 | 88 |
| Other vegetation (lichen, pith, cambium) | 40 | 100 | 40 | 2 % | 29 | 29 |
| Greens, Tea, Medicines, Spices | 133 | 30 | 40 | 2 % | 98 | 29 |
| Honey, Sweeteners | 15 | 275 | 41 | 2 % | 11 | 29 |
| Seeds, Nuts, Grain | 24 | 500 | 120 | 5 % | 15 | 73 |
| **Belowground Produce** | | | | | | |
| Bulbs (onions, other) | 40 | 30 | 12 | 1 % | 49 | 15 |
| Roots, Tubers | 400 | 100 | 400 | 18 % | 264 | 264 |
| **Meats** | | | | | | |
| Fish | 620 | 175 | 1085 | 49 % | 410 | 718 |
| Game, large & small | 125 | 175 | 219 | 10 % | 84 | 147 |
| Fowl & Eggs | 62 | 200 | 124 | 6 % | 44 | 88 |
| **Totals** | 1584 | 1685 | 2205.9 | 100 % | 1091 | 1466 |

See Figure 1 of Harris (2008).

Aboveground produce consumption rates were derived by summing applicable consumption rate data for selected food types as shown below.

**Consumption rate for aboveground produce ($CR_{ag}$)**

| | Adult | | Child | |
|---|---|---|---|---|
| **Food Category** | **g/day** | **kg/kg·day** | **g/day** | **kg/kg·day** |
| Berries, Fruits | 125 | 0.0018 | 88 | 0.0059 |
| Other vegetation (lichen, pith, cambium) | 40 | 0.00057 | 29 | 0.0020 |
| Greens, Tea, Medicines, Spices | 133 | 0.0019 | 98 | 0.0065 |
| Honey, Sweeteners | 15 | 0.00021 | 11 | 0.00071 |
| Seeds, Nuts, Grain | 24 | 0.00034 | 15 | 0.00098 |
| **Totals ($CR_{ag}$)** | **337** | **0.0048** | **240** | **0.016** |

[2] Conversion of g/day to kg/kg·day (divide by receptor body weight and convert g to kg):
Adult: kg/kg·day = (g/day) / 70 kg / (1000 g/kg)
Child: kg/kg·day = (g/day) / 15 kg / (1000 g/kg)

Belowground produce consumption rates were derived by summing applicable consumption rate data for selected food types as shown below.

**Consumption rate for belowground produce ($CR_{bg}$)**

| | Adult | | Child | |
|---|---|---|---|---|
| **Food Category** | **g/day** | **kg/kg·day** | **g/day** | **kg/kg·day** |
| Bulbs (onions, other) | 40 | 0.00057 | 49 | 0.0033 |
| Roots, Tubers | 400 | 0.0057 | 264 | 0.018 |
| **Totals ($CR_{bg}$)** | **440** | **0.0063** | **313** | **0.021** |

Fish, game and fowl consumption rates are summarized below. Per Harris (2008), organ consumption is assumed to account for 10 % of the caloric intake for fish and game.

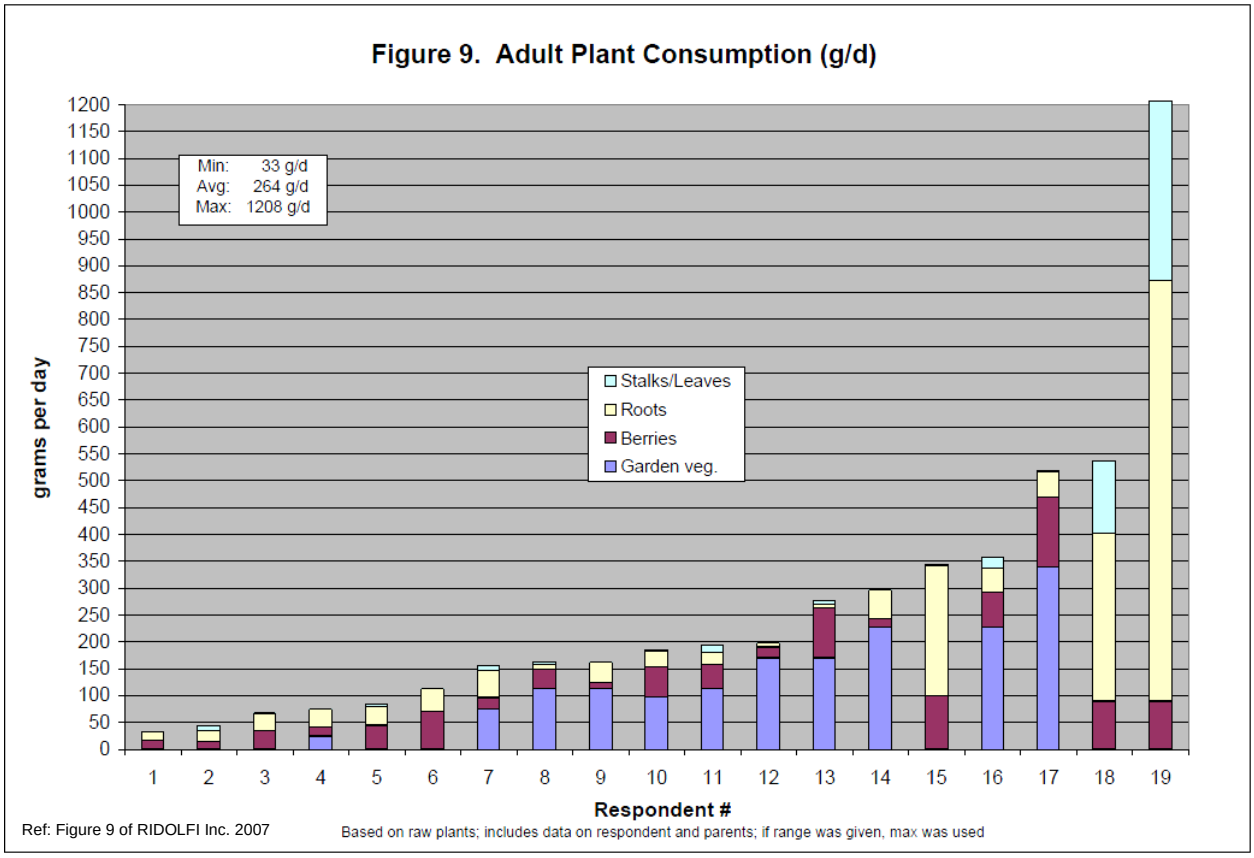
**Consumption rates for fish, game and fowl**

| | Adult | | Child | |
|---|---|---|---|---|
| **Food Category** | **g/day** | **kg/kg·day** | **g/day** | **kg/kg·day** |
| Fish ($CR_{fish}$) | 558 | 0.0080 | 369 | 0.025 |
| Fish ($CR_{fish\ organs}$) | 62 | 0.00089 | 41 | 0.0027 |
| Game, large & small ($CR_{game}$) | 112 | 0.0016 | 75 | 0.0050 |
| Game organs ($CR_{game\ organs}$) | 13 | 0.00018 | 8.4 | 0.00056 |
| Fowl & Eggs ($CR_{fowl}$) | 62 | 0.00089 | 44 | 0.0029 |

## A.8.2 Alternate American Indian Scenario #2

The lifestyle and exposure parameters of the second alternate subsistence American Indian resident are primarily based on data from *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment* (RIDOLFI Inc. 2007). Other parameters were taken from the EFH. Children's exposure parameters were developed by proportioning the child caloric intake reported in the CSEFH according to the various proportions of meat, vegetable, roots, etc. in the diet of the adult American Indian member as reported in the guidance documents provided by the Yakama Nation. The derivation of food consumption rates is shown in the following tables. Data from Figure 9 of RIDOLFI Inc. (2007) was used to derive consumption rates by extracting data in Figure 9 and computing the averages and relative dietary proportions of domestic and wild food stuffs. The bar graphs of Figure 9 were manually inspected and used to determine the grams per day of domestic (garden) and wild (other) produce consumed by survey respondents. These values were then used to compute domestic and wild produce consumption as percent of the respondent's diet. An average dietary distribution of domestic, wild aboveground, and wild belowground produce of 36 %, 34 %, and 31 % (respectively) was then computed. Consumption rates are converted to units appropriate to RAWP equations by dividing the daily intake by the receptor weight[3].

[3] Conversion of g/day to kg/kg·day (divide by receptor body weight and convert g to kg):
Adult: kg/kg·day = (g/day) / 70 kg / (1000 g/kg)
Child: kg/kg·day = (g/day) / 16 kg / (1000 g/kg) (note: per RIDOLFI Inc. 2007, a child weight of 16 kg is assumed)




**Assessment (manual tally) of Figure 9 data (RIDOLFI Inc. 2007)**

| Respondent | Total produce (g/day) | From domestic produce (g/day) | Percent of diet that is domestic produce | From wild aboveground produce (stalks, leaves, berries) (g/day) | Percent of diet that is wild aboveground produce | From wild belowground produce (roots) (g/day) | Percent of diet that is wild belowground produce |
|---|---|---|---|---|---|---|---|
| 1 | 33 | 0 | 0 % | 20 | 61 % | 13 | 39 % |
| 2 | 45 | 0 | 0 % | 25 | 56 % | 20 | 44 % |
| 3 | 70 | 0 | 0 % | 40 | 57 % | 30 | 43 % |
| 4 | 75 | 25 | 33 % | 20 | 27 % | 30 | 40 % |
| 5 | 110 | 0 | 0 % | 50 | 45 % | 60 | 55 % |
| 6 | 110 | 0 | 0 % | 65 | 59 % | 45 | 41 % |
| 7 | 155 | 75 | 48 % | 35 | 23 % | 45 | 29 % |
| 8 | 160 | 110 | 69 % | 45 | 28 % | 5 | 3 % |
| 9 | 160 | 110 | 69 % | 15 | 9 % | 35 | 22 % |
| 10 | 180 | 95 | 53 % | 65 | 36 % | 20 | 11 % |
| 11 | 190 | 110 | 58 % | 60 | 32 % | 20 | 11 % |
| 12 | 200 | 165 | 83 % | 25 | 13 % | 10 | 5 % |
| 13 | 275 | 165 | 60 % | 105 | 38 % | 5 | 2 % |
| 14 | 300 | 225 | 75 % | 20 | 7 % | 55 | 18 % |
| 15 | 345 | 0 | 0 % | 100 | 29 % | 245 | 71 % |
| 16 | 360 | 225 | 63 % | 85 | 24 % | 50 | 14 % |

**Assessment (manual tally) of Figure 9 data (RIDOLFI Inc. 2007)**

| Respondent | Total produce (g/day) | From domestic produce (g/day) | Percent of diet that is domestic produce | From wild aboveground produce (stalks, leaves, berries) (g/day) | Percent of diet that is wild aboveground produce | From wild belowground produce (roots) (g/day) | Percent of diet that is wild belowground produce |
|---|---|---|---|---|---|---|---|
| 17 | 520 | 340 | 65 % | 125 | 24 % | 55 | 11 % |
| 18 | 540 | 0 | 0 % | 230 | 43 % | 310 | 57 % |
| 19 | 1208 | 0 | 0 % | 428 | 35 % | 780 | 65 % |
| **Average** | **264** | **87** | **36 %** | **82** | **34 %** | **96** | **31 %** |

The dietary distribution of domestic, wild aboveground, and wild belowground produce was then applied to the proposed vegetable and fruit consumption rates reported in Table 6 of RIDOLFI Inc. (2007). A diet distribution of 36 % domestic produce, 34 % wild aboveground produce, and 31 % belowground produce was applied to an adult intake of 1417 g/d, and a child intake of 314 g/day.

**Consumption rates for produce**

| | Adult Consumption | | Child Consumption | |
|---|---|---|---|---|
| **Food Category** | **g/day** | **kg/kg·day** | **g/day** | **kg/kg·day** |
| Vegetables | 1118 | 0.016 | 187 | 0.012 |
| Fruit | 299 | 0.0043 | 127 | 0.0079 |
| Total Produce | 1417 | 0.020 | 314 | 0.020 |
| | | | | |
| Domestic Produce ($CR_{aq}$) (36 % of diet) | 504 | 0.0072 | 112 | 0.0070 |
| Wild Aboveground Produce ($CR_{aq\ wild}$) (34 % of diet) | 481 | 0.0069 | 106 | 0.0067 |
| Wild Belowground Produce ($CR_{bq}$) (31 % of diet) | 433 | 0.0062 | 96 | 0.0060 |

Assumptions regarding the receptor's dietary distribution of domestic livestock and wild game animals was based on a compendium of available sources. RIDOLFI Inc. (2007) states that approximately 60 % of meat consumed is domestic (Section 3.2.2, p. 20). Data from Harris (2008) indicates poultry is 33 % of the game & fowl diet for adults (Figure 1 shows 62 g/day for fowl and eggs, verses 125 g/day for game animals). Thus, for this American Indian scenario, it is assumed that of the meats consumed, 60 % is domestic livestock, and 40 % is game. Furthermore, it is assumed within those categories, approximately 1/3 is poultry and fowl, while the remaining 2/3 is other livestock (beef) and game (deer). From Table 7 or RIDOLFI Inc. (2007), it is assumed that the daily adult consumption of meat products is 704 g/day, and the child's consumption is 212 g/day. This is summarized in the table below.

**Consumption rates for meat products**

| | Adult Consumption | | Child Consumption | |
|---|---|---|---|---|
| **Food Category** | **g/day** | **kg/kg·day** | **g/day** | **kg/kg·day** |
| Total meat products | 704 | 0.010 | 212 | 0.013 |
| Domestic poultry (60 %×33 %) | 139 | 0.0020 | 42 | 0.0026 |
| Domestic livestock (beef) (60 %×67 %) | 283 | 0.0040 | 85 | 0.0053 |
| Wild fowl (40 %×33 %) | 93 | 0.0013 | 28 | 0.0017 |
| Wild game (deer) (40 %×67 %) | 189 | 0.0027 | 57 | 0.0036 |

Per RIDOLFI Inc. (2007), the adult American Indian receptor consumes 1.2 L/day of milk, and the child consumes 0.5 L/day of milk presumable from local dairy cows. This receptor also consumes 519 g/day

and 363 g/day of fish (adult and child consumption, respectively). These consumption rates are summarized below.

**Other consumption rates**

| | Adult Consumption | | Child Consumption | |
|---|---|---|---|---|
| **Food Category** | **L/day or g/day** | **kg/kg·day** | **L/day or g/day** | **kg/kg·day** |
| Milk | 1.2 | 0.017 | 0.5 | 0.031 |
| Fish | 519 | 0.0074 | 363 | 0.023 |

No organ consumption is reported in RIDOLFI Inc. (2007) so no organ consumption is assumed for this receptor.

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**Table A-1 Soil EPC Equations**

| Timeframe | Noncarcinogen Instantaneous Soil Concentration [a] | | Carcinogen Average Soil Concentration | |
|---|---|---|---|---|
| | **Without Soil Loss** ks = 0 | **With Soil Loss** ks > 0 | **Without Soil Loss** ks = 0 | **With Soil Loss** ks > 0 |
| **Current** $0 \le T_1 < T_2 \le tD$ | $Ds \cdot tD$ | $Cs_{tD}$ | $\frac{Ds}{2} \cdot (T_2 + T_1)$ | $\frac{Ds}{ks \cdot (T_2 - T_1)} \cdot \left[ \left( T_2 + \frac{e^{-ks \cdot T_2}}{ks} \right) - \left( T_1 + \frac{e^{-ks \cdot T_1}}{ks} \right) \right]$ |
| **Future** $0 < tD \le T_1 < T_2$ | $Ds \cdot tD$ | $Cs_{tD}$ | $\frac{Ds}{2} \cdot (T_2 + tD)$ | $\frac{Cs_{tD}}{ks \cdot (T_2 - T_1)} \cdot \left( e^{-ks \cdot (T_1 - tD)} - e^{-ks \cdot (T_2 - tD)} \right)$ |
| **Spanning Current to Future** $0 \le T_1 < tD < T_2$ | $Ds \cdot tD$ | $Cs_{tD}$ | $\frac{Ds}{2 \cdot (T_2 - T_1)} \cdot \left( T_2 \cdot tD - T_1^2 \right)$ | $\frac{Ds}{ks \cdot (T_2 - T_1)} \cdot \left[ tD - T_1 + \frac{e^{-ks \cdot tD} - e^{-ks \cdot T_1}}{ks} + \frac{Cs_{tD}}{Ds} \cdot \left( 1 - e^{-ks \cdot (T_2 - tD)} \right) \right]$ |

Note: $Cs_{tD} = \frac{Ds \cdot \left(1 - e^{-ks \cdot tD}\right)}{ks}$

$Cs_{tD}$ = instantaneous soil concentration at time $tD$ (mg COPC/kg soil).

$Ds$ = deposition term to soil (mg/kg·yr). $Ds$ is constituent-specific, site-specific, and depth-specific.

$ks$ = overall soil loss constant due to all processes ($yr^{-1}$).

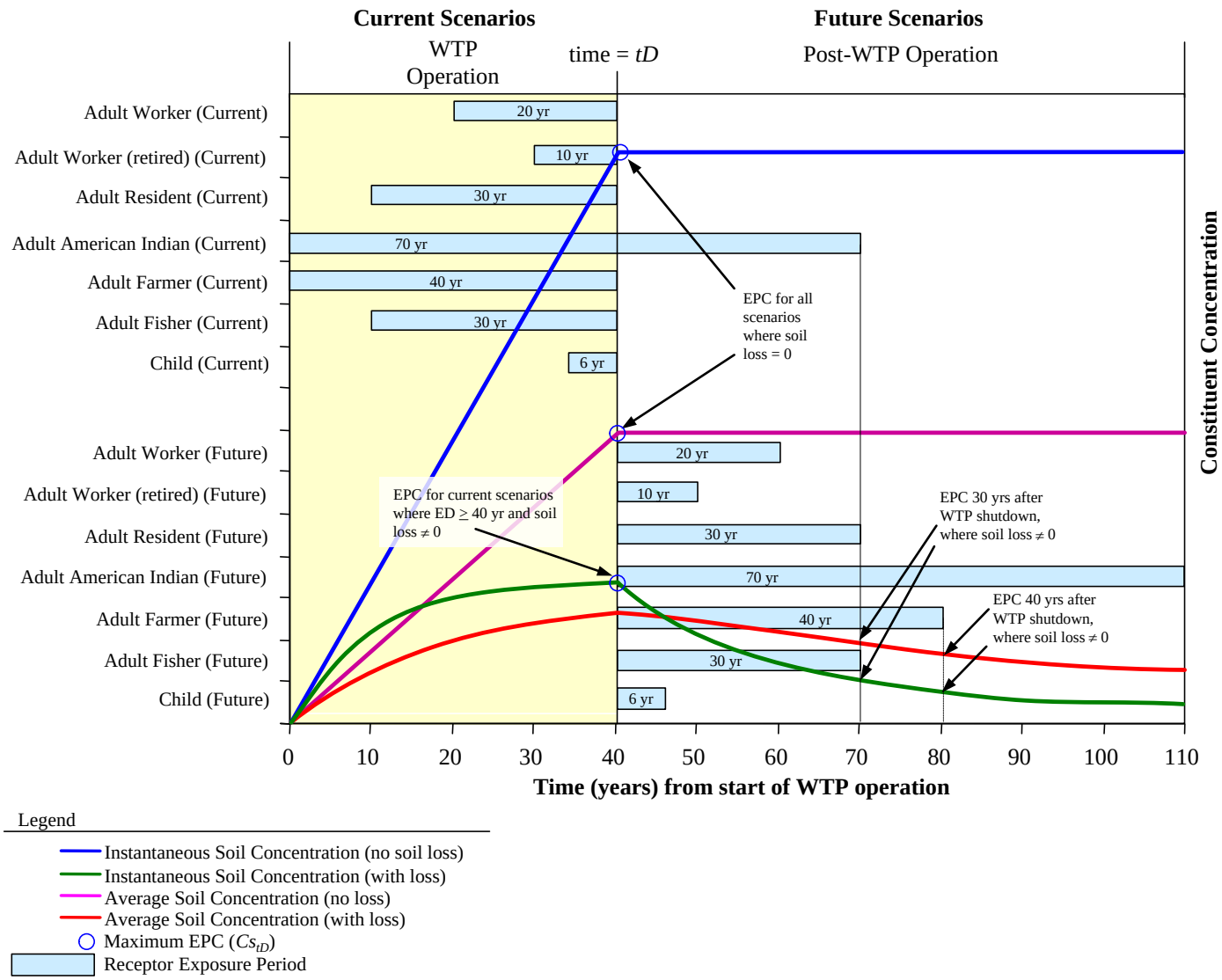
$T_1$ = the time at the start of exposure (yr).

$T_2$ = the time at the end of exposure (yr).

$tD$ = the time at the end of emissions/deposition (yr).

a Since noncarcinogenic risk is based on a threshold value (the reference dose), HHRAP (Section 5.2.1) recommends that the maximum instantaneous concentration should be used for risk assessment.

**Figure A-1 Receptor Exposure and Instantaneous and Average Soil Concentrations**



# Appendix B

# Radiological Risk Assessment Issues

# Appendix B
# Radiological Risk Assessment Issues

## Contents

**Tetra Tech EM Inc. Letter** ..... **B-1**

**Attachment to Tetra Tech EM Inc. Letter** ..... **B-3**

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1 Tetra Tech EM Inc. Letter extracted from record copy CCN 019247.

# Tetra Tech EM Inc. Letter

**Tetra Tech EM Inc.**

One Dallas Centre ◆ 350 N. St. Paul St. ◆ Dallas, TX 75201 ◆ (214) 754-8765 ◆ FAX (214) 922-9715

March 27, 2001

Mr. Jerry Yokel, Project Officer
Department of Ecology
Nuclear Waste Program
1315 W. 4th Street
Kennewick, Washington 99336-6018

Subject: Contract C0000084
Hanford River Protection Privatization Project
Screening Level Risk Assessment Work Plan
Radiological Risk Assessment Issues

Dear Mr. Yokel:

On November 2, 2000, the Washington State Department of Ecology (Ecology) and the U.S. Department of Energy (DOE) met to discuss the screening level risk assessment work plan for DOE's Hanford River Protection Privatization Project . During this meeting, Ecology asked Tetra Tech EM Inc., to evaluate several outstanding radiological risk assessment issues, as follows:

Issue 1: Prepare a brief report addressing the potential volatility of the radionuclides listed in RAWP-72 with respect to their becoming airborne in a sweat lodge (i.e., water containing these radionuclides splashed onto hot rocks to make steam).

Issue 2: Briefly review the list of 46 radionuclides and identify any other radionuclides, in addition to those listed in RAWP-72, that may become airborne and represent a potential inhalation exposure pathway in a sweat lodge.

Issue 3: Check if the HEAST slope factor for inhalation of tritiated water vapor includes uptake by dermal absorption.

Issue 4: Check to determine if dermal absorption of I-129 can be a significant contributor to risk relative to inhalation.

Issue 5: Check to determine if I-129 can represent an external exposure risk from plume immersion which may be significant relative to the risk it represents by inhalation.

Issue 6: Prepare a brief statement defining the level of exposure that may be considered a LOAEL for radionuclides (e.g., 1 to 5 rem).

Issue 7: Provide a brief report on the concentrations of naturally occurring and ubiquitous manmade radionuclides in mother's breast milk.

Mr. Jerry Yokel
Washington State Department of Ecology
March 27, 2001
Page 2

The Attachment presents responses to each issue. The responses were prepared by Dr. John Mauro of Sandy Cohen & Associates. If you have any questions, please me at (214) 740-2022.

Sincerely,

William P. Desmond, Ph.D.
Senior Environmental Scientist

cc: J. Pankanin, Tetra Tech EM Inc.
J. Mauro, Sandy Cohen & Assoc.
file

# Attachment to Tetra Tech EM Inc. Letter

**ATTACHMENT**

This attachment discusses several issues raised during a meeting on November 2, 2000, between the Washington Department of Ecology and the U.S. Department of Energy (DOE). The purpose of the meeting was to discuss the draft screening level risk assessment work plan for DOE's Hanford River Protection Privatization Project.

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**Issue 1: Prepare a brief report addressing the potential volatility of the radionuclides listed in RAWP-72 with respect to their becoming airborne in a sweat lodge (i.e., water containing these radionuclides splashed onto hot rocks to make steam).**

**Issue 2: Briefly review the list of 46 radionuclides and identify any other radionuclides, in addition to those listed in RAWP-72, that may become airborne and represent a potential inhalation exposure pathway in a sweat lodge.**

---

These two issues were formulated into the following question, which the discussion below attempts to answer:

> Assuming that radionuclides contaminate surfaces waters and these surface waters are used by native Americans in sweat lodges, what radioactivity exposure problems might result?

**The Sweat Lodge**

To prepare for a sweat lodge ceremony, igneous rocks such as lava are heated outside the lodge in a fire pit fueled with wood logs. (Lava tends to hold its heat well.) It takes several hours to heat the rocks, which may be about the size and shape of a man's head, to the required red heat. According to William Grosshandler, Acting Chief, Fire Sciences Division, National Institute for Science and Technology (301- 971-2310), the temperature of the glowing coals in an intense wood fire is about 1700°C, while the flame temperature is about 1200°C. Rocks heated to a dull red heat will have a temperature of about 650°C. The sweat lodge generally consists of a frame of bent willow boughs covered with blankets and tarpaulins. The entrance to the sweat lodge is covered with blankets. A typical sweat lodge might be about 10 feet in diameter and roughly hemispherical in shape. When it is time for the ceremony to begin, a certain number of heated rocks are brought into the lodge one-by-one and placed in a central pit in a ritual manner. Depending on the particular ritual, this might involve twelve rocks. When the heated rocks are in place, the entrance is sealed, water is sprinkled onto the rocks, and prayers and meditation begin. Four such rounds of ritual comprise the ceremony. Each round is about 45 minutes.

Information presented above was obtained at the following Internet sites:

- http://www.ausbcomp.com/redman/sweat_lodge.htm
- http://www.crystalinks.com/sweatlodges.html
- http://www.welcomehome.org/rob/sweat/sweat.html

**Aerosols in a Sweat Lodge**

Emissions from the vitrification process are expected to be either gaseous species (e.g., $CO_2$, $H_2O$, $I_2$) or solid

particulates (e.g., metal oxides and/or silicates).[1] Some fraction of the gaseous species will dissolve in surface waters on contact as $H_2O$, $I^-$, or $CO_3^{-2}$. Some fraction of the solid particulate emissions may also fall onto surface waters. Some of these particulates may settle to the bottom of the body of water, some may dissolve in the water, and some may remain suspended in the water as colloidal particles. Particles which settle out will not contribute to the types of exposures addressed here.

As noted above, water is sprinkled on the heated rocks in the sweat lodge to produce a steam-laden atmosphere. Any tritium, as tritiated water, would be vaporized in the sweat lodge. Similarly, any carbon-14 existing as dissolved carbonic acid would also be vaporized. Other dissolved radioactive species (e.g., metal ions and I-) would most likely remain on the igneous rocks as metal salts which might or might not subsequently evaporate depending on the chemical form of the resulting compounds. The melting and boiling points of some possible compounds are listed in Table 1 (Hdbk 1954).

It can be seen from Table 1 that if antimony chloride or antimony iodide is formed as a result of evaporation, these compounds could volatilize. Selenium, if present as the oxide, could also vaporize under expected sweat lodge conditions.

Ruthenium metal is quite stable and oxides slowly in air at temperatures above 800°C. The metal does not react with air at room temperature. The oxide, $RuO_4$, is highly volatile with a quoted boiling point of either 40°C or 130°C (**http://www.emsdiasum.com/ems/techdata/57.html).** However, this oxide can not be formed from the elements (http://www.britannica.com/bcom/eb/article/2/0,5716,119792+28+110614,00.html); rather, complex chemical synthesis techniques are required. Consequently, volatilization of ruthenium is not expected to represent a realistic exposure pathway.

Cesium metal boils at about 690°C. As a result, if the element is present in metallic form, perhaps due to decomposition of the oxide, it would not be vaporized in the sweat lodge. If the cesium salts, such as the chloride, iodide, or sulfate, reformed on the heated rocks after the steam had evaporated, none of these compounds would be expected to volatilize.

It is also possible that, instead of remaining as evaporative salts on the heated rocks, some of the dissolved species could be physically airborne as an aerosol mist if the boiling process is sufficiently violent. Whatever mechanism is responsible for the generation of aerosols (vaporization, mechanical entrainment, or volatilization), the quantities of such materials will likely be relatively small since only small quantities of water are used in the ceremonies. For example, consider a sweat lodge in the form of a hemisphere 10 feet in diameter. The lodge will contain about 262 cubic feet of air. Assuming that the lodge contains saturated air at a temperature of 100°F (38°C), then the air will contain 0.043 lb of water vapor per pound of dry air, and the saturated air will have a specific volume of 15.1 cubic feet per pound of dry air (Perry's 1984). Thus, there will be 0.74 lbs of water vapor in the lodge (261 $ft^3$ x 0.043 lb of $H_2O$/lb dry x lb dry air/15.1 $ft^3$). Since a gallon of water weighs 8.3 pounds, the amount of water required to saturate the air in the sweat lodge is about 0.1 gallons. Hence, the total amount of a contaminant airborne in the sweat lodge at any given time would not exceed the amount of the contaminant that is in about 0.1 gallon of water.

[1] Joule-heated ceramic melters used in the vitrification of HLW operate at about 1100°C in an oxidizing atmosphere. Thus, solid particulates are expected to be oxides or silicates.

**TABLE 1**
**MELTING POINTS AND BOILING POINTS OF SELECTED INORGANIC COMPOUNDS**

| Compound | Melting Point (°C) | Boiling Point (°C) |
|---|---|---|
| $SbCl_3$ | 74 | **234** |
| $SbI_3$ | 167 | **401** |
| $Sb_2O_3$ | 656 | 1550 |
| $BaCl_2$ | 962 | 1560 |
| $BaI_2$ | 740 | NR |
| BaO | 1923 | NR |
| $BaSiO_3$ | 1604 | NR |
| $BaSO_4$ | 1580 | NR |
| $CdCl_2 \cdot 2.5H_2O$ | 568 | 960 |
| $CdI_2$ | NR | 713 |
| CdO | NR | 900-1000 (d) |
| $CdSO_4$ | NR | 1000 |
| CsCl | 646 | 1290 |
| CsI | 621 | 1280 |
| $Cs_2O$ | 360-400 (d) | NR |
| $Cs_2SO_4$ | 1010 | NR |
| CoO | 1800 (d) | NR |
| $CoSO_4$ | 989 | NR |
| $EuCl_3$ | 623 | NR |
| NiO | 2090 | NR |
| $RaCl_2$ | 1000 | NR |
| $RuCl_3$ | >500 (d) | NR |
| $SmCl_3$ | 678 | NR |
| $SmI_3$ | 820 | NR |
| $SeO_2$ | NR | **316** |
| $SrCl_2$ | 873 | NR |

| Compound | Melting Point (°C) | Boiling Point (°C) |
|---|---|---|
| $Sr(OH)_2$ | 375 | NR |
| SrO | 2430 | NR |
| $SrSO_4$ | 1580 (d) | NR |
| $SnCl_2$ | 246 | **623** |
| SnO | 700-950 (d) | NR |
| $ThCl_4$ | 720-750 (sub) | NR |
| $YCl_3$ | 680 | NR |

Notes:

Chemicals with boiler point values in **bold** would be expected to volatilize.

d Decomposes
sub Sublimes
NR Not reported

---

**Recommendation:**

Clearly, the quantity of radionuclides that may become airborne in this exposure scenario, and the potential significance of this scenario, will depend on many factors related to the chemical form of the radionuclides, the radionuclide concentration in the water, the temperature of the hot rocks, and the amount of water used in the ceremony. Given the many uncertainties, and the potential that aerosols may be generated by mechanical entrainment in addition to volatilization, it is recommended that a two-step process be employed for the assessment of this pathway. The first step would be a screening process, wherein it would be assumed that all of the radionuclides in the water used in the sweat lodge become airborne. If these levels result in potential risks exceeding 1E-6, a more refined analysis could then be initiated for the more limiting radionuclides.

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**Issue 3: Check if the HEAST slope factor for inhalation of tritiated water vapor includes uptake by dermal absorption.**

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The inhalation slope factor for tritiated water vapor reported in Table 4 of HEAST is 9.59 E-14 lifetime risk of cancer per pCi inhaled. This value includes both the risk contribution from the internal dose delivered by the tritium that is inhaled plus the tritium that is taken into the body by dermal absorption. This can be demonstrated by the following calculation:

The risk from inhalation of 1 pCi of tritiated water vapor, not including dermal absorption, is derived as follows:

Dose = 1 pCi x .037 dis/sec-pCi x .0057 MeV/dis x 10 d/.693 x 86400 s/d
x 1.6E-06 erg/MeV x .01 rad-g/erg /70,000 g = 6.0E-11 rad/pCi inhaled

Risk = 6.0E-11 rad/pCi x 7.6E-4 risk/rad = 4.56E-14 risk/pCi inhaled

Other than physical constants, the key parameters in this equation are the effective half-life of tritiated water in the body of 10 days, the body weight of reference man of 70 kg, and the risk coefficient for uniform whole body exposure to ionizing radiation of 7.6E-4 lifetime risk per rad uniform whole body exposure.

As may be noted, the result of the above calculation is about one half the slope factor. Since, it is widely acknowledged that the internal dose from immersion in a plume of tritiated water vapor is about 50% from inhalation and 50% from dermal absorption[2], it is clear that the HEAST slope factor includes a factor of two to account for dermal absorption. A telephone conversation with Michael Boyd of the Office of Radiation and Indoor Air (11/3/00) confirmed this understanding.

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**Issue 4: Check to determine if dermal absorption of I-129 can be a significant contributor to risk relative to inhalation.**

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Guidance on the possible significance of dermal absorption, relative to inhalation, as a route of exposure to airborne toxicants is provided in "Dermal Exposure Assessment: Principles and Applications," EPA/600/8-91/001B, January 1992. As indicated on page 7-1, as a general rule, "many chemicals due to their low vapor pressure cannot achieve adequate vapor concentrations to pose a dermal hazard" and "for chemicals that can achieve adequate vapor concentrations, it has been assumed that they are primarily absorbed by the respiratory tract." In order to confirm this generalization, it is instructive to evaluate the permeability constant that an I-129 vapor must have in order for it to contribute significantly to uptake relative to inhalation.

Assuming a typical breathing rate of 8,000 m3/yr and an exposed skin surface area of 5,800 cm2[3], the permeability constant (Kp) for a vapor that would correspond to an uptake rate via dermal absorption that is comparable to the uptake by inhalation is derived as follows:

Kp (cm/hr) = (15.2 m3/day x 1E06 cm3/m3)/(5,200 cm2 x 24 hr/day) = 122 cm/hr

Therefore, the permeability constant for I-129 vapor would have to be 122 cm/hr in order for dermal absorption to contribute as much to I-129 uptake as does inhalation. The permeability constants reported in Table 7-1 in the above cited EPA guidance for a broad range of vapor phase organic compounds, which have a high potential for dermal absorption, range from .01 to 14.9 cm/hr. Clearly, dermal absorption of I-129 vapor cannot be a significant contributor to risk relative to inhalation of I-129 vapor.

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[2] See Section 9.3.2 (page 9-4) of "Radiological Assessment - A Textbook on Environmental dose Analysis," Edited by John E. Till and H. Robert Meyer, NUREG/CR-3332, September 1983.

[3] These are the recommended adult long term inhalation value and the upper end exposed skin surface area value on pages 5-24 and 6-5 of "Exposure Factors Handbook," EPA/600/P-95/002Fa, August 1997.

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**Issue 5: Check to determine if I-129 can represent an external exposure risk from plume immersion which may be significant relative to the risk it represents by inhalation.**

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I-129 is a pure beta emitter and will, therefore, not deliver a significant external effective whole body dose from plume immersion. As a result, the lifetime risk associated with immersion in an airborne plume of I-129 will essentially be entirely due to the I-129 taken up by inhalation, with a negligible contribution from dermal absorption and external exposure. For example, assuming an airborne plume of 1 pCi/m3 of I-129, and using the HEAST inhalation slope factors, the lifetime risk of cancer due to one year of inhalation exposure to the plume is estimated as follows:

$R_{inh}$ = 1 pCi/m3 x 8000 m3/yr x 1.22E-10 risk per pCi inhaled = 9.76E-07 lifetime cancer risk of cancer from inhalation

Using the external risk conversion factors for I-129 in Federal Guidance Report No. 13, the lifetime risk from the external exposures from one year exposure to a plume containing 1 pCi/m3 of I-131 is estimated as follows:

$R_{ext}$ = 1 pCi/m3 x 1.85E-17 risk per Bq per m3 per sec x 0.037 Bq/pCi x 3.15E7 sec/yr = 2.15E-11 lifetime risk of cancer from external exposure

Hence, the risk from external exposure is over four orders of magnitude smaller than the inhalation risk.

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**Issue 6: Prepare a brief statement defining the level of exposure that may be considered a LOAEL for radionuclides (e.g., 1 to 5 rem).**

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The Appendix presents a review of the literature which establishes the bases for defining LOAELs and NOAELs for radionuclides. Though the subject is complex, requiring a number of qualifying statements, in brief, the lowest levels of exposure where clinically significant non-stochastic effects (i.e., the acute effects of radiation) have been observed is about 10 rem. The lowest doses where a statistically significant increase in the incidence of stochastic effects (i.e., cancer) have been observed in an exposed population was about 1 rem uniform whole body exposure delivered over a short period of time to a large population.

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**Issue 7: Provide a brief report on the concentrations of naturally occurring and ubiquitous manmade radionuclides in mother's breast milk.**

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Available sources of information were searched, however no data on the natural background and ubiquitous manmade levels of radionuclides in human milk were located. The best evidence found were data on the radionuclide content in cow's milk and produce in the vicinity of Hanford. These data are included in an EXCEL spreadsheet (electronic copy transmitted with this memorandum). These data, which were kindly provided by Bruce Napier of Pacific Northwest National Laboratory, are the results of the 1999 environmental radiological surveillance program (Poston and others 1999). Dr. Napier explained that these annual reports can be obtained at **http://www.hanford.gov/doe/98annualrp/index.html.**

Human milk for women in the vicinity of Hanford would likely contain substantially lower levels of radionuclides than cow's milk since the amount of food and the radionuclide content of the cows' diet is likely to be considerably greater than that of a person. Nevertheless, the data in the spreadsheets represent a baseline that may be useful. In theory, the radionuclide content in human milk in the vicinity of Hanford can be estimated based on human dietary intake, along with the application of biokinetic models on the uptake and retention of radionuclides in human milk.

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# Appendix to Tetra Tech EM Inc. Letter

**APPENDIX**

**LOAELs ASSOCIATED WITH RADIATION EXPOSURE**

## 1.0 BACKGROUND INFORMATION

The concept of the "Lowest Observed Adverse Effect Level" (LOAEL) has been employed by public health professionals to assist their efforts to provide policies, guidance, and set regulatory limits in behalf of individuals exposed to radiation and radioactive materials. A LOAEL is the lowest dose in a given study that resulted in an observable harmful health effect. Radiation health effects are generally categorized as either deterministic or stochastic.

## 1.1 DETERMINISTIC HEALTH EFFECTS

Deterministic effects are those with a threshold dose and where the severity of the health effect(s) is largely defined by the total dose of radiation that is delivered to tissue(s), organ(s), or the whole body of the individual. These health effects are termed "acute radiation health effects" and are generally seen only for relatively large doses above the threshold level that are delivered within a short time period.

Modifying factors that affect the dose-response relationship are numerous and include (1) the rate at which the dose is delivered, (2) the type of radiation (alpha, beta, gamma, or neutron), (3) the exposure pathway (external versus internal exposure from ingestion or inhalation), and (4) age, sex, and health status of the individual.

## 1.2 STOCHASTIC HEALTH EFFECTS

In contrast to deterministic health effects, the severity of stochastic health effects is not affected by radiation dose. By definition, stochastic radiation effects are "probabilistic" health effects that include (1) cancer induction, (2) genetic effects, and (3) in-utero effects. Thus, the distinguishing feature of the dose-response relationship of a stochastic effect is that the severity is not dose dependent; rather the probability that a stochastic effect may occur is directly proportional to the dose of radiation. A second distinguishing feature of the dose-response relationship of a stochastic effect is that it is assumed to have no threshold. Thus, stochastic health effects associated with chronic low doses or low dose rates of radiation are assumed to represent a linear no-threshold (LNT) dose response. Thus, for even a very small dose of radiation, it is assumed that there is a small but finite risk of cancer, genetic, or in-utero effect.

A familiar example of a stochastic effect is that of smoking and lung cancer. Indisputably, cigarette smoking is a direct cause of human lung cancer, but not all smokers develop lung cancer. Moreover, lung cancer may also be observed in some non-smokers. It is important to note that the "severity" of a lung cancer is independent of whether the individual was a heavy smoker, light smoker, or non-smoker. Thus, the causal relationship of cigarette smoking and lung cancer was established when a higher incidence rate of lung cancer was observed among smokers than among non-smokers. The level of increase was found to be proportional to the amount and duration of cigarette smoking. While large differences in lung cancer rates were readily observable when heavy smokers were compared to non-smokers, these differences diminished to indistinguishable levels for very light smokers or individuals who had smoked only for a very short time.

A similar relationship exists between radiation exposure and several types of stochastic effects. For small doses of radiation, the likelihood that even a single cell will undergo a selective alteration, which leads to a

cancer or some other health effect, is extremely low. Furthermore, genetic effects, disturbances in growth and development of an embryo, and cancer can also be caused by chemical, physical, and biological agents, many of which exist naturally in the environment. Thus, even for large doses of radiation, stochastic health effects can be observed only as relatively small increases above the spontaneous incidence that is observable in the normal population.

It must also be acknowledged that the slope of the dose-response relationship for stochastic health effects is also modified by (1) the type of cancer, (2) sex and age at time of exposure, and (3) the type of radiation, pathway of exposure, etc. For example, for a given dose of radiation to the thyroid, the risk of thyroid cancer is highest when radiation is external and the exposed individual is a female child.

A quantitative assessment of the radiation dose-response relationship is further complicated by the fact that cancers (and other stochastic effects) induced by radiation are indistinguishable from those arising spontaneously or caused by other carcinogens. Physicians and pathologists cannot determine, based on tissue type, whether certain lung cancers, for example, are caused by radiation, cigarette smoking, air pollutants, chemicals, or other cancer-causing agents. The ability to detect the common cancers caused by any specific agent is, therefore, limited to statistical analyses. These statistical methods rely on the fact that the incidence of various cancers in a well-defined population can be predicted with reasonable accuracy. For a sufficiently large group of people who have received radiation exposure, an incidence of cancer above the expected level would suggest radiation was a possible cause of the excess number of cancers, but it would not identify radiation as the cause of cancer in any specific individual. Only epidemiologic studies of people exposed to relatively high doses of radiation (greater than 10,000 mrem or 10 rem) have shown such an excess of cancer and have, therefore, demonstrated a causal relationship.

In brief, there exists a voluminous body of data that describes the dose-response relationship and, while there is general consensus at the high end of the dose response, there remains uncertainty and controversy at the low end.

It is the combination of these factors that complicate a quantitative assessment of LOAELs associated with radiation exposure. A detailed and comprehensive discussion is, therefore, beyond the scope of this task. Presented below, however, are select citations of observed radiation health effects and their reported doses that provide useful reference values for LOAELs representing both deterministic and stochastic radiation health effects.

## 2.0 LOAELs FOR DETERMINISTIC OR ACUTE RADIATION HEALTH EFFECTS

Radiation affects the individual cells that are the building blocks of the tissues and organs of the body. Although all cells can be affected by radiation, some are more sensitive to radiation injury than others. In general, the degree of sensitivity depends on the rate of cell division and the degree of cell differentiation. Thus, the most sensitive cells are undifferentiated rapidly dividing cells that include somatic stem cells and precursor cells to male sperm. The key feature of deterministic effects is that they require a minimum dose that in turn induces cell death in a significant fraction of the exposed cell population that represents a particular tissue/organ.

Human exposure to a single whole-body dose of rapidly delivered radiation of 50 rem or more results in the development of a complex of clinical symptoms, signs, and laboratory findings, which are collectively termed the Acute Radiation Syndrome. In the acute radiation syndrome, the very radiosensitive hemopoietic system is the most prone to manifest evidence of injury. It is only when injury is more severe that gastrointestinal symptoms dominate the picture.

Presented below is a brief discussion of prominent features of the acute radiation syndrome in terms of time of onset and required radiation doses.

## 2.1 EARLY PRODROMAL SYMPTOMS

The first phase of the acute radiation syndrome is characterized by nausea, vomiting, and diarrhea. The best statistical information on the amount of radiation required to cause various levels of early acute radiation sickness (known clinically as prodromal gastro-intestinal distress) has been derived largely from an analysis of clinical data obtained from the histories of therapeutically and accidentally irradiated persons. The radiation exposures predicted to cause 50% probabilities of loss of appetite, nausea, vomiting, diarrhea and fatigue in such patients are listed in Table A-1.

## 2.2 HEMOPOIETIC SYNDROME

Clinical changes that develop in the blood following acute exposure often are referred to as the *hemopoietic syndrome*. The earliest change is a fall in the absolute peripheral lymphocyte count. This commences in the first few hours and continues for several days to levels commensurate with the amount of radiation exposure within certain limits. Reduced lymphocyte levels may persist for several weeks. There often is a prompt increase in the leukocyte count during the first few days, then a leveling off for a few more days, following which the granulocyte count will continue to fall with maximum leukopenia developing in two to five weeks. Large doses of radiation may result in severe granulocytopenia within the first seven to ten days, a poor prognostic indicator. Recovery may take several weeks to months. The platelet count usually begins to fall one to two weeks after exposure. Massive radiation exposure doses may cause severe thrombocytopenia to develop much earlier. It may take several months before the platelet counts return to normal. Usually there is a slow decline in the erythrocyte count associated

**TABLE A-1**
**ESTIMATES OF SINGLE RADIATION EXPOSURES THAT WILL CAUSE 50% INCIDENCE OF PRODROMAL RESPONSES (EARLY SYMPTOMS) IN MAN[a]**

| Level of Radiation Sickness | Single Radiation Exposure (R[b]) | 95% Confidence Range(R) |
|---|---|---|
| Anorexia (loss of appetite for food) | 180 | 150-210 |
| Nausea | 260 | 220-290 |
| Fatigue | 280 | 230-310 |
| Vomiting | 320 | 290-360 |
| Diarrhea | 360 | 310-410 |

[a] Source: *Radiobiological Factors in Manned Space Flight*, Edited by W. Langham, National Academy of Sciences Publication 1487, National Academy of Sciences, National Research Council, Washington, D.C., 1967.
[b] Measured in air.

with reticulocytopenia, the extent of which depends on the amount of radiation exposure and the severity of the acute radiation syndrome.

One difficulty for defining a LOAEL that involves a deterministic effect is the subjective interpretation for deciding when an "observed effect" is truly "adverse." Acute doses between 300 and 400 rem to hemopoietic tissue are generally regarded as mid-lethal doses, while doses below 100 rem are sub-lethal and result in cell depression that are transient and reversible. Wald and others (1962) provides the following doses-response relationship for each of the major blood-cell components:

> Lymphocyte Count. The absence of any observable decrease has been equated with an exposure dose of less than 25 R; mild decrease and minor lymphopenia with less than a 100 R dose; a fall of greater than 50% and 90% with a dose of greater than 100 R. A pronounced fall has been taken to indicate a dose in the "dangerous range" from 300 to 1,000 rad. A lymphocyte count above 1,500/mm$^3$ has been considered to mean less than 200 R; less than 1,000/mm$^3$ to mean 200 to 400 or 500 R; less than 500/mm$^3$ to mean 400 or 500 to 900 R; and "virtually zero" to mean greater than 900 R.
>
> Neutrophile Count. A depression count in the fourth and fifth week has been equated with a dose of less than 200 R; severe depression in 3 to 5 weeks with a dose of 200 to 400 or 500 R; and severe depression on days 10 to 20 with a dose of 400 or 500 to 900 R.
>
> Platelet Count. A moderate depression of the platelet count during the fourth and fifth week has been associated with less than 200 R; severe depression in 3 to 5 weeks with 200 to 400 or 500 R; and sever depression on days 10 to 20 with 400 or 500 to 900 R.
>
> Reticulocyte Count. An "unequivocal fall" in the reticulocyte count in five days has been equated with a dose or greater than 300 rad.
>
> Mitotic Index. A progressive decrease has been equated with a dose in the 50 to 200 rad range. The absence of mitoses by the fourth day has been equated with a dose of 200 rad or more.

## 2.3 LOAELs ASSOCIATED WITH HUMAN GAMETES

Fertility. Radiation exposure of an individual's reproductive tissues may affect the production of mature male sperm or female egg cells. Reduced production of these cells may result in the temporary or permanent loss of ability to father or bear children. Sources of information about radiation effects on human fertility are limited to several studies involving medically exposed individuals (Rowley 1974; Upton 1974) and atomic bomb survivors (Blot 1977; Seigel 1966). Additionally, data from animal studies are generally thought to be applicable for estimating these effects on humans. Collectively, human and animal studies indicate that cells responsible for producing sperm in men and ova in women are among the more radiation-sensitive cells of the body. Nevertheless, radiation sensitivity differs between males and females with regard to reproductive fertility. These differences reflect the dynamics of sperm and egg production.

In the females, the ovary contains the complete inventory of about 2 million immature eggs (i.e., oocytes) at the time of birth. Following sexual maturation at puberty, monthly ovulation induces the production of a mature female egg. About 360 to 400 mature female oocytes are produced over her reproductive years. Immature preovulatory egg cells are relatively radioresistant, and following puberty, fertility is impaired only after moderately high doses of 300-400 rad (300,000-400,000 mrad) (NAS 1980). It is not surprising, therefore, that follow-up studies of female Japanese atomic bomb survivors have failed to demonstrate any long-term effects on female (and male) fertility (Blot 1977).

The male testes continuously produce reproductive sperm cells throughout life following sexual maturity. In this steady state of sperm cell renewal, cells are continuously produced to replace functional sperm cells that are expelled or leave the system. The production of mature sperm cells from testicular stem cells involves several cell divisions in which cells undergo dramatic changes. Radiation, which can profoundly impair cell division, is most detrimental in the early stages of sperm cell differentiation. Acute doses of a few rad can temporarily halt cell division at this stage and result in a transient reduction of sperm cell count (ICRP 1984). For increasing doses, the reduction in sperm cell count may lead to temporary or permanent male sterility. Sperm-count studies of males exposed to partial-body irradiation indicate that for gonadal doses of about one hundred to several hundred rad, sterility is temporary, and normal sperm counts resume within one to three years (Upton 1974). Thus, a dose that would permanently sterilize a man is thought to be greater than 500 rad (500,000 mrad) and would exceed the lethal whole body dose for acute radiation exposure (NAS 1980 (BEIR III)).

Among the 38,000 children born to parents irradiated at Hiroshima and Nagasaki with average doses between 31,000 and 39,000 mrem (31 and 39 rem), no statistically significant increase in genetic defects has been seen (Neel 1988; Schull 1981).

## 3.0 LOAELs ASSOCIATED WITH STOCHASTIC RADIATION HEALTH EFFECTS

Stochastic health effects may not appear for years or even decades after exposure to radiation. Such effects result from specific changes that occur in a few cells or a single cell. Although these selective cellular changes occur rarely, when they do, there is a probability that the altered cell may develop into cancer. If the altered cell is a reproductive cell, there is a possibility of transmitting genetic defects to the progeny of irradiated parents. Also, a developing embryo or fetus could possibly suffer injury if a pregnant woman is exposed to radiation. Thus, radiation-induced stochastic effects may exhibit long latency periods, are probabilistic, and involve biological end-points that occur relatively frequently among unexposed individuals. Because of these constraints, the most informative studies are those that involve (1) a large number of individuals, (2) large radiation doses, and (3) a follow-up period of several decades. These three parameters are frequently used to assess the strength of a study and are quantitatively expressed in person-rem-years.

Summarized in this section are epidemiologic studies grouped by the circumstances in which radiation was received. The categories include:

- Atomic Bomb Survivors
- Medical Exposures
- Fallout from Experimental Weapons Testing
- Occupational Exposures
- Others

## 3.1 CANCER

This section summarizes information concerning incidences of cancer related to radiation exposures.

### 3.1.1 Japanese A-Bomb Data

The most intensely studied human populations are the Japanese survivors of the 1945 atomic bombing of Hiroshima and Nagasaki. A-bomb survivors represent the largest group of humans exposed to radiation for whom estimates of individual doses are available. Survivors in the two cities were exposed to the immediate

external radiation produced by bomb blasts and to a lesser extent subsequent internal/external exposure from fallout. Of the 75,991 survivors for whom doses were estimated, 34,272 were so far from the hypocenters that their radiation doses were considered negligible (less than 500 mrem or 0.5 rem); thus, they serve as a comparison or "control" group, leaving 41,719 whose doses are estimated at 500 mrem (0.5 rem) or greater. Table A-2 provides the dose distribution for this group of nearly 76,000 individuals.

**TABLE A-2**
**DISTRIBUTION OF EXPOSURE AMONG A-BOMB SURVIVORS**

| Dose Range (Rad) | Number of Individuals |
|---|---|
| ≈ 0 | 34,272 |
| 1 - 5 | 19,192 |
| 6-9 | 4,129 |
| 10 - 99 | 15,346 |
| 100 - 199 | 1,946 |
| 200 + | 1,106 |

Data on cancer mortality among these 76,000 individuals have been collected and reported over the years. Relative to "controls" and adjusted for age and sex distribution, the number of observed cancer mortalities among the 76,000 A-bomb survivors has been compared to the number of expected mortalities if exposure to radiation had not occurred. The difference between the observed and expected numbers of cancer is assumed to be attributable to radiation exposure. The data in Table A-3 indicate that, of the 5,936 A-bomb survivors who died of cancer, about 340 of these cancers deaths are thought to have been the result of radiation exposure.

The data also define a dose-response in which increasing doses yield an increased percentage of excess cancers, especially for leukemia. However, some numerical estimates embody substantial statistical uncertainties as to the number of cancer deaths induced by radiation. Thus, for doses between 10,000 and 50,000 mrem (10-50 rem), the small number of excess cancers above normal expected levels is difficult to interpret and may reflect random fluctuations that are not linked to radiation exposure. When doses exceed 50,000 mrem (50 rem), the number of excess cancers is sufficient to support a causal link to human cancers.

**TABLE A-3**
**OBSERVED CANCER DEATHS AND NUMBER OF EXPECTED CANCER DEATHS AMONG A-BOMB SURVIVORS**

| Dose(Rem) | Approx. No. of Survivors | Leukemia | | | | Non-Leukemia | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Observed | Expected | Excess | | Observed | Expected | Excess | |
| | | | | No. | % | | | No. | % |
| 0 | 34,272 | 58 | 88 | - | 0 | 2443 | 2593 | - | 0 |
| 1 - 10 | 23,321 | 38 | 61 | - | 0 | 1655 | 1688 | - | 0 |
| 10 - 50 | 11,500 | 32 | 20 | 12 | 38 | 927 | 866 | 61 | 7 |
| 50 - 100 | 3,500 | 19 | 6 | 13 | 68 | 329 | 273 | 56 | 17 |
| 100 - 200 | 2,000 | 23 | 3 | 20 | 87 | 218 | 147 | 71 | 33 |
| 200 + | 1,000 | 30 | 2 | 28 | 93 | 132 | 68 | 64 | 48 |
| Total | 76,000 | 202 | 122 | 80 | 40 | 5,734 | 5,474 | 260 | 5 |

Source: Shimizu 1987.

Hanford Cancer Study. An observable excess cancer rate corresponding to much lower doses of radiation has been reported in studies involving occupationally exposed individuals. Mortality studies of Department of Energy (DOE) site workers were initiated in 1965 by the Atomic Energy Commission, under the direction of Dr. Thomas Mancuso of the University of Pittsburgh. Using mortality data dating to the 1940s, researchers examined the death rates among 44,100 Hanford employees. In 1977, Mancuso and his associates, Alice Stewart and George Kneale, first reported their findings (Mancuso 1977). Their analysis of death certificates for 1,336 "non-exposed" and 2,184 "exposed" male workers who died between 1944 and 1972 found statistically significant associations between cumulative external radiation dose and cancer mortality involving the lung, pancreas, and bone marrow. A subsequent analysis of 4,033 deaths among "radiation monitored" male and female workers also indicated elevated cancer risks among male and female workers for cancer of the pancreas, stomach, lung, and bone marrow (Kneale 1978).

The estimates of cancer risks from these two studies are markedly higher than estimates based on data from the Japanese A-bomb survivors and medically exposed populations. However, many scientists, including those belonging to the National Academy of Sciences, have criticized the studies' methodologies (NAS 1980).

### 3.1.2 Cancer Induction for Childhood Exposures

Some epidemiologic data suggest that young children may be more sensitive to the carcinogenic effects of radiation. However, these data also have not been without controversy. One such study involved thyroid cancers among individuals exposed during childhood.

Scalp Irradiation for Tinea Capitis in Israel. A total of 10,902 Jewish children immigrating into Israel were studied after having received scalp irradiation for ringworm. All but 60 of the patients were successfully traced and matched against an equal number of nonirradiated controls with tinea capitis and a nonirradiated sibling group of half the size. A sixfold increase in malignant thyroid tumors was found in the irradiated group, compared with the controls. Nine of the 12 thyroid cancers in the irradiated group occurred in females, most of them of the papillary-cell type. Ten of the tumors occurred between 9 and 16 years after therapy. A total of 10 patients who developed cancer had an estimated dose of about 6-9 rads to the thyroid, and the other two received 12 and 18 rads (Modan and others 1974; Modan and others 1977a; Modan and others 1977b).

Scalp Irradiation for Tinea Capitis in New York. Shore, Albert, and Pasternak reported on the second survey of a population of 2,215 irradiated and 1,395 nonirradiated control subjects with tinea capitis (Shore and others 1976). Scalp epilation was accomplished with essentially the same technique as in the Israeli population just discussed; the authors produced almost exactly the dosimetry estimates of 6-10 rads to the thyroid. The average age at irradiation was about 8 years, and the average interval of follow-up was about 20 years after irradiation. No thyroid cancers were observed, although patients with benign ademonas were identified. The variance of this study from that of Modan and others (1977b) may be due to the much smaller size of the population.

### 3.1.3 Cancer Among Children Exposure in Utero

Earlier epidemiologic studies of in-utero exposure have also yielded inconsistent data regarding the risks of in-utero exposure and subsequent childhood cancers. No significant excess mortalities from juvenile leukemia or other cancers were observed among the 1,630 pregnant Japanese women for embryo/fetus doses of 1,000 to 50,000 mrem (1 to 5 rem) (Jablon 1970). However, a tentative link between in-utero exposure and childhood cancers was reported in a study of pregnant women exposed to diagnostic radiation to the abdomen in doses in the range of 500 to 5,000 mrem (0.5 to 5 rem) (Stewart 1970; Monson 1984).

For years, the surveys involving diagnostic radiation remained controversial. It was suggested that the original surveys were flawed by certain selection criteria of study subjects since many of the radiological procedures were requested by physicians for medical reasons (Oppenheim 1974; Totter 1981). To rule out the possibility that the diagnostic exposure and observed childhood cancers were not causally linked but merely shared a common risk factor, additional studies involving twin pregnancies were undertaken (Harvey 1985). The study focused on twin pregnancies where the diagnostic x-rays were performed solely because the pregnancy involved twins rather than an existing (or suspected) medical problem, as in the previously studied singleton births. When irradiated twin pregnancies were compared to non-irradiated twin pregnancies, a small increase in childhood cancers was observed (Harvey 1985). However, even this improved study design was clouded by the fact that the majority of individual twins affected by childhood cancers were children of mothers with a history of previous pregnancy loss, which may have predisposed these children to cancer. The National Academy of Sciences (NAS 1990) in its reevaluation of all current data nevertheless stated:

> "These complications, notwithstanding the concordance of the studies of twins with the studies of prenatally irradiated singleton births, prompt the tentative conclusion that susceptibility to the carcinogenic effects of irradiation is high during prenatal life."

Based on the limited available human data, the National Academy of Sciences estimated the risk per unit absorbed dose to be between 0.2 and 0.25 excess cancer deaths in the first 10 years of life per 1,000 children each receiving 1,000 mrem (1 rem) of exposure before birth. About 50% of the excess cancers would be expected to be leukemia.

## 3.2 LOAELS FOR OTHER IN-UTERO DEVELOPMENT EFFECTS

Although animal experiments have shown developmental health effects in the embryo/fetus for radiation doses as low as 5,000 to 10,000 mrem (5 to 10 rem), it can not be demonstrated with certainty that such low doses can induce injury to a human fetus. The evidence is based on the epidemiologic studies of children born to women of Hiroshima and Nagasaki who were exposed to atomic radiation in- utero. The atomic bomb studies were not able to associate doses below 25,000 mrem (25 rem) with developmental abnormalities of the newborn, such as central nervous system defects, skeletal abnormalities, or reduced stature. For doses above 25,000 mrem (25 rem), the most definitive human data concerning the effects of prenatal irradiation are related to brain development (Beebe 1981). In humans, impaired central nervous system development may lead to small-head size and/or severe mental retardation. Severe mental retardation in the fetus is most likely to result from exposure during the 8th to the 15th week of pregnancy, a period when specific cells, including those of the brain, are undergoing crucial development.

Among the approximately 1,600 Japanese subjects studied who had been exposed to radiation in-utero, there were 30 cases of severe mental retardation (Otake 1987). Severe mental retardation was defined as unable to perform simple calculations, to make simple conversation, or to care of himself or herself (i.e., institutionalized). The association between severe mental retardation and small-head size is not clear. Of the 30 cases of severe mental retardation, 18 individuals exhibited small-head size. For the entire study cohort, the number of individuals exhibiting small heads totaled 71 (Wood 1965).

Aside from the classification of severe mental retardation, the study cohort of 1,600 individuals exposed in-utero were also given intelligence tests (i.e., Koga test). Intelligence test (Koga) scores of the exposed individuals revealed that radiation-related effects on intelligence was most pronounced when exposure in-utero occurred 8-15 weeks after conception. The distribution of test scores suggests a progressive reduction in IQ scores with increasing radiation exposure. For fetal exposure in the 8th through 15th week, the reduction in intelligence score under a linear dose-response model was 21-29 points at a dose of 100 rad (100,000 mrad)

(Schull 1988). For a fetal dose of 1 rad (1,000 mrad), the corresponding risk implies a reduction of about one-quarter of one IQ point.

## 3.3 LOAELs ASSOCIATED WITH GENETIC MUTATIONS

Radiation exposure of reproductive cells have the potential for inducing developmental malformation, still births, neonatal deaths, and ill-health (inclusive of cancer) in the offspring that is the result of a genetic mutation in the exposed gamete. Japanese A-bomb survivors to date have provided important information using biochemical indicators to screen for mutations. In a total of 289,868 locus tests, involving measurements of 28 different protein phenotypes using one-dimensional electrophoresis to detect protein variants, Neel et al. (1980), have found one probable mutation in the offspring of proximally exposed parents, who received an estimated average gonadal exposure of 31-39 rem in the atomic bombings of Hiroshima and Nagasaki. There were no mutations in 208,196 locus tests involving children of distally exposed parents, who received essentially no radiation exposure. These findings correspond to mutation rates of $0.34 \times 10^{-5}$ per locus per generations in the proximally exposed parents and zero in the distally exposed parents.

However, the significance of this observed gene mutation remains uncertain. Among the 38,000 children born to parents irradiated at Hiroshima and Nagasaki with average doses between 31,000 and 39,000 mrem (31 and 39 rem), no statistically significant increase in genetic defects has been seen (Neel 1988; Schull 1981).

It is also recognized that certain types of cancers have a heritable component. To test the hypothesis that a parent's job exposure to ionizing radiation affects his or her child's risk of cancer, investigators compared this occupational exposure the year before the child's birth for parents of children with and without cancer (Hicks 1984). The parents of 283 children diagnosed with cancer and the parents of controls were identified and classified by profession (i.e., dentists, radiologists, x-ray technicians, etc.) and industry (i.e., nuclear industry, veterinary medicine, industrial radiography, etc.) in which the potential for occupational exposure was high, moderate, or none. The researchers found no evidence of increased cancer risks among children whose parent(s) worked in occupations classified as having high potential exposures. Another study, however, found that leukemia incidence was higher than normal among children fathered by men who had previously received comparatively high exposures (Gardner 1990). However, this observation by no means proves a causal connection between occupational irradiation of a parent and leukemia in the offspring. In fact, any assumed causal relationship is inconsistent with what is known about radiation genetics, mechanisms of leukemogenesis, and the results of other independent epidemiologic studies.

## 4.0 CONCLUSION

LOAELs between 10 rem (10,000 mrem) and several 10's of rem have been cited in the literature in behalf of <u>deterministic</u> radiation health effects. These effects principally reflect fractional cell death and reduced division of hemopoietic and male reproductive stem cells. However, these low-level effects are transient and reversible and, therefore, require a subjective interpretation of the definition of an “adverse” health effect. Moreover, for deterministic effects, the radiation dose must be delivered over a very short time and would reflect “accidental” conditions of exposure.

LOAELs of less than 10 rem (10,000 mrem) for <u>stochastic</u> radiation health effects are primarily linked to childhood cancer induction (i.e., cancers that result from radiation exposure received during childhood/in-utero, or by genetic mutation of male sperm), and other in-utero effects. However, some of these data remain controversial.

These values must be viewed in context with regulatory exposure limits for members of the public. For all anthropogenic sources of radiation, the regulatory limit for public exposure is 0.1 rem per year (or 100 mrem per year). For a discrete source of radiation exposure, the most common limit for public exposure is 0.025 rem per year (or 25 mrem per year).

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# Appendix C

# Lists of Common Vascular Plants, Mammals, Common Birds, Amphibians and Reptiles, Plant Species of Concern, and Wildlife Species of Concern

# Appendix C
# Lists of Common Vascular Plants, Mammals, Common Birds, Amphibians and Reptiles, Plant Species of Concern, and Wildlife Species of Concern

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# References for Appendix C

## Project Documents

None

## Codes and Standards

None

## Other Documents

Neitzel DA, ed., Bunn AL, Cannon SD, Duncan JP, Fowler RA, and others. 2005. *Hanford Site National Environmental Policy Act (NEPA) Characterization*, PNL-6415, Rev 17, Pacific Northwest National Laboratory, Richland, Washington.

**Table C-1 Common Vascular Plants Found on the Hanford Site**

| Scientific Name | Common Name |
|---|---|
| **Shrub-Steppe Species** | |
| **Shrub** | |
| *Artemisia tridentata* | big sagebrush |
| *Artemisia tripartita* | threetip sagebrush |
| *Chrysothamnus viscidiflorus* | green rabbitbrush |
| *Ericameria nauseousa* | gray rabbitbrush |
| *Eriogonum niveum* | snow buckwheat |
| *Grayia (Atriplex) spinosa* | spiny hopsage |
| *Purshia tridentata* | bitterbrush |
| **Perennial Grasses** | |
| *Achnatherum hymenoides* | Indian ricegrass |
| *Agropyron desertorum (cristatum)*[(a)] | crested wheatgrass |
| *Elymus elymoides* | bottlebrush squirreltail |
| *Elymus macrourus* | thickspike wheatgrass |
| *Koeleria cristata* | prairie junegrass |
| *Poa sandbergii (secunda)* | Sandberg's bluegrass |
| *Pseudoroegnaria spicata* | bluebunch wheatgrass |
| *Sporobolus cryptandrus* | sand dropseed |
| *Stipa comata* | needle-and-thread grass |
| **Biennial/Perennial Forbs** | |
| *Achillea millefolium* | yarrow |
| *Arenaria franklinii* | Franklin's sandwort |
| *Astragalus caricinus* | buckwheat milkvetch |
| *Astragalus sclerocarpus* | stalked-pod milkvetch |
| *Balsamorhiza careyana* | carey's balsamroot |
| *Brodiaea douglasii* | Douglas' clusterlily |
| *Chaenactis douglasii* | hoary falseyarrow |
| *Comandra umbellata* | bastard toad flax |
| *Crepis atrabarba* | slender hawksbeard |
| *Cymopteris terebinthinus* | turpentine spring parsley |
| *Erigeron filifolius* | threadleaf fleabane |
| *Erysimum asperum* | rough wallflower |
| *Fritillaria pudica* | yellow bell |
| *Helianthus cusickii* | Cusick's sunflower |
| *Lomatium grayi* | Gray's desertparsley |
| *Machaeranthera canescens* | hoary aster |

**Table C-1 Common Vascular Plants Found on the Hanford Site**

| Scientific Name | Common Name |
|---|---|
| *Oenothera pallida* | pale eveningprimrose |
| *Penstemon acuminatus* | sand beardtongue |
| *Phlox longifolia* | longleaf phlox |
| *Psoralea lanceolata* | dune scurfpea |
| *Rumex venosus* | winged dock |
| *Sphaeralcea munroana* | Munro's globemallow |
| *Thelypodium laciniatum* | cutleaf ladysfoot mustard |
| *Tragopogon dubius*[(a)] | yellow salsify |
| **Annual Forbs** | |
| *Ambrosia acanthicarpa* | bur ragweed |
| *Amsinckia lycopsoides* | tarweed fiddleneck |
| *Chorispora tenella*[(a)] | blue mustard |
| *Cryptantha circumscissa* | matted cryptantha |
| *Cryptantha pterocarya* | winged cryptantha |
| *Descurainia pinnata* | western tansymustard |
| *Draba verna*[(a)] | spring whitlowgrass |
| *Epilobium paniculatum* | tall willowherb |
| *Erodium cicutarium*[(a)] | storksbill |
| *Holosteum umbellatum*[(a)] | jagged chickweed |
| *Lactuca serriola*[(a)] | prickly lettuce |
| *Lepidium perfoliatum* | clasping pepperweed |
| *Mentzelia albicaulis* | whitestem stickleaf |
| *Microsteris gracilis* | pink microsteris |
| *Phacelia linearis* | threadleaf scorpion weed |
| *Plantago patagonica* | Indian wheat |
| *Plectritis macrocera* | white cupseed |
| *Polemonium micranthum* | annual Jacob's ladder |
| *Salsola kali*[(a)] | Russian thistle (tumbleweed) |
| *Sisymbrium altissimum*[(a)] | Jim Hill's tumblemustard |
| **Annual Grasses** | |
| *Bromus tectorum*[(a)] | cheatgrass |
| *Festuca microstachys* | small sixweeks |
| *Festuca octoflora* | slender sixweeks |
| **Riparian Species** | |
| **Trees and Shrubs** | |
| *Morus alba*[(a)] | white mulberry |

**Table C-1 Common Vascular Plants Found on the Hanford Site**

| Scientific Name | Common Name |
|---|---|
| *Populus trichocarpa* | black cottonwood |
| *Prunus* spp. | peach, apricot, cherry |
| *Robinia pseudo-acacia* | black locust |
| *Salix amygdaloides*[(a)] | peachleaf willow |
| *Salix exigua* | coyote willow |
| *Salix* spp. | willow |
| **Perennial Grasses and Forbs** | |
| *Agrostis* spp. [(b)] | bentgrass |
| *Alopecurus aequalis* [(b)] | meadow foxtail |
| *Apocynum cannabinum* | dogbane |
| *Artemisia campestris* | Pacific sage |
| *Artemisia ludoviciana* | prairie sagebrush |
| *Carex* spp.[(b)] | sedge |
| *Centaurea repens*[(a)] | Russian knapweed |
| *Coreopsis atkinsoniana* | horseweed tickseed |
| *Equisetum* spp. | horsetails |
| *Eragrostis* spp. [(b)] | lovegrass |
| *Gaillardia aristata* | blanket flower |
| *Grindelia columbiana* | Columbia River gumweed |
| *Heterotheca villosa* | hairy golden aster |
| *Juncus* spp. | rushes |
| *Lupinus* spp. | lupine |
| *Phalaris arundinacea*(a,b) | reed canary grass |
| *Polygonum persicaria* | heartweed |
| *Scirpus* spp.[(b)] | bulrushes |
| *Solidago occidentalis* | western goldenrod |
| *Typha latifolia*[(b)] | cattail |
| *Veronica anagallis-aquatica* | water speedwell |
| **Aquatic Vascular Species** | |
| *Elodea canadensis* | Canadian waterweed |
| *Lemna minor* | duckweed |
| *Myriophyllum spicatum* | spiked water milfoil |
| *Potamogeton* spp. | pondweed |
| *Rorippa nasturtium-aquaticum* | watercress |

**Table C-1 Common Vascular Plants Found on the Hanford Site**

| Scientific Name | Common Name |
|---|---|

Source: Neitzel DA, ed., Bunn AL, Cannon SD, Duncan JP, Fowler RA, and others. 2005. *Hanford Site National Environmental Policy Act (NEPA) Characterization*, PNL-6415, Rev 17, Pacific Northwest National Laboratory, Richland, Washington.

[(a)] Introduced
[(b)] Perennial grasses and graminoids.

**Table C-2 List of Mammals Occurring on the Hanford Site**

| Scientific Name | Common Name |
|---|---|
| **Shrews (family Soricidae)** | |
| *Sorex merriami* | Merriam's shrew |
| *Sorex vagrans* | vagrant shrew |
| **Evening bats (family Vespertilionidae)** | |
| *Antrozous pallidus* | pallid bat |
| *Eptesicus fuscus* | big brown bat |
| *Lasionycteris noctivagans* | silver-haired bat |
| *Lasiurus cinereus* | hoary bat |
| *Myotis californicus* | California myotis |
| *Myotis leibii* | small-footed myotis |
| *Myotis lucifugus* | little brown myotis |
| *Myotis volans* | long-legged myotis |
| *Myotis yumanensis* | Yuma myotis |
| *Pipistrellus hesperus* | western pipistrelle |
| **Hares, rabbits (family Leporidae)** | |
| *Lepus californicus* | black-tailed jackrabbit |
| *Lepus townsendii* | white-tailed jackrabbit |
| *Sylvilagus nuttallii* | Nuttall's (or mountain) cottontail |
| **Chipmunks, marmots, Squirrels (family Sciuridae)** | |
| *Marmota flaviventris* | yellow-bellied marmot |
| *Spermophilus townsendii* | Townsend's ground squirrel |
| *Spermophilus washingtoni* | Washington ground squirrel |
| *Tamias minimus* | least chipmunk |
| **Pocket gophers (family Geomyidae)** | |
| *Thomomys talpoides* | northern pocket gopher |
| **Heteromyid rodents, pocket mice (family Heteromyidae)** | |
| *Perognathus parvus* | Great Basin pocket mouse |
| **Beavers (family Castoridae)** | |
| *Castor canadensis* | beaver |
| **Campagnols, mice, rats, souris, voles (family Muridae)** | |
| *Lemmiscus curtatus* | sagebrush vole |
| *Microtus montanus* | montane vole |
| *Mus musculus* | house mouse |
| *Neotoma cinerea* | bushy-tailed woodrat |
| *Ondatra zibethicus* | muskrat |
| *Onychomys leucogaster* | northern grasshopper mouse |
| *Peromyscus maniculatus* | deer mouse |

**Table C-2 List of Mammals Occurring on the Hanford Site**

| Scientific Name | Common Name |
|---|---|
| *Rattus norvegicus* | Norway rat |
| *Reithrodontomys megalotis* | western harvest mouse |
| **New World porcupines (family Erethizontidae)** | |
| *Erethizon dorsatum* | porcupine |
| **Coyotes, dogs, foxes, jackals, wolves (family Canidae)** | |
| *Canis latrans* | coyote |
| **Raccoons (family Procyonidae)** | |
| *Procyon lotor* | raccoon |
| **Martins, weasels, wolverines, otters, badgers (family Mustelidae)** | |
| *Lontra canadensis* | river otter |
| *Mustela erminea* | short-tail weasel |
| *Mustela frenata* | long-tailed weasel |
| *Mustela vison* | mink |
| *Taxidea taxus* | badger |
| **Skunks (family Mephitidae)** | |
| *Mephitis mephitis* | striped skunk |
| **Cats (family Felidae)** | |
| *Lynx rufus* | bobcat |
| *Puma concolor concolor* | mountain lion |
| **Caribou, cervids, deer, moose, Wapiti (family Cervidae)** | |
| *Cervus elaphus* | Rocky Mountain elk |
| *Odocoileus hemionus* | mule deer |
| *Odocoileus virginianus* | white-tailed deer |

Source: Neitzel DA, ed., Bunn AL, Cannon SD, Duncan JP, Fowler RA, and others. 2005. *Hanford Site National Environmental Policy Act (NEPA) Characterization*, PNL-6415, Rev 17, Pacific Northwest National Laboratory, Richland, Washington.

**Table C-3 Common Birds Occurring on the Hanford Site**

| Scientific Name | Common Name |
|---|---|
| **Gaviiformes - Loons or divers** | |
| *Gavia immer* | common loon |
| **Podicipediformes - Grebes** | |
| *Aechmophorus occidentalis* | western grebe |
| *Podiceps auritus* | horned grebe |
| *Podiceps nigricollis* | eared grebe |
| *Podilymbus podiceps* | pied-billed grebe |
| **Pelecaniformes - Pelicans and allies** | |
| *Pelecanus erythrorhynchos* | American white pelican |
| *Phalacrocorax auritus* | double-crested cormorant |
| **Anseriformes - Waterfowl** | |
| *Anas acuta* | northern pintail |
| *Anas americana* | American wigeon |
| *Anas clypeata* | northern shoveler |
| *Anas crecca* | American green-winged teal |
| *Anas cyanoptera* | cinnamon teal |
| *Anas discors* | blue-winged teal |
| *Anas platyrhynchos* | mallard |
| *Anas strepera* | gadwall |
| *Aythya americana* | redhead |
| *Branta canadensis* | Canada goose |
| *Bucephala albeola* | bufflehead |
| *Bucephala clangula* | common goldeneye |
| *Bucephala islandica* | Barrow's goldeneye |
| *Lophodytes cucullatus* | hooded merganser |
| *Mergus merganser* | common merganser |
| *Oxyura jamaicensis* | ruddy duck |
| **Gruiformes - Cranes, rails, and allies** | |
| *Fulica americana* | American coot |
| *Porzana carolina* | sora |
| *Rallus limicola* | Virginia rail |
| **Charadriiformes - Shorebirds and allies** | |
| *Ardea herodias* | great blue heron |
| *Calidris alpinis* | dunlin |
| *Gallinago gallinago* | common snipe |
| *Larus argentatus* | herring gull |
| *Larus glaucescens* | red-necked phalarope |

**Table C-3 Common Birds Occurring on the Hanford Site**

| Scientific Name | Common Name |
|---|---|
| *Limnodromus scolopaceus* | long-billed dowitcher |
| *Nycticorax nycticorax* | black-crowned night-heron |
| *Tringa flavipes* | lesser yellowlegs |
| *Tringa melanoleuca* | greater yellowlegs |
| *Tringa solitaria* | solitary sandpiper |
| *Actitis macularia* | spotted sandpiper |
| *Calidris mauri* | western sandpiper |
| *Casmerodius albus* | great egret |
| *Charadrius vociferus* | killdeer |
| *Grus canadensis* | sandhill crane |
| *Larus californicus* | California gull |
| *Larus delawarensis* | ring-billed gull |
| *Leucosticte tephrocotis* | glaucous-winged gull |
| *Numenius americanus* | long-billed curlew |
| *Recurvirostra americana* | American avocet |
| *Sterna caspia* | Caspian tern |
| *Sterna forsteri* | Forster's tern |
| **Galliformes - Chicken-like birds** | |
| *Callipepla californica* | California quail |
| *Alectoris chukar* | chukar |
| *Perdix perdix* | grey partridge |
| *Phasianus colchicus* | ring-necked pheasant |
| **Falconiformes - Diurnal birds of prey** | |
| *Accipiter cooperii* | Cooper's hawk |
| *Accipiter striatus* | sharp-shinned hawk |
| *Buteo jamaicensis* | red-tailed hawk |
| *Buteo regalis* | ferruginous hawk |
| *Buteo swainsoni* | Swainson's hawk |
| *Circus cyaneus* | northern harrier |
| *Falco columbarius* | merlin |
| *Pandion haliaetus* | osprey |
| *Aquila chrysaetos* | golden eagle |
| *Buteo lagopus* | northern rough-legged hawk |
| *Falco mexicanus* | prairie falcon |
| *Falco sparverius* | American kestrel |
| *Haliaeetus leucocephalus* | bald eagle |

**Table C-3 Common Birds Occurring on the Hanford Site**

| Scientific Name | Common Name |
|---|---|
| **Strigiformes - Owls** | |
| *Asio flammeus* | short-eared owl |
| *Asio otus* | long-eared owl |
| *Athene cunicularia* | burrowing owl |
| *Bubo virginianus* | great horned owl |
| *Tyto alba* | common barn-owl |
| **Coraciiformes - Rollers and allies** | |
| *Ceryle alcyon* | belted kingfisher |
| **Columbiformes - Pigeons** | |
| *Columba livia* | rock dove |
| *Zenaida macroura* | mourning dove |
| **Caprimulgiformes - Nightjars and allies** | |
| *Chordeiles minor* | common nighthawk |
| *Phalaenoptilus nuttallii* | common poorwill |
| **Apodiformes - Hummingbirds, swifts** | |
| *Selasphorus rufus* | rufous hummingbird |
| **Piciformes - Woodpeckers and allies** | |
| *Colaptes auratus* | Northern flicker |
| **Passeriformes - Perching birds** | |
| *Agelaius phoeniceus* | red-winged blackbird |
| *Ammodramus savannarum* | grasshopper sparrow |
| *Amphispiza belli* | sage sparrow |
| *Bombycilla cedrorum* | cedar waxwing |
| *Carduelis tristis* | American goldfinch |
| *Carpodacus mexicanus* | house finch |
| *Catherpes mexicanus* | canyon wren |
| *Chondestes grammacus* | lark sparrow |
| *Cistothorus palustris* | marsh wren |
| *Contopus sordidulus* | western wood-pewee |
| *Corvus brachyrhynchos* | American crow |
| *Corvus corax* | common raven |
| *Dendroica coronata* | yellow-rumped warbler |
| *Dendroica petechia* | yellow warbler |
| *Dendroica townsendi* | Townsend's warbler |
| *Empidonax difficilis* | Pacific-slope flycatcher |
| Empidonax hammondii | Hammond's flycatcher |
| *Eremophila alpestris* | horned lark |

**Table C-3 Common Birds Occurring on the Hanford Site**

| Scientific Name | Common Name |
|---|---|
| *Euphagus cyanocephalus* | Brewer's blackbird |
| *Hirundo pyrrhonota* | cliff swallow |
| *Hirundo rustica* | barn swallow |
| *Icteria virens* | yellow-breasted chat |
| *Icterus galbula* | Bullock's oriole |
| *Ixoreus naevius* | varied thrush |
| *Junco hyemalis* | dark-eyed junco |
| *Lanius ludovicianus* | loggerhead shrike |
| *Melospiza lincolnii* | Lincoln's sparrow |
| *Melospiza melodia* | song sparrow |
| *Molothrus ater* | brown-headed cowbird |
| *Myadestes townsendi* | Townsend's solitaire |
| *Oporornis tolmiei* | MacGillivray's warbler |
| *Oreoscoptes montanus* | sage thrasher |
| *Passer domesticus* | house sparrow |
| *Passerculus sandwichensis* | savannah sparrow |
| *Passerina amoena* | lazuli bunting |
| *Phalaropus lobatus* | gray-crowned rosy finch |
| *Pheucticus melanocephalus* | black-headed grosbeak |
| *Pica pica* | black-billed magpie |
| *Pipilo erythrophthalmus* | rufous-sided towhee |
| *Piranga ludoviciana* | western tanager |
| *Pooecetes gramineus* | vesper sparrow |
| *Regulus calendula* | ruby-crowned kinglet |
| *Regulus satrapa* | golden-crowned kinglet |
| *Riparia riparia* | bank swallow |
| *Salpinctes obsoletus* | rock wren |
| *Sayornis saya* | Say's phoebe |
| *Sitta canadensis* | red-breasted nuthatch |
| *Spizella breweri* | Brewer's sparrow |
| *Spizella passerina* | chipping sparrow |
| *Stelgidopteryx serripennis* | northern rough-winged swallow |
| *Sturnella neglecta* | western meadowlark |
| *Sturnus vulgaris* | European starling |
| *Tachycineta bicolor* | tree swallow |
| *Tachycineta thalassina* | violet-green swallow |

**Table C-3 Common Birds Occurring on the Hanford Site**

| Scientific Name | Common Name |
|---|---|
| *Thryomanes bewickii* | Bewick's wren |
| *Troglodytes aedon* | house wren |
| *Troglodytes troglodytes* | winter wren |
| *Turdus migratorius* | American robin |
| *Tyrannus tyrannus* | eastern kingbird |
| *Tyrannus verticalis* | western kingbird |
| *Vermivora celata* | orange-crowned warbler |
| *Vermivora ruficapilla* | Nashville warbler |
| *Vireo gilvus* | warbling vireo |
| *Vireo solitarius* | Blue-headed vireo |
| *Wilsonia pusilla* | Wilson's warbler |
| *Xanthocephalus xanthocephalus* | yellow-headed blackbird |
| *Zonotrichia atricapilla* | golden-crowned sparrow |
| *Zonotrichia leucophrys* | white-crowned sparrow |

Source: Neitzel DA, ed., Bunn AL, Cannon SD, Duncan JP, Fowler RA, and others. 2005. *Hanford Site National Environmental Policy Act (NEPA) Characterization*, PNL-6415, Rev 17, Pacific Northwest National Laboratory, Richland, Washington.

**Table C-4 Amphibians and Reptiles Occurring on the Hanford Site**

| Scientific Name | Common Name |
|---|---|
| **Amphibians** | |
| *Ambystoma tigrinum* | tiger Salamander |
| *Bufo boreas* | western toad |
| *Bufo woodhousii* | Woodhouse's toad |
| *Hila regilla* | Pacific tree frog |
| *Rana catesbeiana* | bullfrog |
| *Scaphiopus intermontanus* | Great Basin spadefoot |
| **Reptiles** | |
| *Chrysemys picta* | painted turtle |
| *Coluber constrictor* | western yellow-bellied racer |
| *Crotalus viridis* | western rattlesnake |
| *Hypsiglena torquata* | night snake |
| *Masticophis taeniatus* | striped whipsnake |
| *Phrynosoma douglassii* | short-horned lizard |
| *Pituiphis melanoleucus* | Great Basin gopher snake |
| *Scleroporus graciosus* | northern sagebrush lizard |
| *Thamnophis sirtalis* | common garter snake |
| *Uta stansburiana* | side-blotched lizard |

Source: Neitzel DA, ed., Bunn AL, Cannon SD, Duncan JP, Fowler RA, and others. 2005. *Hanford Site National Environmental Policy Act (NEPA) Characterization*, PNL-6415, Rev 17, Pacific Northwest National Laboratory, Richland, Washington.

**Table C-5 Fish Species Occurring in the Hanford Reach**

| Scientific Name | Common Name |
|---|---|
| **Paddlefishes, spoonfishes, sturgeons (family Acipenseridae)** | |
| *Acipenser transmontanus* | white sturgeon |
| **Anchovies, herrings (family Clupeidae)** | |
| *Alosa sapidissima* | American shad |
| **Cyprins, minnows, suckers (family Catostomidae)** | |
| *Acrocheilus alutaceus* | chiselmouth |
| *Catostomus columbianus* | bridgelip sucker |
| *Catostomus macrocheilus* | largescale sucker |
| *Catostomus platyrhynchus* | mountain sucker |
| *Cyprinus carpio* | common carp |
| *Mylocheilus caurinus* | peamouth |
| *Ptychocheilus oregonensis* | northern pikeminnow |
| *Rhinichthys cataractae* | longnose dace |
| *Rhinichthys falcatus* | leopard dace |
| *Rhinichthys osculus* | speckled dace |
| *Richardsonius balteatus* | redside shiner |
| **Livebearers (family Poeciliidae)** | |
| *Gambusia affinis* | Western mosquitofish |
| **Cods (family Gadidae)** | |
| *Lota lota* | burbot |
| **Pipefishes, sticklebacks (family Gasterosteidae)** | |
| *Gasterosteus aculeatus* | threespine stickleback |
| *Pungitius pungitius* | ninespine stickleback |
| **Anchovies, herrings (family Clupeidae)** | |
| *Alosa sapidissima* | American shad |
| **Cyprins, minnows, suckers (family Catostomidae)** | |
| *Acrocheilus alutaceus* | chiselmouth |
| *Catostomus columbianus* | bridgelip sucker |
| *Catostomus macrocheilus* | largescale sucker |
| *Catostomus platyrhynchus* | mountain sucker |
| *Cyprinus carpio* | common carp |
| *Mylocheilus caurinus* | peamouth |
| *Ptychocheilus oregonensis* | northern pikeminnow |
| *Rhinichthys cataractae* | longnose dace |
| *Rhinichthys falcatus* | leopard dace |

**Table C-5 Fish Species Occurring in the Hanford Reach**

| Scientific Name | Common Name |
|---|---|
| *Rhinichthys osculus* | speckled dace |
| *Richardsonius balteatus* | redside shiner |
| **Livebearers (family Poeciliidae)** | |
| *Gambusia affinis* | Western mosquitofish |
| **Cods (family Gadidae)** | |
| *Lota lota* | burbot |
| **Pipefishes, sticklebacks (family Gasterosteidae)** | |
| *Gasterosteus aculeatus* | threespine stickleback |
| *Pungitius pungitius* | ninespine stickleback |
| **Perch-like fishes (family Centrarchidae)** | |
| *Lepomis gibbosus* | pumpkinseed |
| *Lepomis macrochirus* | bluegill |
| *Micropterus dolomieui* | smallmouth bass |
| *Micropterus salmoides* | largemouth bass |
| *Perca flavenscens* | yellow perch |
| *Pomoxis annularis* | white crappie |
| *Pomoxis nigromaculatus* | black crappie |
| *Sander vitreus* | walleye |
| **Trout perches (family Perocpsidae)** | |
| *Percopsis transmontana* | sand roller |
| **Lampreys (family Petromyzontidae)** | |
| *Lampetra ayresii* | river lamprey |
| *Lampetra tridentata* | Pacific lamprey |
| **Salmonids, salmons, trouts (family Salmonidae)** | |
| *Coregonus clupeaformis* | lake whitefish |
| *Oncorhynchus clarkii* | cutthroat trout |
| *Oncorhynchus kisutch* | coho salmon |
| *Oncorhynchus mykiss* | rainbow trout (steelhead) |
| *Oncorhynchus nerka* | sockeye salmon |
| *Oncorhynchus tshawytscha* | Chinook salmon |
| *Prosopium williamsoni* | mountain whitefish |
| *Salvelinus confluentus* | bull trout |
| *Salvelinus malma* | Dolly Varden |
| **Chabots, sculpins (family Cottidae)** | |
| *Cottus asper* | prickley sculpin |
| *Cottus bairdii* | mottled sculpin |

**Table C-5 Fish Species Occurring in the Hanford Reach**

| Scientific Name | Common Name |
|---|---|
| *Cottus beldingii* | Piute sculpin |
| *Cottus perplexus* | reticulate sculpin |
| *Cottus rhotheus* | torrent sculpin |
| **Bullhead catfishes, North American freshwater catfishes (family Ictaluridae)** | |
| *Ameiurus melas* | black bullhead |
| *Ameiurus natalis* | yellow bullhead |
| *Ameiurus nebulosus* | brown bullhead |
| *Ictalurus punctatus* | channel catfish |

Source: Neitzel DA, ed., Bunn AL, Cannon SD, Duncan JP, Fowler RA, and others. 2005. *Hanford Site National Environmental Policy Act (NEPA) Characterization*, PNL-6415, Rev 17, Pacific Northwest National Laboratory, Richland, Washington.

**Table C-6 Plant Species of Concern on the Hanford Site**

| Scientific Name | Common Name | Status | |
|---|---|---|---|
| | | Federal | State |
| *Aliciella (= Gilia) leptomeria* | Great Basin gilia | | T |
| *Allium robinsonii* | Robinson's onion | | W |
| *Allium scilloides* | scilla onion | | W |
| *Ammannia robusta* | grand redstem | | T |
| *Anagallis (= Centunculus)minimus* | chaffweed | | T |
| *Artemisia lindleyana* | Columbia River mugwort | | W |
| *Astragalus columbianus* | Columbia milkvetch | SC | S |
| *Astragalus conjunctus var. rickardii* | basalt milkvetch | | W |
| *Astragalus geyeri* | Geyer's milkvetch | | T |
| *Astragalus sclerocarpus* | stalked-pod milkvetch | | W |
| *Astragalus speirocarpus* | medic milkvetch | | W |
| *Astragalus succumbens* | crouching milkvetch | | W |
| *Balsamorhiza rosea* | rosy balsamroot | | W |
| *Camissonia (= Oenothera) minor* | small-flowered evening-primrose | | S |
| *Camissonia (= Oenothera) pygmaea* | dwarf evening primrose | | S |
| *Carex hystericina* | porcupine sedge | | W |
| *Castilleja exilis* | annual paintbrush | | W |
| *Cistanthe (= Calyptridium) roseum* | rosy pussypaws | | T |
| *Crassula aquatica* | pigmy-weed | | W |
| *Cryptantha leucophaea* | gray cryptantha | SC | S |
| *Cryptantha scoparia* | desert cryptantha | | S |
| *Cryptantha spiculifera (= C. interrupta)* | Snake River cryptantha | | S |
| *Cuscuta denticulata* | desert dodder | | T |
| *Cyperus bipartitus (rivularis)* | shining flatsedge | | W |
| *Delphinium multiplex* | Kittitas larkspur | | W |
| *Eatonella nivea* | white eatonella | | T |
| *Eleocharis rostellata* | beaked spike-rush | | S |
| *Epipactis gigantea* | giant helleborine | | W |
| *Erigeron piperianus* | Piper's daisy | | S |
| *Eriogonum codium* | Umtanum desert buckwheat | C | E |
| *Hierchloe odorata =(Anthoxanthm hirtum)* | vanilla grass | | R1 |
| *Hypericum majus* | Canadian St. John's wort | | S |
| *Limosella acaulis* | southern mudwort | | W |
| *Lindernia dubia var. anagallidea* | false pimpernel | | W |
| *Lipocarpha (= Hemicarpha) aristulata* | awned halfchaff sedge | | T |

## Table C-6 Plant Species of Concern on the Hanford Site

| Scientific Name | Common Name | Status | |
|---|---|---|---|
| | | Federal | State |
| *Loeflingia squarrosa var. squarrosa* | loeflingia | | T |
| *Lomatium tuberosum* | Hoover's desert parsley | SC | S |
| *Mimulus suksdorfii* | Suksdorf's monkey flower | | S |
| *Minuartia pusilla var. pusilla* | annual sandwort | | R1 |
| *Nama densum var. parviflorum* | small-flowered nama | | W |
| *Nicotiana attenuata* | coyote tobacco | | S |
| *Oenothera caespitosa* | desert evening-primrose | | S |
| *Pectocarya penicillata* | winged combseed | | W |
| *Pectocarya setosa* | bristly combseed | | W |
| *Pediocactus simpsonii var. robustior =(P. nigrispinus)* | hedgehog cactus | | R1 |
| *Pellaea glabella simplex* | smooth cliffbrake | | W |
| *Penstemon eriantherus whitedii* | fuzzytongue penstemon | | S |
| *Physaria (= Lesquerella) tuplashensis* | White Bluffs bladderpod | C | T |
| *Rorippa columbiae* | Columbia yellowcress | SC | E |
| *Rotala ramosior* | lowland toothcup | | T |

**Federal Definitions**
(50 CFR 17, Endangered and Threatened Wildlife and Plants, as amended)

Federal Status under the U.S. Endangered Species Act (USESA) as published in the Federal Register:

C = Candidate species. Sufficient information exists to support listing as Endangered or Threatened.

SC = Species of Concern. An unofficial status, the species appears to be in jeopardy, but insufficient information to support listing.

**State Definitions**
(WSDNR. 2011. Washington Dept. of Natural Resources, Washington Natural Heritage Program, Olympia, WA)

State Status of plant species is determined by the Washington Natural Heritage Program. Factors considered include abundance, occurrence patterns, vulnerability, threats, existing protection, and taxonomic distinctness. Values include:

E = Endangered. In danger of becoming extinct or extirpated from Washington.

T = Threatened. Likely to become Endangered in Washington.

S = Sensitive. Vulnerable or declining and could become Endangered or Threatened in the state.

R1 = Review group 1. Of potential concern but needs more field work to assign another rank.

W = Watch. More abundant and/or less threatened than previously thought.

Source: PNNL. 2010. *Hanford Site Environmental Report for Calendar Year 2009*, PNNL 19455, Pacific Northwest National Laboratory, Richland, Washington.

**Table C-7 Wildlife Species of Concern on the Hanford Site**

| Scientific Name | Common Name | Status | |
|---|---|---|---|
| | | Federal | State |
| **Mammals** | | | |
| *Antrozous pallidus* | pallid bat | | W |
| *Lagurus curtatus* | sagebrush vole | | W |
| *Lepus californicus* | black-tailed jackrabbit | | SC |
| *Lepus townsendii* | white-tailed jackrabbit | | SC |
| *Myotis leibii* | small-footed myotis | | W |
| *Myotis volans* | long-legged myotis | | W |
| *Onychomys leucogaster* | northern grasshopper mouse | | W |
| *Pipistrellus hesperus* | western pipistrelle | | W |
| *Sorex merriami* | Merriam's shrew | | SC |
| *Spermophilus townsendii* | Townsend's ground squirrel | FCo | SC |
| *Spermophilus washingtoni* | Washington ground squirrel[(a)] | FC | SC |
| *Taxidea taxus* | badger | | W |
| **Birds** | | | |
| *Accipter gentilis* | northern goshawk[(a)] | FCo | SC |
| *Aechmophorus clarkii* | Clark's grebe | | W |
| *Aechmophorus occidentalis* | western grebe | | SC |
| *Ammodramus savannarum* | grasshopper sparrow | | W |
| *Amphispiza belli* | sage sparrow | | SC |
| *Aquila chrysaetos* | golden eagle | | SC |
| *Ardea alba* | great egret | | W |
| *Ardea herodias* | great blue heron | | W |
| *Athene cunicularia* | burrowing owl | FCo | SC |
| *Buteo regalis* | ferruginous hawk | FCo | ST |
| *Buteo swainsoni* | Swainson's hawk | | W |
| *Carduelis psaltria* | lesser goldfinch | | W |
| *Cathartes aura* | turkey vulture[(b)] | | W |
| *Centrocercus urophasianus* | greater sage grouse | FC | ST |
| *Chlidonias niger* | black tern[(b)] | | W |
| *Contopus cooperi* | olive-sided flycatcher | FCo | |
| *Dolichonyx oryzivorus* | bobolink[(b)] | | W |
| *Empidonax wrightii* | gray flycatcher | | W |
| *Falco columbarius* | merlin | | SC |
| *Falco mexicanus* | prairie falcon | | W |

**Table C-7 Wildlife Species of Concern on the Hanford Site**

| Scientific Name | Common Name | Status | |
|---|---|---|---|
| | | Federal | State |
| *Falco peregrinus* | peregrine falcon | FCo | SS |
| *Falco rusticolus* | gyrfalcon[(b)] | | W |
| *Gavia immer* | common loon | | SS |
| *Grus canadensis* | sandhill crane | | SE |
| *Haliaeetus leucocephalus* | bald eagle[(c)] | FCo | SS |
| *Himantopus mexicanus* | black-necked stilt | | W |
| *Lanius ludovicianus* | loggerhead shrike | FCo | SC |
| *Melanerpes lewis* | Lewis's woodpecker[(a)] | | SC |
| *Myiarchus cinerascens* | ash-throated flycatcher[(b)] | | W |
| *Numenius americanus* | long-billed curlew | | W |
| *Nyctea scandiaca* | snowy owl | | W |
| *Nycticorax nycticorax* | black-crowned night-heron | | W |
| *Oreoscoptes montanus* | sage thrasher | | SC |
| *Otus flammeolus* | flamulated owl[(a)] | | SC |
| *Pandion haliaetus* | osprey | | W |
| *Pelecanus erythrorhynchos* | American white pelican | | SE |
| *Podiceps auritus* | horned grebe | | W |
| *Podiceps grisegena* | red-necked grebe[(b)] | | W |
| *Sialia mexicana* | western bluebird | | W |
| *Sterna caspia* | Caspian tern | | W |
| *Sterna forsteri* | Forster's tern | | W |
| *Sterna paradisaea* | Arctic tern[(b)] | | W |
| **Amphibians and Reptiles** | | | |
| *Bufo boreas* | western toad | FCo | SC |
| *Bufo woodhousii* | Woodhouse's toad | | W |
| *Hypsiglena torquata* | night snake | | W |
| *Masticophis taeniatus* | striped whipsnake | | SC |
| *Phrynosoma douglassii* | short-horned lizard | | W |
| *Sceloporus graciosus* | sagebrush lizard | FCo | SC |
| **Fish** | | | |
| *Catastomus platyrhynchus* | mountain sucker[(a)] | | SC |
| *Cottus beldingi* | piute sculpin | | W |
| *Cottus perplexus* | reticulate sculpin | | W |
| *Lampetra ayresi* | river lamprey[(a)] | FCo | SC |

**Table C-7 Wildlife Species of Concern on the Hanford Site**

| Scientific Name | Common Name | Status | |
|---|---|---|---|
| | | Federal | State |
| *Lampetra tridentata* | Pacific lamprey | FCo | W |
| *Oncorhynchus mykiss* | steelhead | FT | SC |
| *Oncorhynchus tshawytscha* | spring-run Chinook salmon | FE | SC |
| *Percopsis transmontana* | sand roller | | W |
| *Rhinichthys flacatus* | leopard dace[(a)] | | SC |
| *Salvelinus confluentus* | bull trout[(a)] | FT | SC |
| **Mollusks** | | | |
| *Anodonta californiensis* | California floater | FCo | SC |
| *Anodonta kennerlyi* | western floater | | W |
| *Anodonta oregonensis* | Oregon floater | | W |
| *Fisherola nuttalli* | shortfaced lanx | | SC |
| *Fluminicola columbiana* | great Columbia River spire snail | FCo | SC |
| *Margaritifera falcata* | western pearlshell | | W |
| **Insects** | | | |
| *Boloria selene atrocostalis* | silver-bordered fritillary | | SC |
| *Callophrys sheridanii neoperplexa* | canyon green hairstreak | | W |
| *Chlosyne palla palla* | northern checkerspot | | W |
| *Cicindela columbica* | Columbia River tiger beetle[(a)] | | SC |
| *Epargyreus clarus californicus* | silver-spotted skipper | | W |
| *Erynnis persius* | Persius' duskywing | | W |
| *Harkenclenus titus immaculosus* | coral hairstreak | | W |
| *Hesperia juba* | juba skipper | | W |
| *Hesperia nevada* | Nevada skipper | | W |
| *Limenitis archippus lahontani* | viceroy | | W |
| *Lycaena helloides* | purplish copper | | W |
| *Lycaena rubida perkinsorum* | ruddy copper | | W |
| *Ochlodes sylvanoides bonnevilla* | Bonneville skipper | | W |
| *Phyciodes cocyta pascoensis* | Pasco pearl | | W |

## Table C-7 Wildlife Species of Concern on the Hanford Site

| Scientific Name | Common Name | Status | |
|---|---|---|---|
| | | Federal | State |

**Federal Definitions**
(from *Endangered Species Act*, Public Law 93-205, as amended).

FC = Federal candidate: A species that that is identified for listing as a federally protected species under the Endangered Species Act. These species are not currently under federal protection but are being considered for inclusion if the scientific data supports listing.

FE = Federal endangered: A species in danger of extinction or extirpation throughout all or a substantial portion of its range.

FT = Federal threatened: A species that is likely to become endangered within the near future because of threats to its population.

FCo = Species of Concern: species about which there are concerns regarding status and threats, but for which insufficient information is available to indicate a need to list the species under the Endangered Species Act.

**State Definitions**
(WSDNR. 2011. Washington Dept. of Natural Resources, Washington Natural Heritage Program, Olympia, WA)
Defined in WAC 232-12-297.

SC = State candidate: A wildlife species native to Washington State that the Department of Wildlife (WDFW) will review for possible listing as endangered, threatened, or sensitive. Candidate species are defined in WDFW Policy M-6001.

SE = State endangered: A species native to Washington State that is seriously threatened with extinction throughout all or a substantial portion of its range within the state. Endangered species are designated in WAC 232-12-014.

SS = State sensitive: A wildlife species native to the state of Washington that is vulnerable or declining and is likely to become endangered or threatened throughout a significant portion of its range within the state without cooperative management or removal of threats. Sensitive species are designated in WAC 232-12-011.

ST = State threatened: A species native to Washington State likely to become endangered within the foreseeable future throughout substantial portions of its range within the state without cooperative management or the removal of threats. Threatened species are designated in WAC 232-12-011.

W = Watch list species: Taxa that are more abundant and/or less threatened than previously assumed.

**Notes:**
[a] Probable, but not observed, on the Hanford Site.
[b] Reported, but seldom observed on the Hanford Site.
[c] Removed from the list of threatened wildlife effective August 8, 2007 (72 FR 37346).

Source: PNNL. 2010. *Hanford Site Environmental Report for Calendar Year 2009*, PNNL 19455, Pacific Northwest National Laboratory, Richland, Washington.

# Supplement 1

# Constituents of Potential Concern for the WTP Air and Dangerous Waste Permits (24590-WTP-RPT-ENV-10-001)

Provided under separate cover.

# Supplement 2

# Integrated Emissions Baseline Report for the Hanford Tank Waste Treatment and Immobilization Plant (24590-WTP-RPT-PO-03-008)

Provided under separate cover.

# Supplement 3

# Estimated Organic Emissions from Process Cells (24590-WTP-HAC-50-00001)

Provided under separate cover.

# Supplement 4

# Chemical Parameters and Toxicological Inputs for the Environmental Risk Assessment for the Hanford Tank Waste Treatment and Immobilization Plant (24590-WTP-RPT-ENV-14-003)

Provided under separate cover.

# Supplement 5

# Hanford Tank Waste Treatment and Immobilization Plant Risk Assessment Air Quality Modeling Protocol (24590-WTP-RPT-ENV-08-001)

Provided under separate cover.