

AR TARGET SHEET

The following document was too large to scan as one unit, therefore, it has been divided into sections.

EDMC#: 0073124
SECTION: 2 OF 3

DOCUMENT #: Letter: 07-AMRC-0224
Document: DOE/RL-2007-21
Draft A

TITLE: Risk Assessment Report for 100
Area and 300 Area Component of
River Corridor Baseline Risk
Assessment (RCBRA)

Figure 4-74. Box Plot of Uranium-235 in Water (pCi/L) by RCBRA Media.

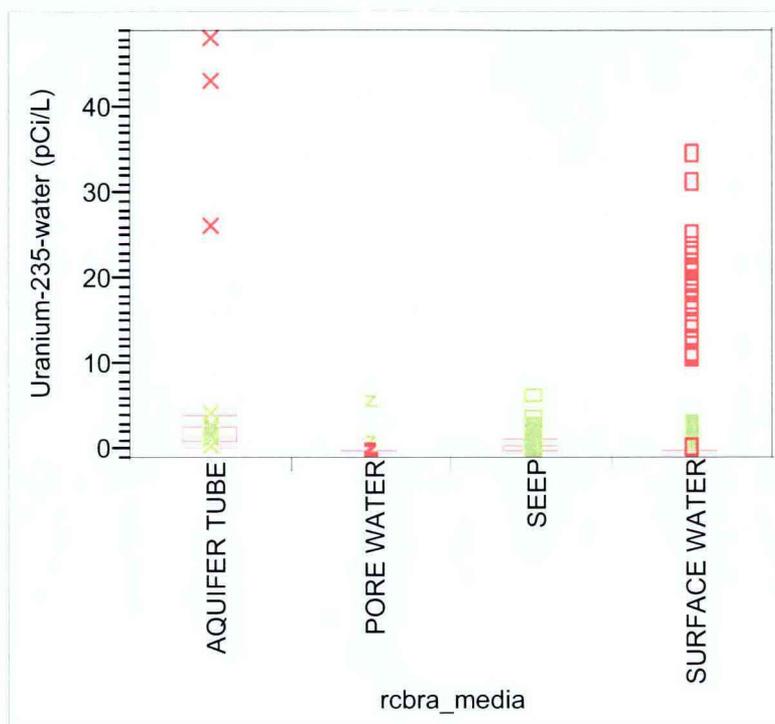


Figure 4-75. Concentrations of Uranium-235 in Sediment (pCi/g) by Hanford River Mile.

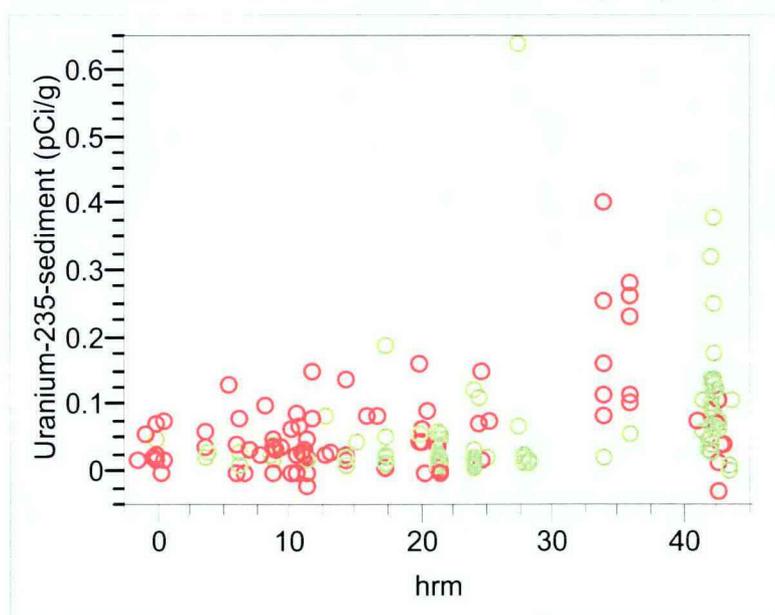


Figure 4-76. Concentrations of Uranium-235 in Sediment (pCi/g) by Date.

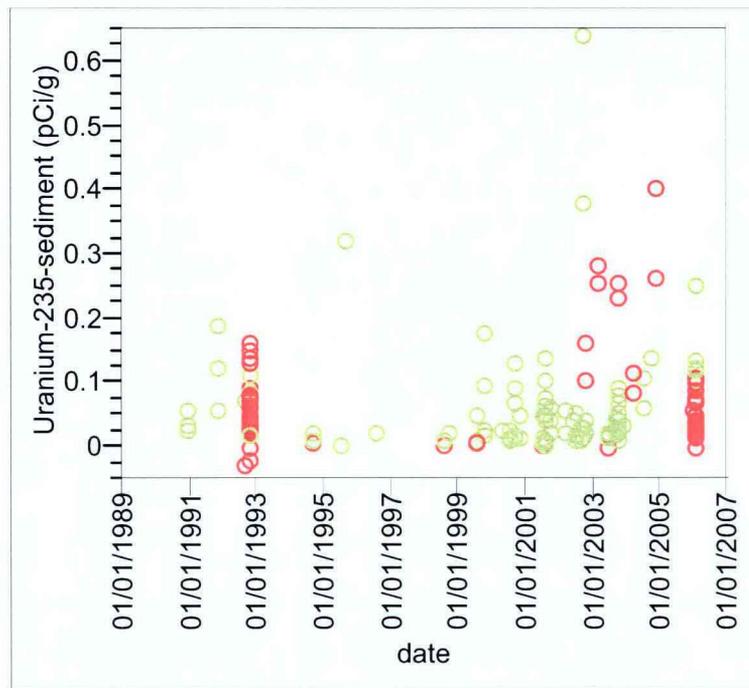
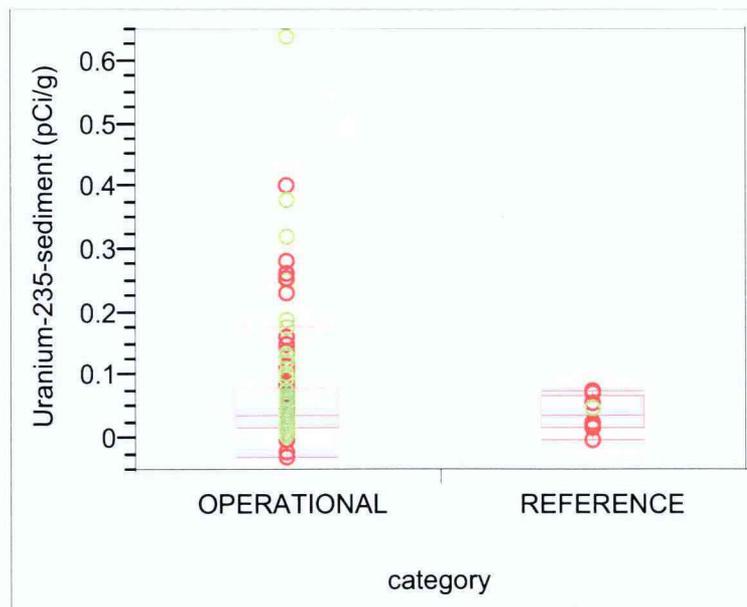


Figure 4-77. Box Plot of Uranium-235 in Sediment (pCi/g) by Category.



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ✦ MUSSEL | ▪ FALSE |
| + AQUATIC VEGETATION | z PORE WATER | • TRUE |
| x AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ◇ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

Figure 4-78. Box Plot of Uranium-235 in Sediment (pCi/g) by RCBRA Media.

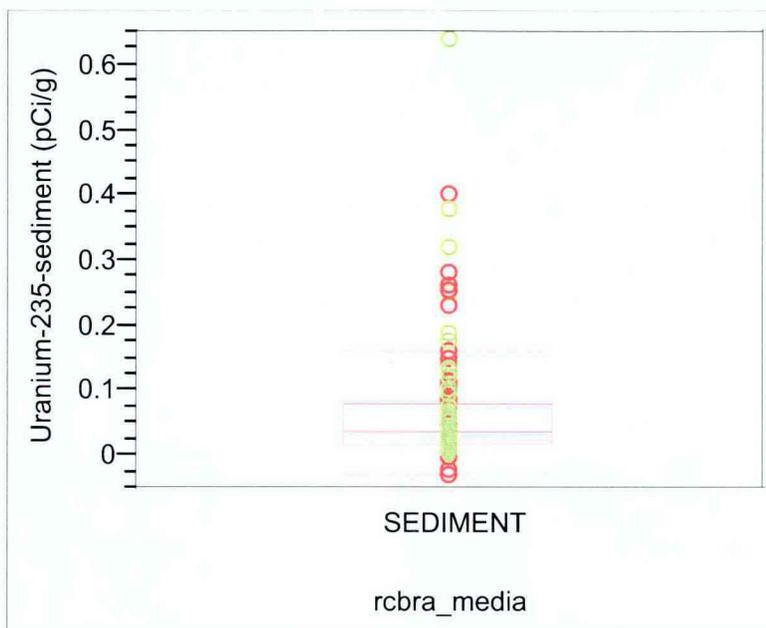
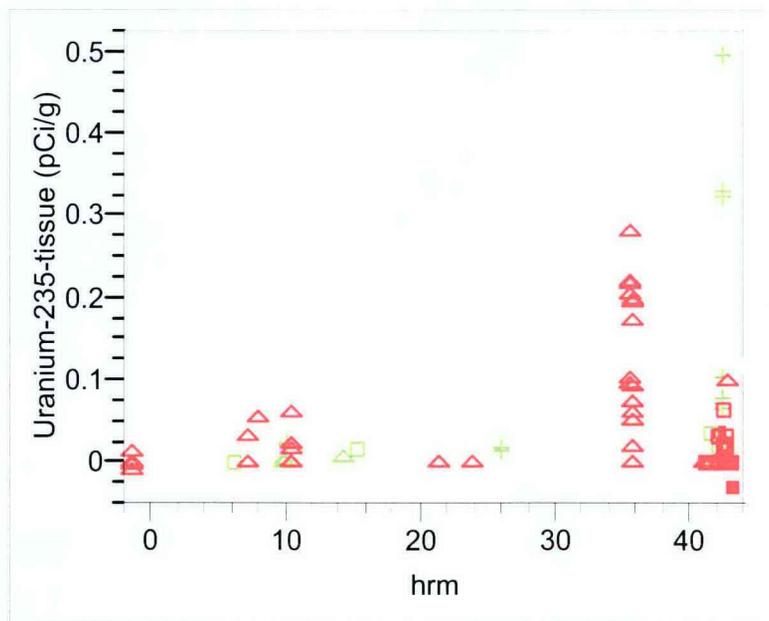


Figure 4-79. Concentrations of Uranium-235 in Tissue (pCi/g) by Hanford River Mile.



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ⋈ MUSSEL | ▪ FALSE |
| ⊕ AQUATIC VEGETATION | ⊚ PORE WATER | • TRUE |
| × AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ⋄ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

Figure 4-80. Concentrations of Uranium-235 in Tissue (pCi/g) by Date.

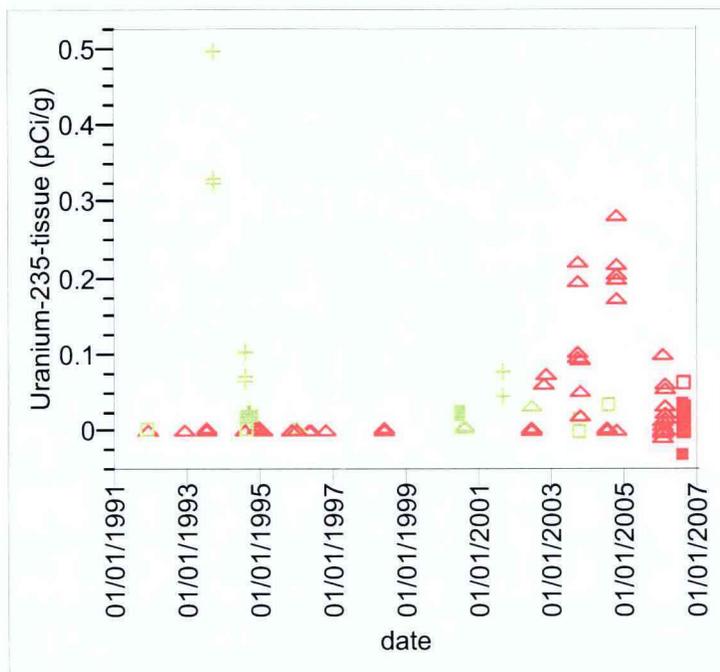
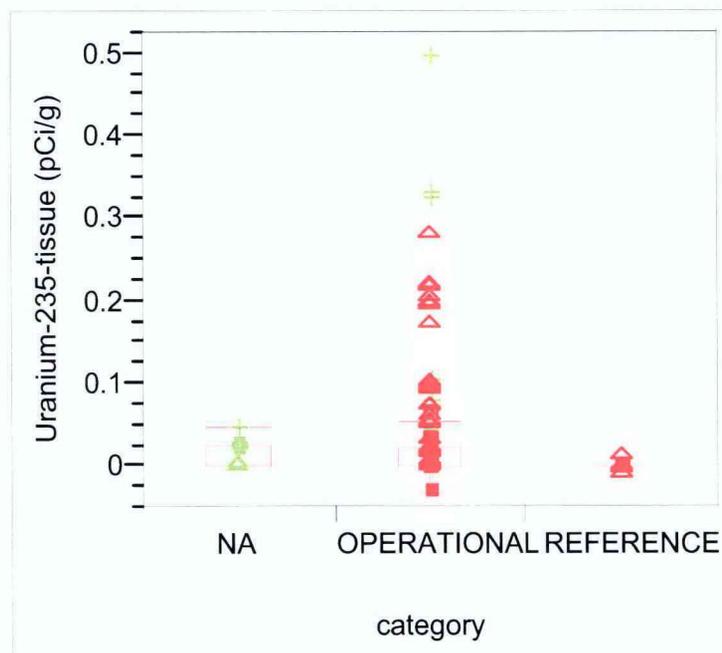
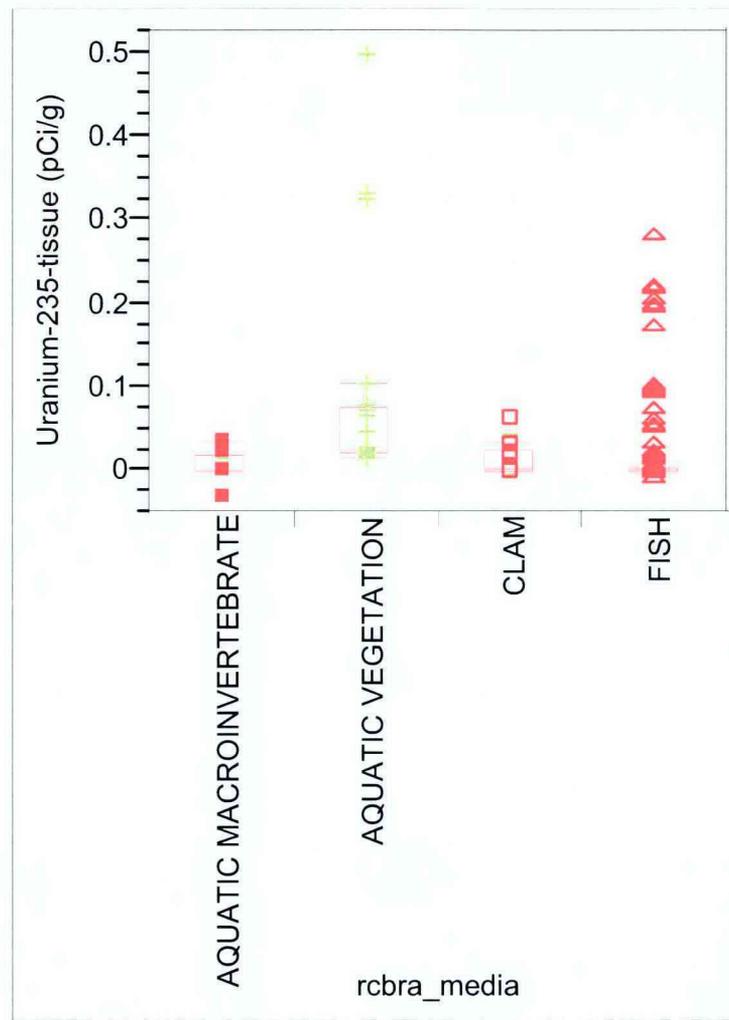


Figure 4-81. Box Plot of Uranium-235 in Tissue (pCi/g) by Category.



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ⋄ MUSSEL | ▪ FALSE |
| + AQUATIC VEGETATION | ⋄ PORE WATER | • TRUE |
| × AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ⋄ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

Figure 4-82. Box Plot of Uranium-235 in Tissue (pCi/g) by RCBRA Media.



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ✕ MUSSEL | ▪ FALSE |
| + AQUATIC VEGETATION | z PORE WATER | • TRUE |
| × AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ◊ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

Figure 4-83. Concentrations of Uranium-238 in Water (pCi/L) by Hanford River Mile.

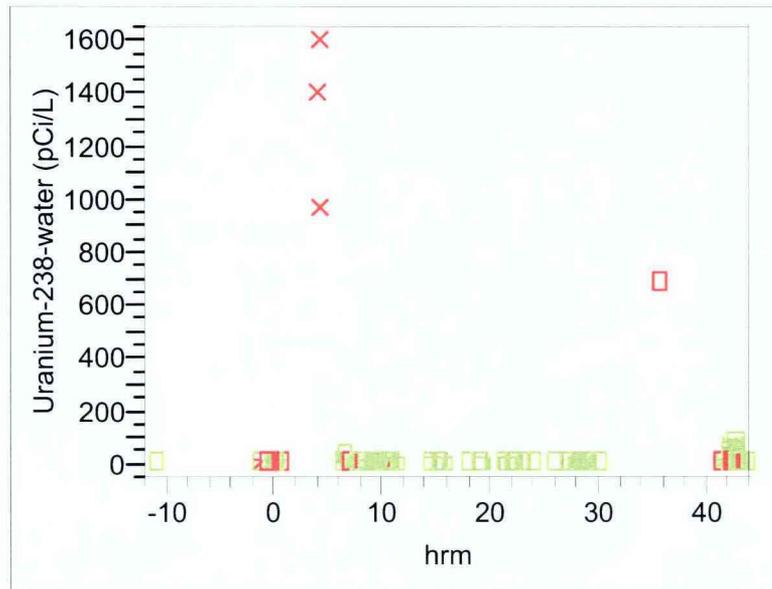
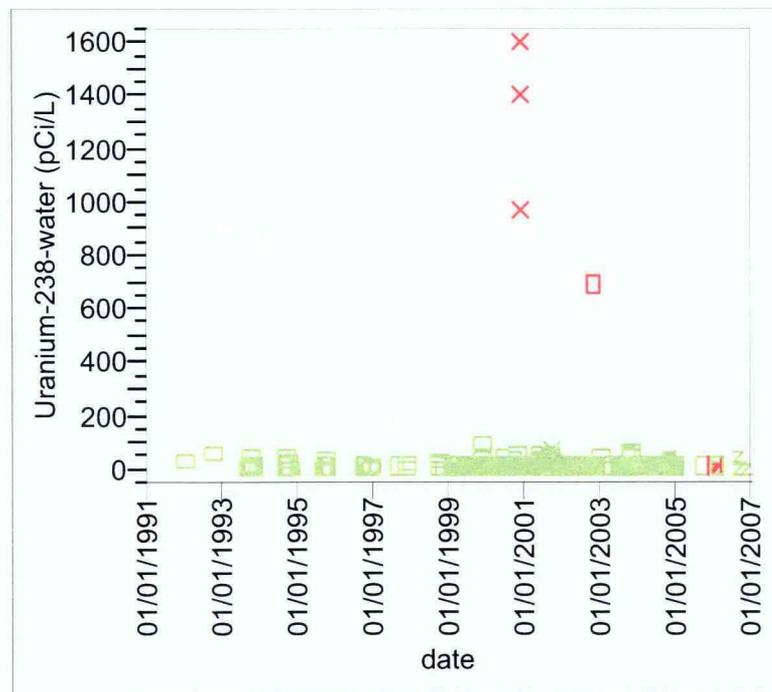
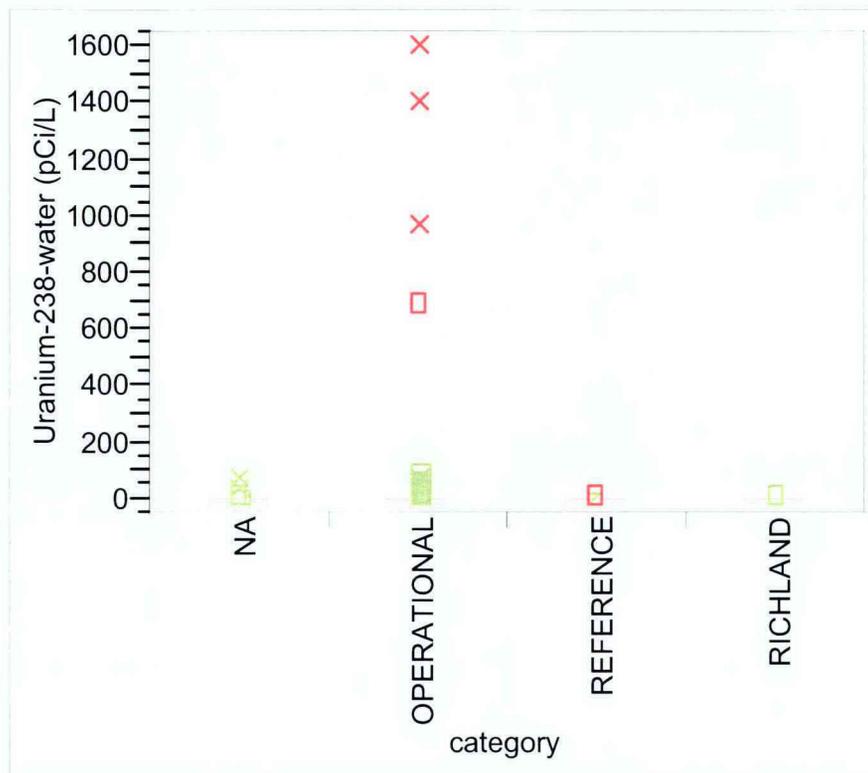


Figure 4-84. Concentrations of Uranium-238 in Water (pCi/L) by Date.



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ⋈ MUSSEL | ▪ FALSE |
| + AQUATIC VEGETATION | z PORE WATER | • TRUE |
| × AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ◊ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

Figure 4-85. Box Plot of Uranium-238 in Water (pCi/L) by Category.



rcbra_media

- AQUATIC MACROINVERTEBRATE
- + AQUATIC VEGETATION
- x AQUIFER TUBE
- ▣ CLAM
- ◊ CRAYFISH
- △ FISH

- ▼ MUSSEL
- z PORE WATER
- SEDIMENT
- ▣ SEEP
- ▣ SURFACE WATER

detect_status

- FALSE
- TRUE

Figure 4-86. Box Plot of Uranium-238 in Water (pCi/L) by RCBRA Media.

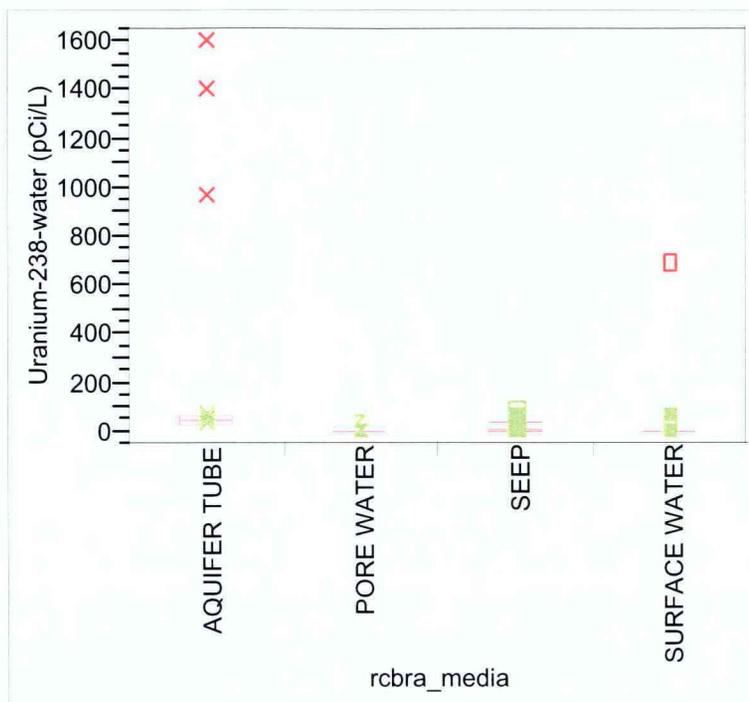


Figure 4-87. Concentrations of Uranium-238 in Sediment (pCi/g) by Hanford River Mile.

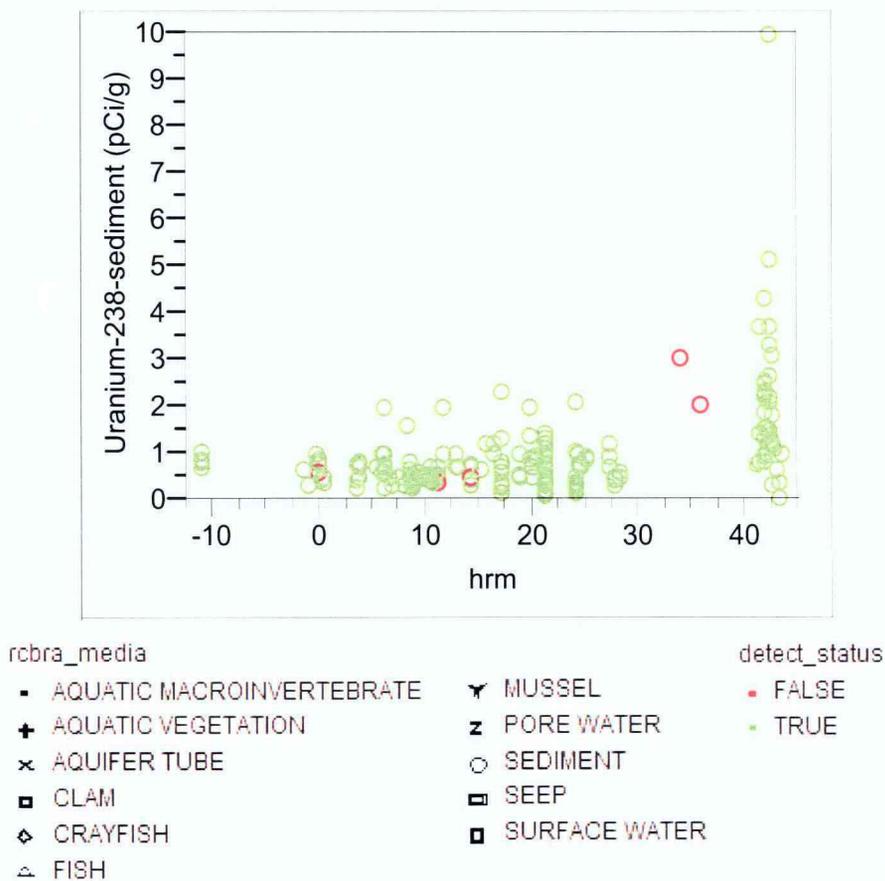


Figure 4-88. Concentrations of Uranium-238 in Sediment (pCi/g) by Date.

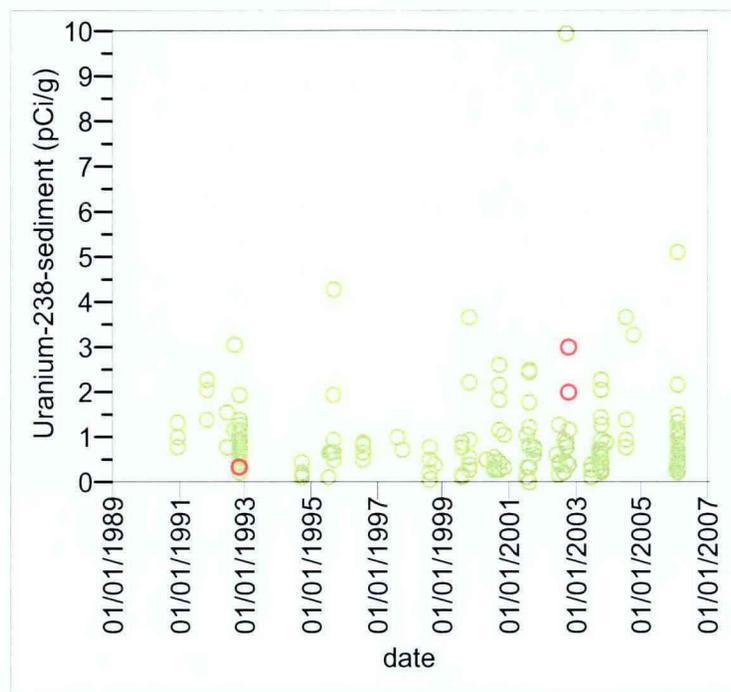
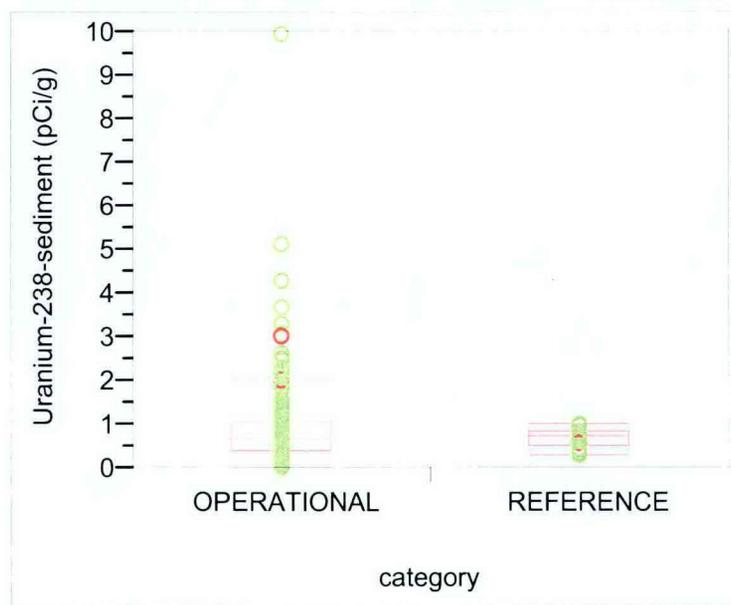


Figure 4-89. Box Plot of Uranium-238 in Sediment (pCi/g) by Category.



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ⚓ MUSSEL | ▪ FALSE |
| + AQUATIC VEGETATION | ⚓ PORE WATER | • TRUE |
| × AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ◊ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

Figure 4-90. Box Plot of Uranium-238 in Sediment (pCi/g) by RCBRA Media.

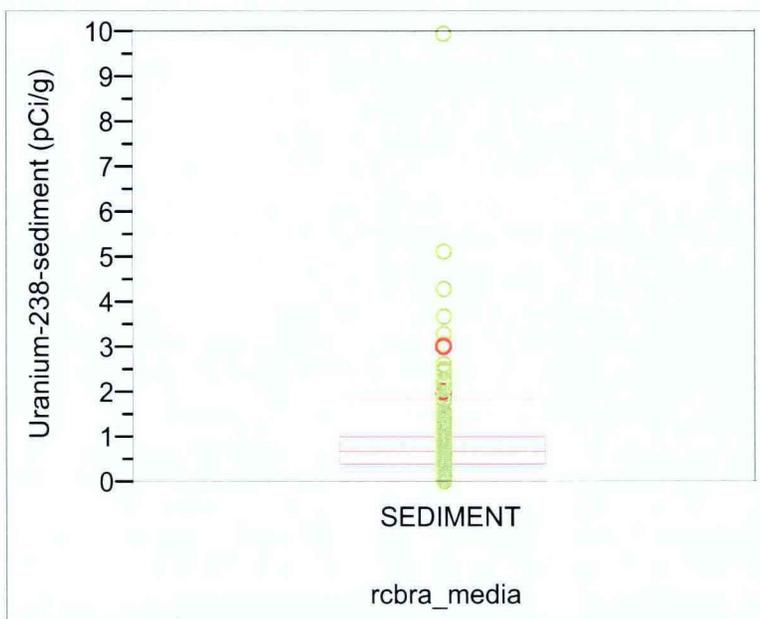
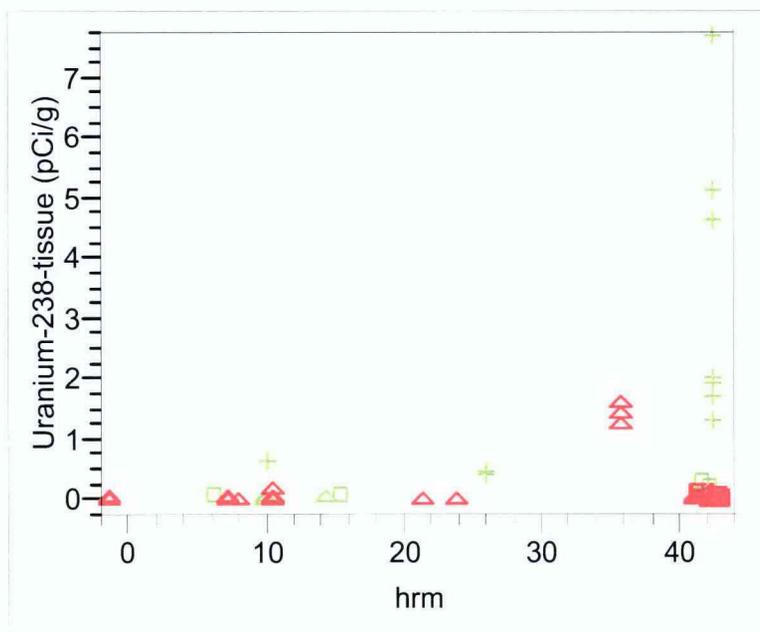


Figure 4-91. Concentrations of Uranium-238 in Tissue (pCi/g) by Hanford River Mile.



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| rcbra_media | detect_status | |
| ■ AQUATIC MACROINVERTEBRATE | ✶ MUSSEL | ■ FALSE |
| + AQUATIC VEGETATION | z PORE WATER | ■ TRUE |
| x AQUIFER TUBE | ○ SEDIMENT | |
| ■ CLAM | ■ SEEP | |
| ◇ CRAYFISH | ■ SURFACE WATER | |
| △ FISH | | |

Figure 4-92. Concentrations of Uranium-238 in Tissue (pCi/g) by Date.

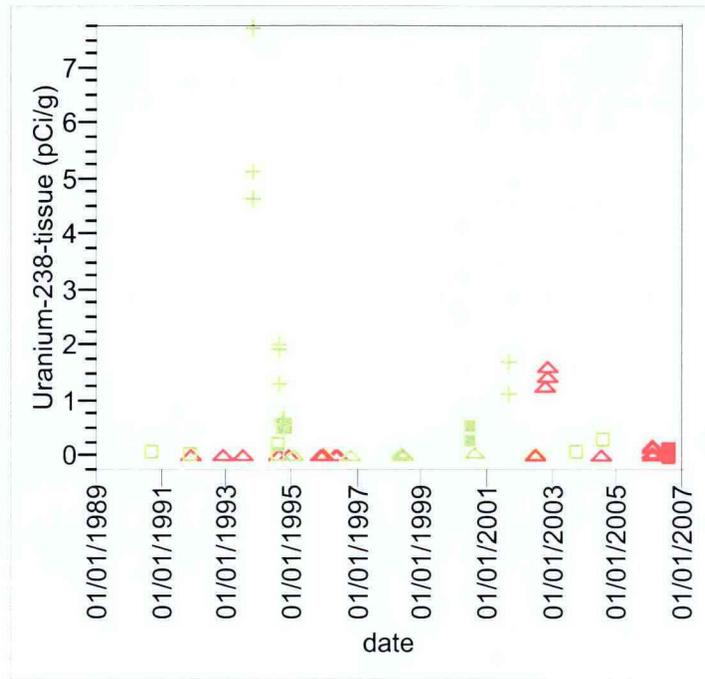
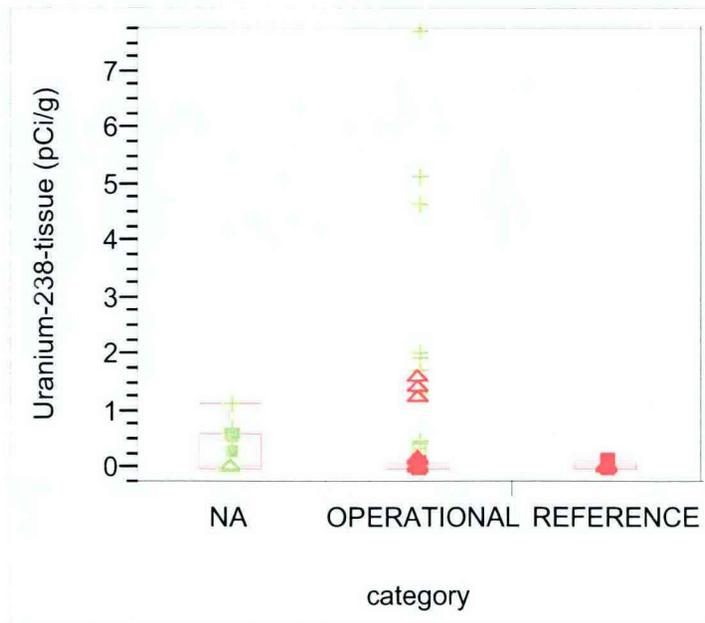
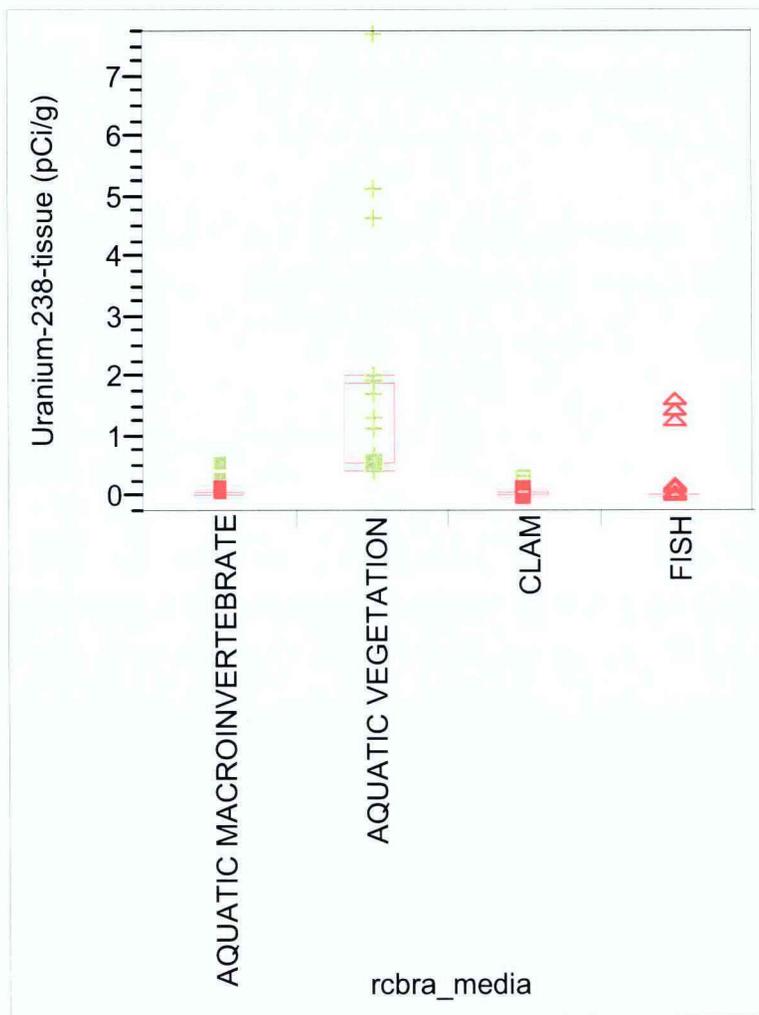


Figure 4-93. Box Plot of Uranium-238 in Tissue (pCi/g) by Category.



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ⋄ MUSSEL | ▪ FALSE |
| + AQUATIC VEGETATION | z PORE WATER | • TRUE |
| x AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ◊ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

Figure 4-94. Box Plot of Uranium-238 in Tissue (pCi/g) by RCBRA Media.



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ⚓ MUSSEL | ■ FALSE |
| + AQUATIC VEGETATION | ⚓ PORE WATER | ■ TRUE |
| × AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ◇ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

Figure 4-95. Overlay Plot of Total Calculated Uranium in Aquatic Macroinvertebrates (mg/kg) and Pore Water (ug/L).

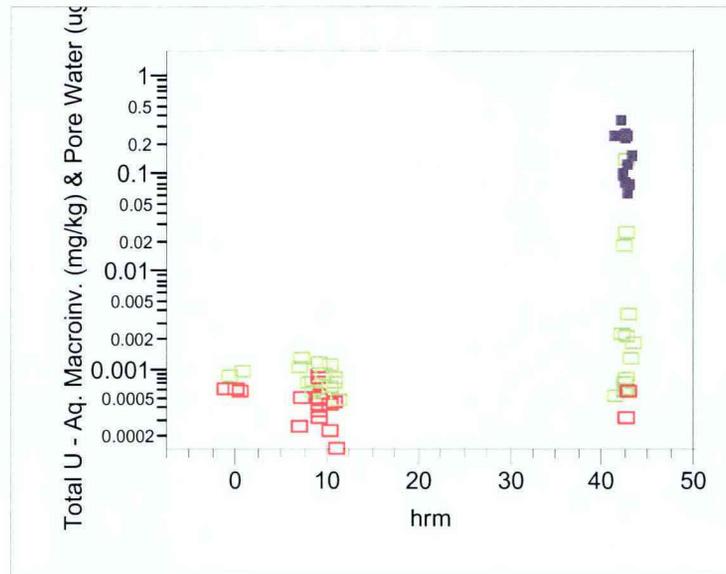
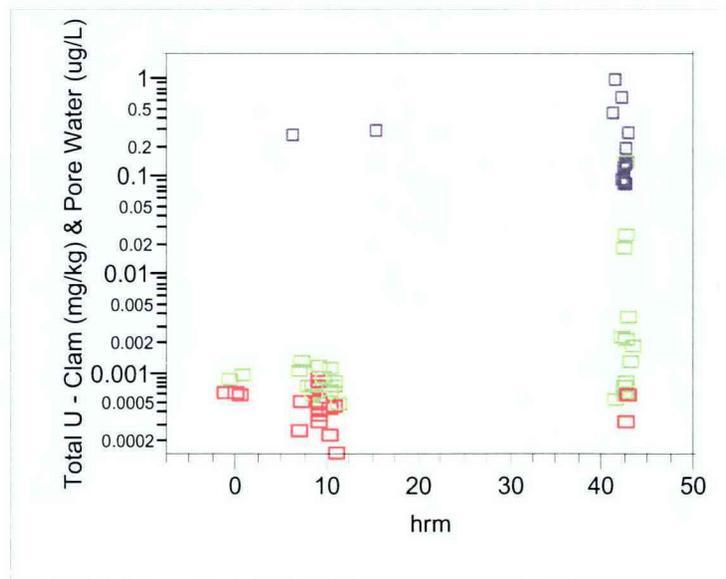


Figure 4-96. Overlay Plot of Total Calculated Uranium in Clams (mg/kg) and Pore Water (ug/L).



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ✕ MUSSEL | ■ FALSE |
| + AQUATIC VEGETATION | z PORE WATER | ■ TRUE |
| x AQUIFER TUBE | ○ SEDIMENT | |
| □ CLAM | ■ SEEP | |
| ◇ CRAYFISH | □ SURFACE WATER | |
| △ FISH | | |

Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-97. Overlay Plot of Total Calculated Uranium in Fish (mg/kg) and Pore Water (ug/L).

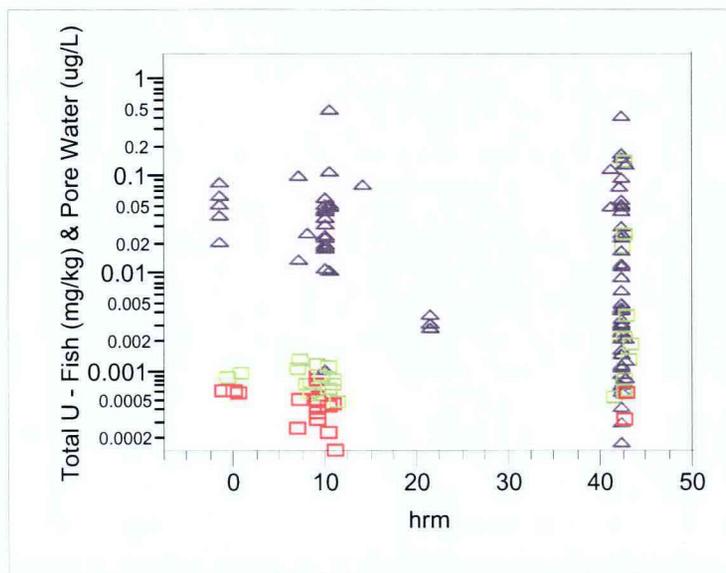
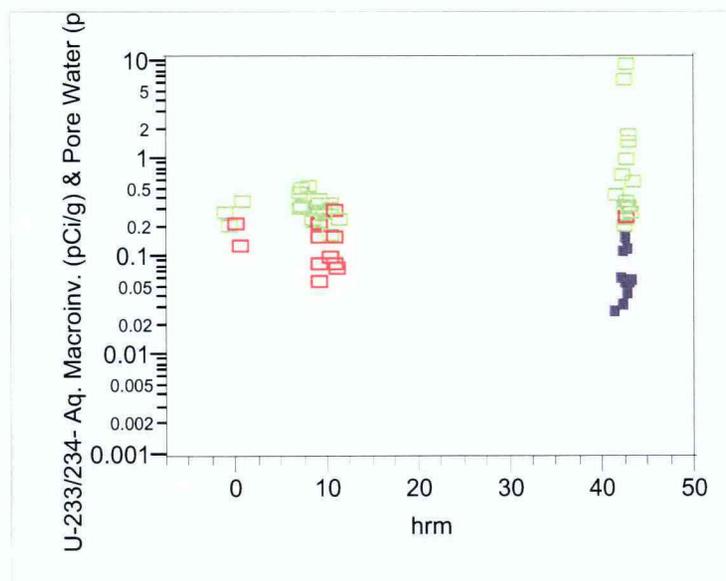


Figure 4-98. Overlay Plot of Uranium-233/234 in Aquatic Macroinvertebrates (pCi/g) and Pore Water (pCi/L).



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| rcbra_media | | detect_status |
| ■ AQUATIC MACROINVERTEBRATE | ▼ MUSSEL | ■ FALSE |
| + AQUATIC VEGETATION | ⌘ PORE WATER | ■ TRUE |
| × AQUIFER TUBE | ○ SEDIMENT | |
| ■ CLAM | ■ SEEP | |
| ◇ CRAYFISH | ■ SURFACE WATER | |
| △ FISH | | |

● Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-99. Overlay Plot of Uranium-233/234 in Clams (pCi/g) and Pore Water (pCi/L).

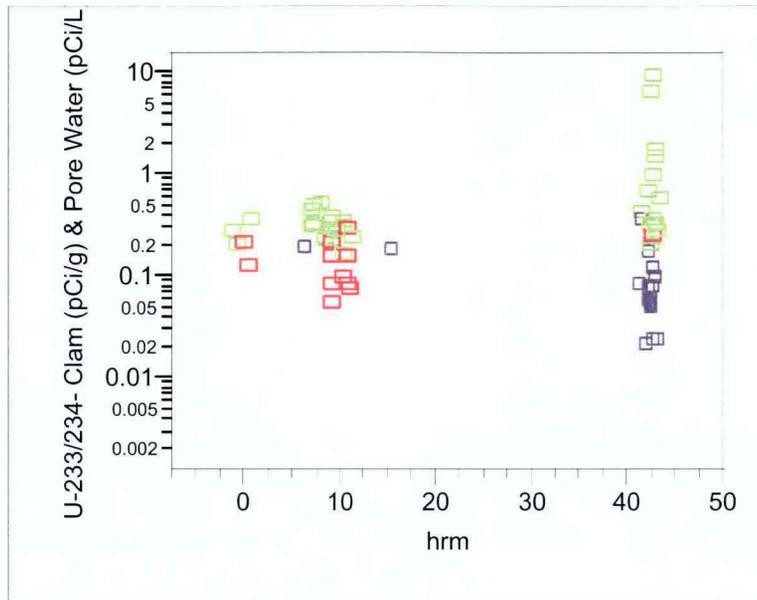
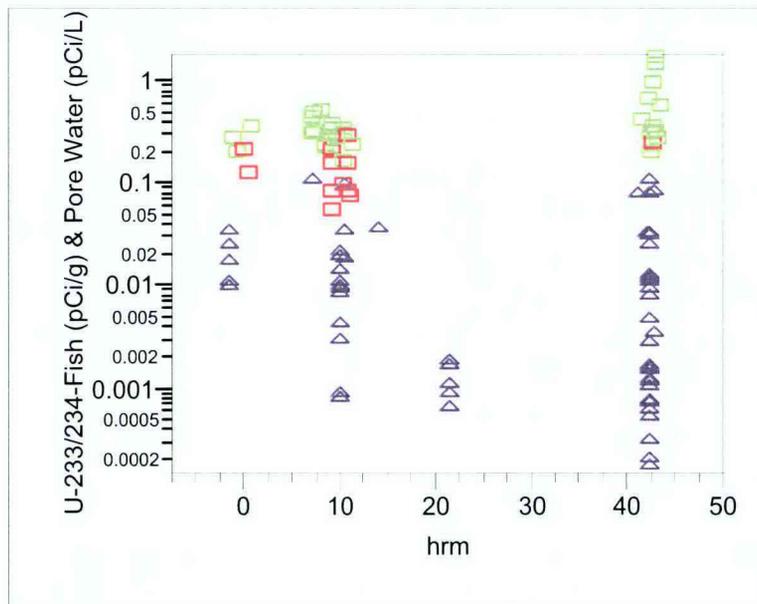


Figure 4-100. Overlay Plot of Uranium-233/234 in Fish (pCi/g) and Pore Water (pCi/L).



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ✕ MUSSEL | ▪ FALSE |
| ✚ AQUATIC VEGETATION | z PORE WATER | • TRUE |
| ✕ AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ◊ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

• Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-101. Overlay Plot of Uranium-235 in Aquatic Macroinvertebrates (pCi/g) and Pore Water (pCi/L).

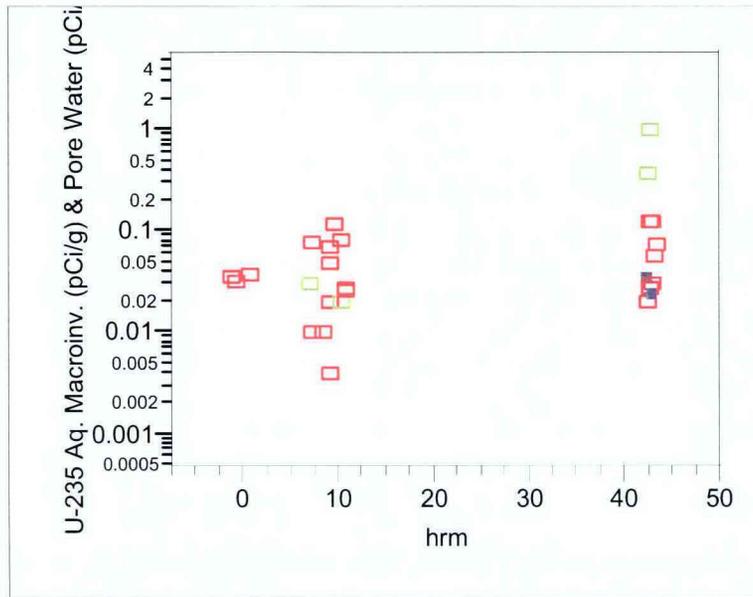
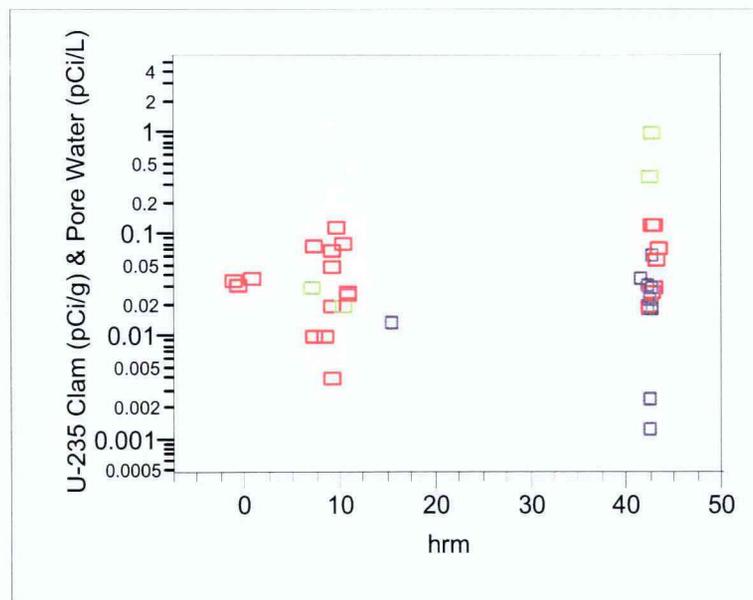


Figure 4-102. Overlay Plot of Uranium-235 in Clams (pCi/g) and Pore Water (pCi/L).



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ■ AQUATIC MACROINVERTEBRATE | ♣ MUSSEL | ■ FALSE |
| + AQUATIC VEGETATION | z PORE WATER | ■ TRUE |
| x AQUIFER TUBE | ○ SEDIMENT | |
| ■ CLAM | ■ SEEP | |
| ◇ CRAYFISH | ■ SURFACE WATER | |
| △ FISH | | |

♣ Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-103. Overlay Plot of Uranium-235 in Aquatic Fish (pCi/g) and Pore Water (pCi/L).

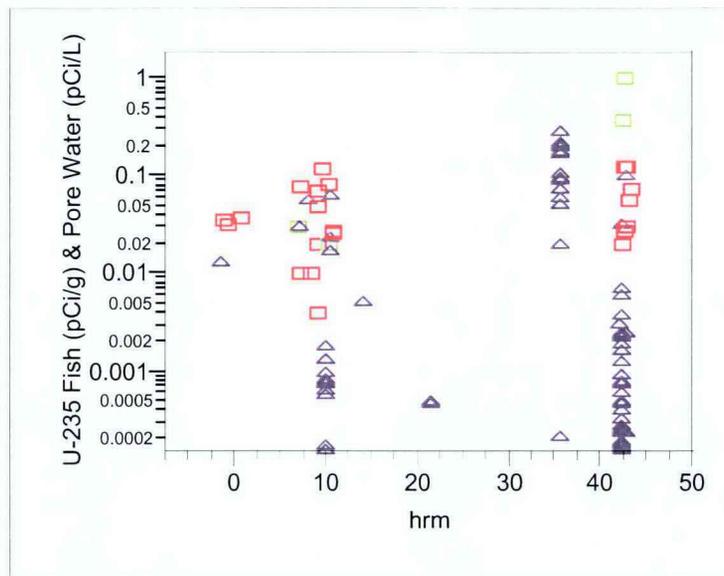
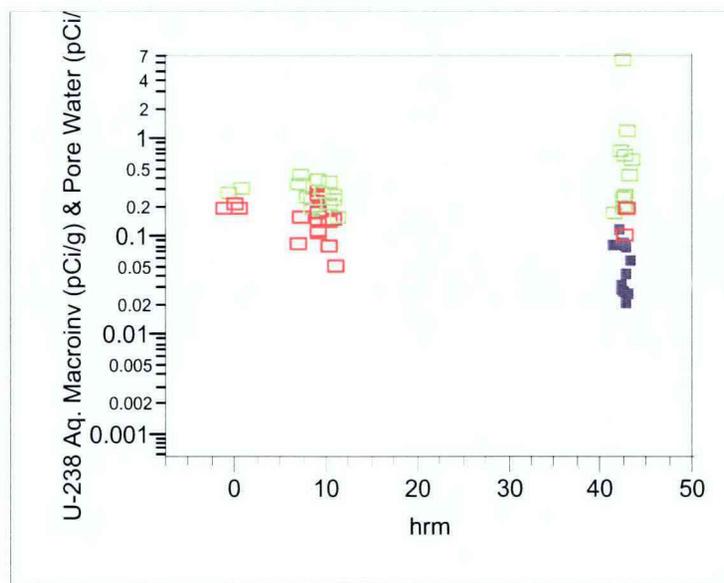


Figure 4-104. Overlay Plot of Uranium-238 in Aquatic Macroinvertebrates (pCi/g) and Pore Water (pCi/L).



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ■ AQUATIC MACROINVERTEBRATE | ▼ MUSSEL | ■ FALSE |
| + AQUATIC VEGETATION | z PORE WATER | ■ TRUE |
| x AQUIFER TUBE | ○ SEDIMENT | |
| □ CLAM | ■ SEEP | |
| ◇ CRAYFISH | ■ SURFACE WATER | |
| △ FISH | | |

● Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-105. Overlay Plot of Uranium-238 in Clams (pCi/g) and Pore Water (pCi/L).

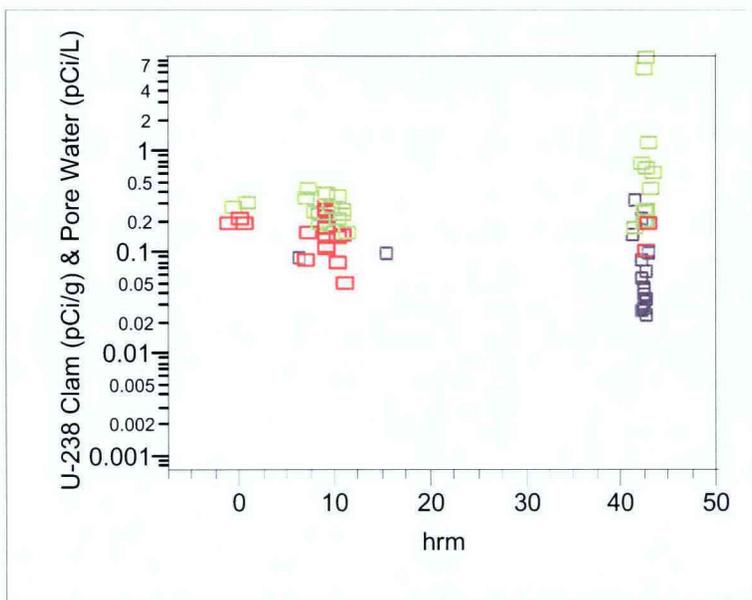
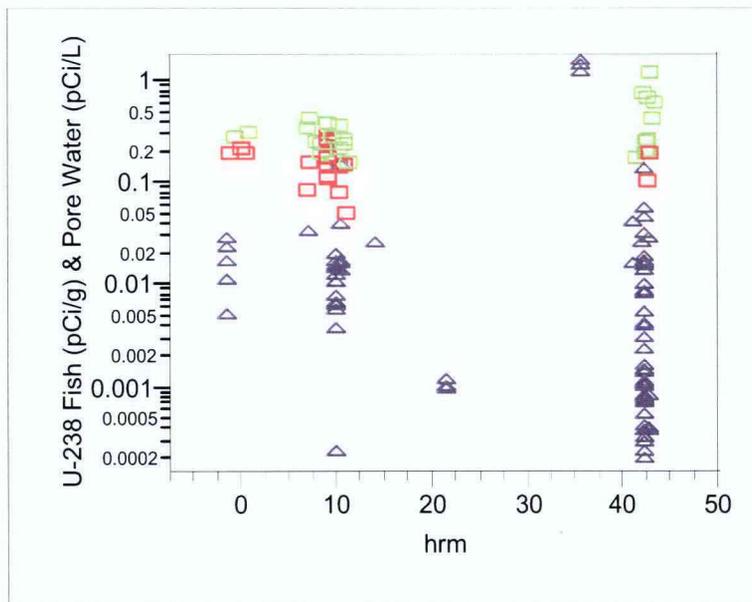


Figure 4-106. Overlay Plot of Uranium-238 in Fish (pCi/g) and Pore Water (pCi/L).



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ⚓ MUSSEL | ▪ FALSE |
| + AQUATIC VEGETATION | ⚓ PORE WATER | ▪ TRUE |
| × AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ◇ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

• Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-107. Overlay Plot of Total Calculated Uranium in Aquatic Macroinvertebrates and Sediment (mg/kg).

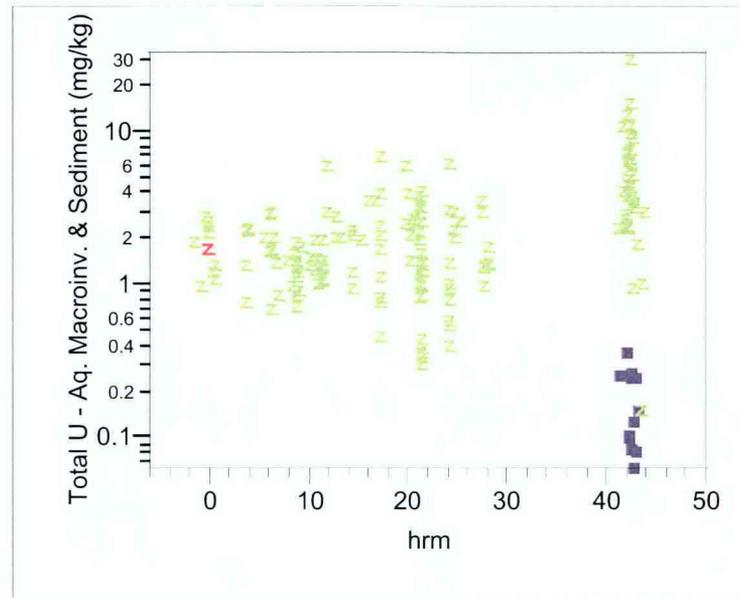
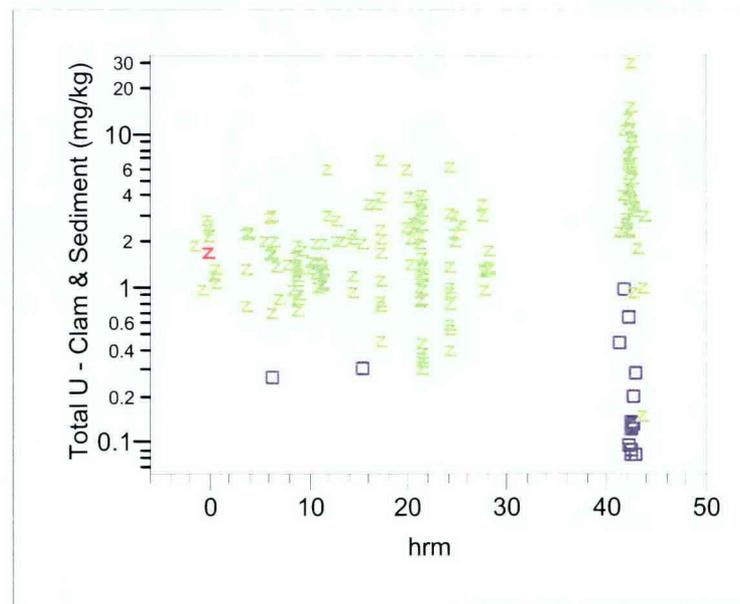


Figure 4-108. Overlay Plot of Total Calculated Uranium in Clams and Sediment (mg/kg).



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|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ✕ MUSSEL | ▪ FALSE |
| + AQUATIC VEGETATION | z PORE WATER | • TRUE |
| x AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ◇ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

• Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-109. Overlay Plot of Total Calculated Uranium in Fish and Sediment (mg/kg).

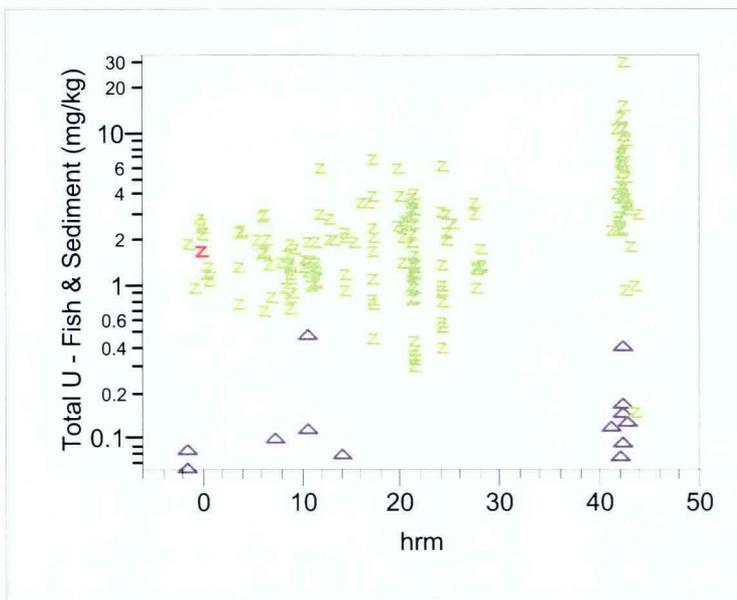
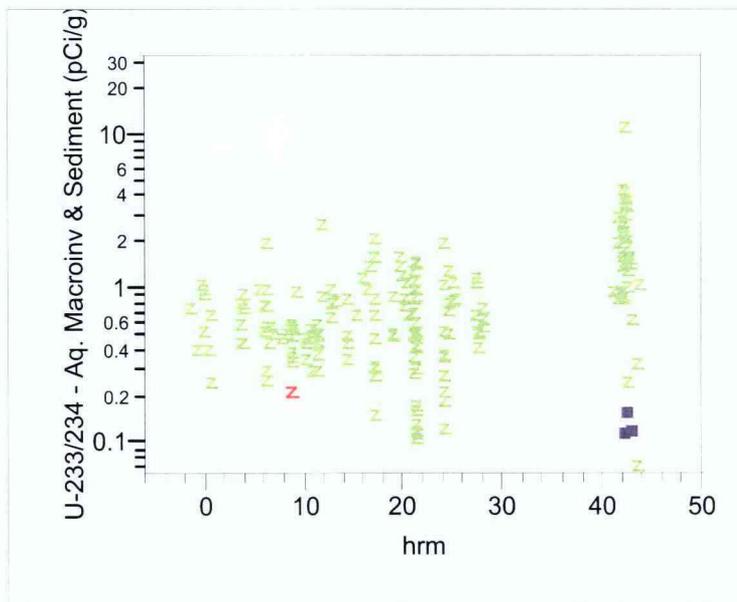


Figure 4-110. Overlay Plot of Uranium-233/234 in Aquatic Macroinvertebrates and Sediment (pCi/g).



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| rcbra_media | | detect_status |
| ■ AQUATIC MACROINVERTEBRATE | Y MUSSEL | ■ FALSE |
| + AQUATIC VEGETATION | Z PORE WATER | ■ TRUE |
| x AQUIFER TUBE | ○ SEDIMENT | |
| ■ CLAM | ■ SEEP | |
| ◇ CRAYFISH | ■ SURFACE WATER | |
| △ FISH | | |

● Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-111. Overlay Plot of Uranium-233/234 in Clams and Sediment (pCi/g).

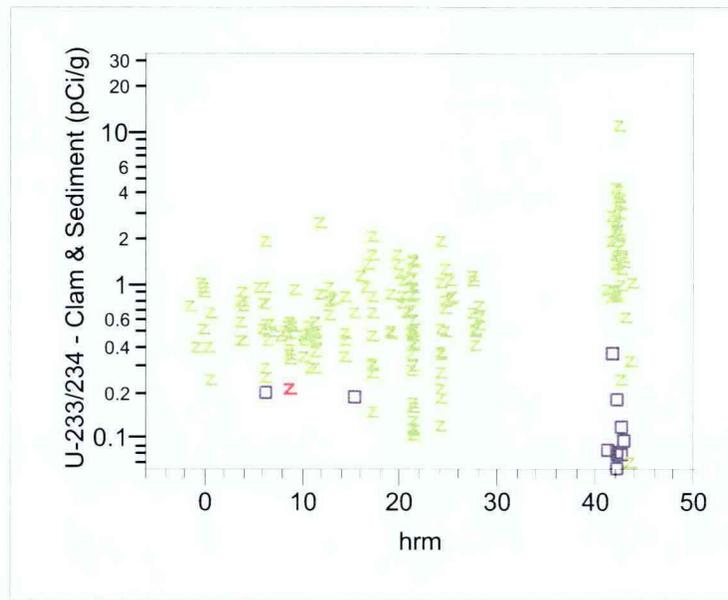
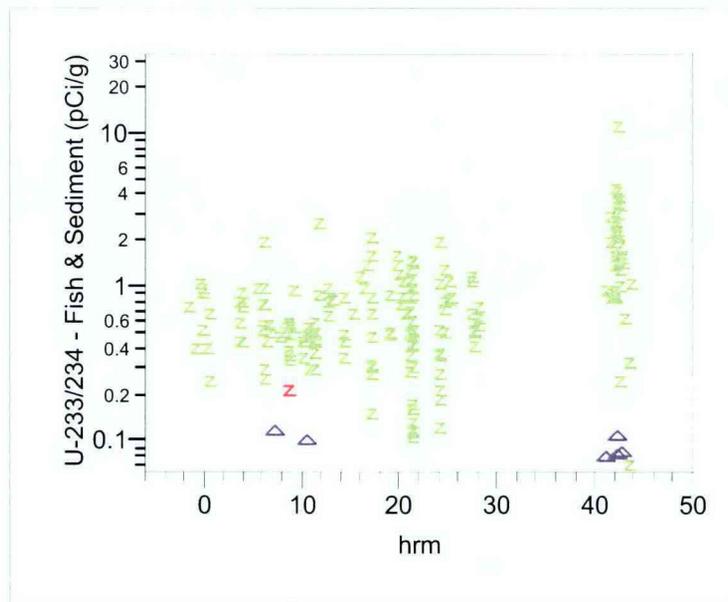


Figure 4-112. Overlay Plot of Uranium-233/234 in Fish and Sediment (pCi/g).



- | rcbra_media | | detect_status | | | |
|-------------|---------------------------|---------------|---------------|---|-------|
| ▪ | AQUATIC MACROINVERTEBRATE | ✕ | MUSSEL | ■ | FALSE |
| + | AQUATIC VEGETATION | z | PORE WATER | ● | TRUE |
| x | AQUIFER TUBE | ○ | SEDIMENT | | |
| □ | CLAM | ■ | SEEP | | |
| ◇ | CRAYFISH | □ | SURFACE WATER | | |
| △ | FISH | | | | |

● Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-113. Overlay Plot of Uranium-235 in Clam and Sediment (pCi/g).

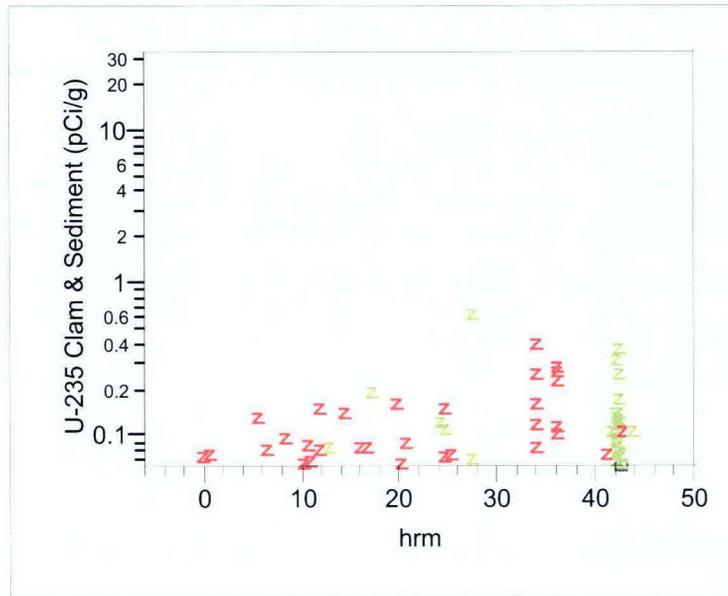
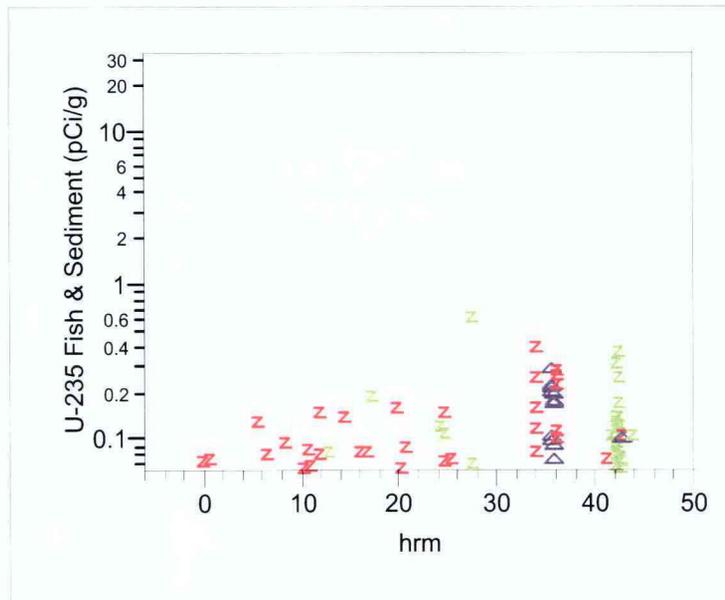


Figure 4-114. Overlay Plot of Uranium-235 in Fish and Sediment (pCi/g).



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| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | ⋈ MUSSEL | ▪ FALSE |
| + AQUATIC VEGETATION | z PORE WATER | • TRUE |
| × AQUIFER TUBE | ○ SEDIMENT | |
| ▣ CLAM | ▣ SEEP | |
| ◇ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

• Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-115. Overlay Plot of Uranium-238 in Aquatic Macroinvertebrates and Sediment (pCi/g).

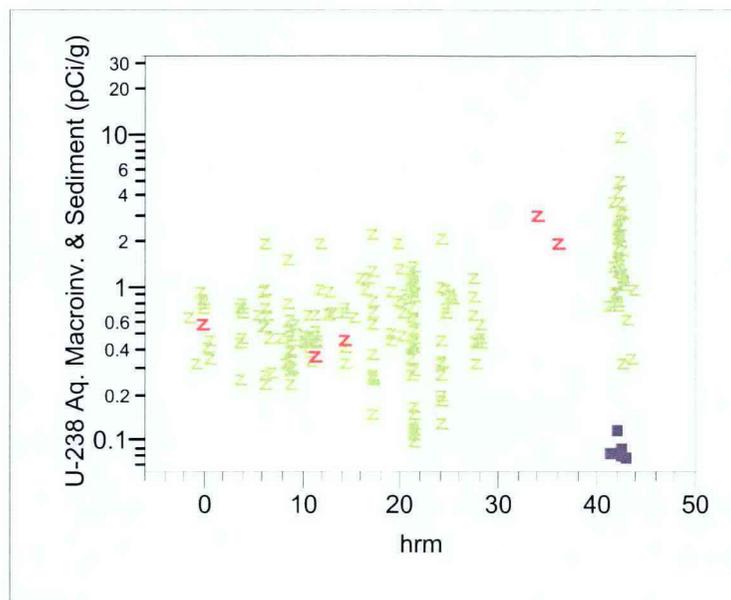
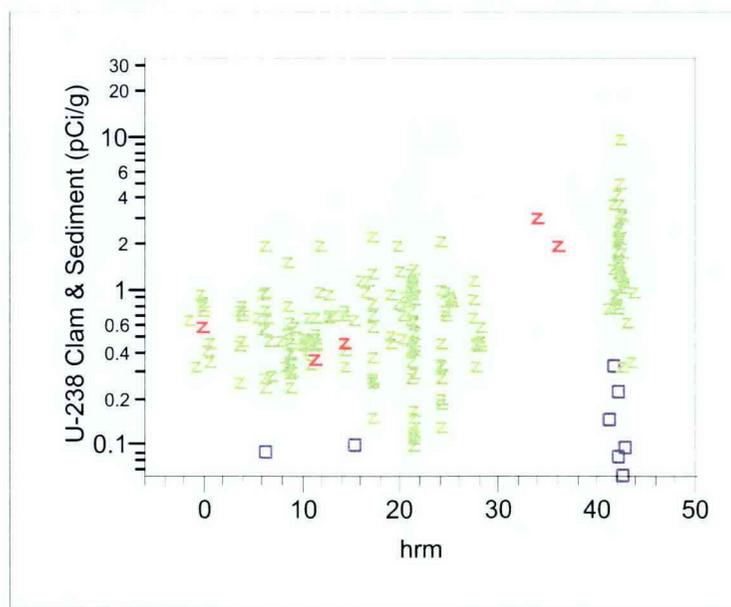


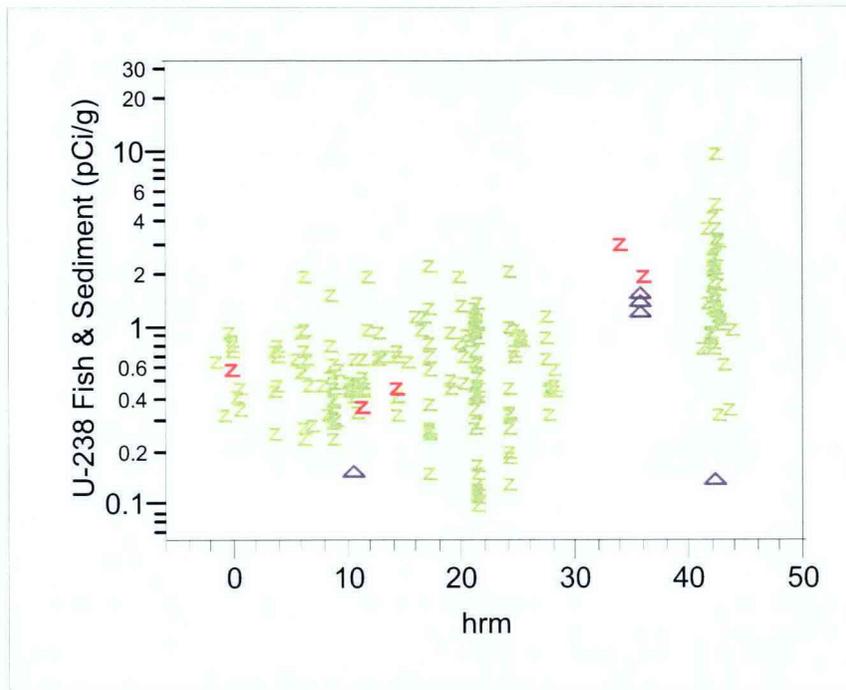
Figure 4-116. Overlay Plot of Uranium-238 in Clams and Sediment (pCi/g).



- | | | |
|-----------------------------|-----------------|---------------|
| rcbra_media | | detect_status |
| ▪ AQUATIC MACROINVERTEBRATE | Y MUSSEL | ▪ FALSE |
| + AQUATIC VEGETATION | Z PORE WATER | • TRUE |
| x AQUIFER TUBE | ○ SEDIMENT | |
| □ CLAM | ▣ SEEP | |
| ◇ CRAYFISH | ▣ SURFACE WATER | |
| △ FISH | | |

• Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-117. Overlay Plot of Uranium-238 in Fish and Sediment (pCi/g).



- | rcbra_media | | detect_status | |
|-------------|---------------------------|---------------|-------|
| ■ | AQUATIC MACROINVERTEBRATE | ■ | FALSE |
| + | AQUATIC VEGETATION | ■ | TRUE |
| x | AQUIFER TUBE | ■ | |
| ■ | CLAM | ■ | |
| ◇ | CRAYFISH | ■ | |
| △ | FISH | ■ | |
| ■ | MUSSEL | ■ | |
| z | PORE WATER | ■ | |
| ○ | SEDIMENT | ■ | |
| ■ | SEEP | ■ | |
| ■ | SURFACE WATER | ■ | |

● Blue marker indicates biota result in overlay plot. Result does not differentiate between detect and non-detect result for biota.

Figure 4-118. Time Series Plot of Hexavalent Chromium (ug/L) at HRM 10.2.

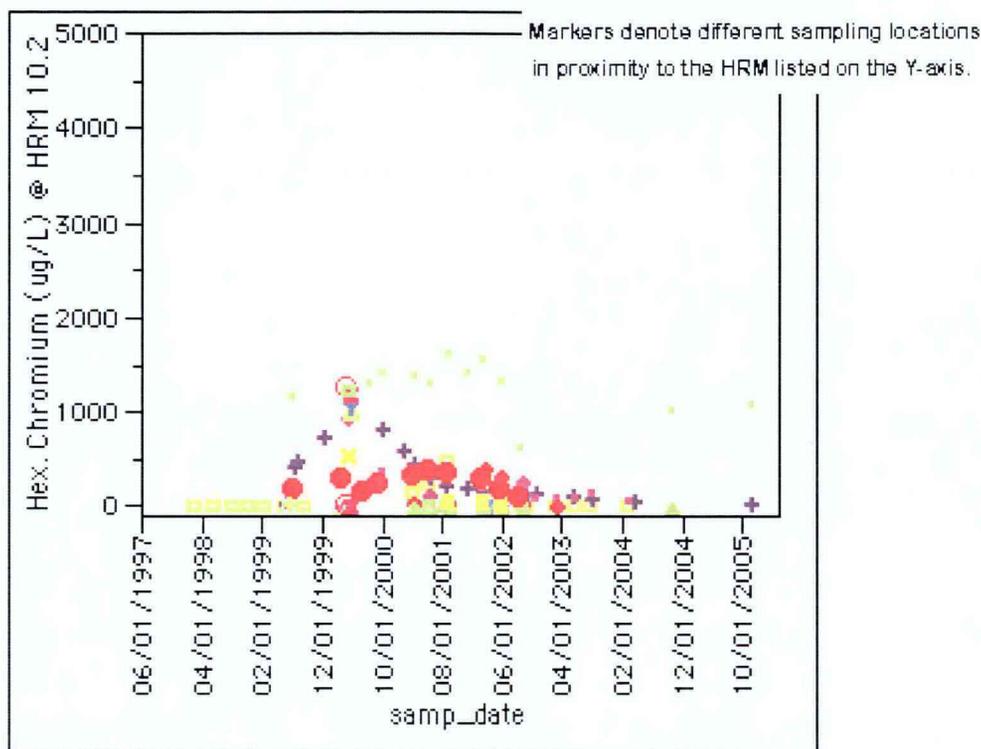


Figure 4-119. Time Series Plot of Hexavalent Chromium (ug/L) at HRM 10.3.

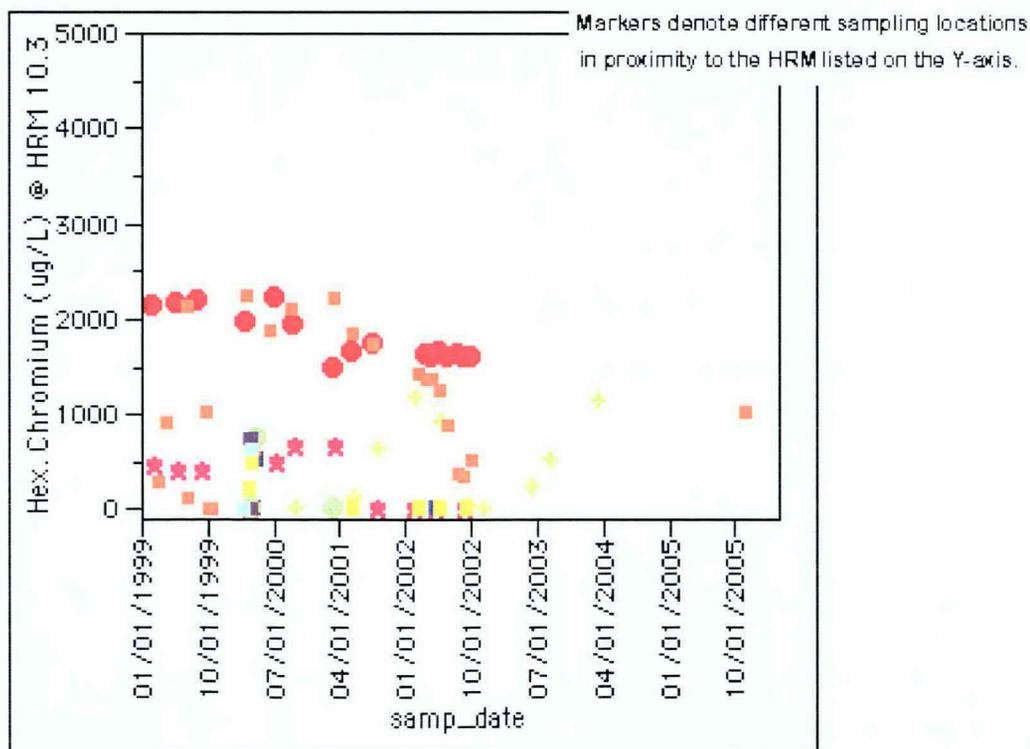


Figure 4-120. Time Series Plot of Hexavalent Chromium (ug/L) at HRM 10.4.

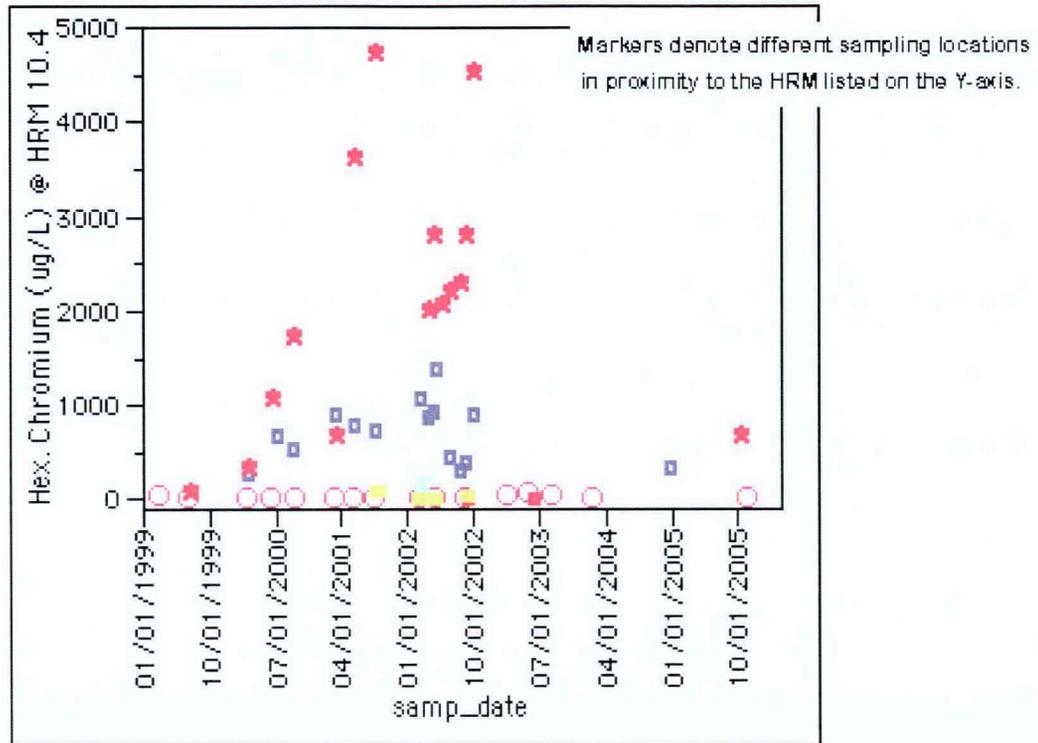


Figure 4-121. Time Series Plot of Strontium-90 (pCi/L) at HRM 6.6.

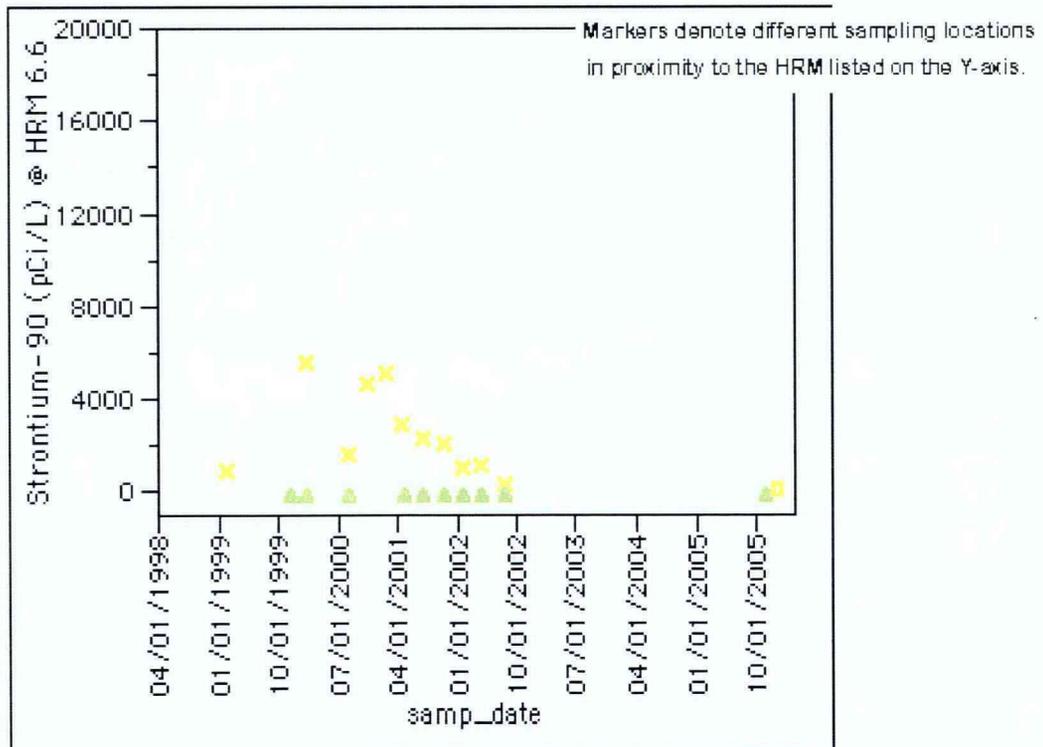


Figure 4-122. Time Series Plot of Strontium-90 (pCi/L) at HRM 8.9.

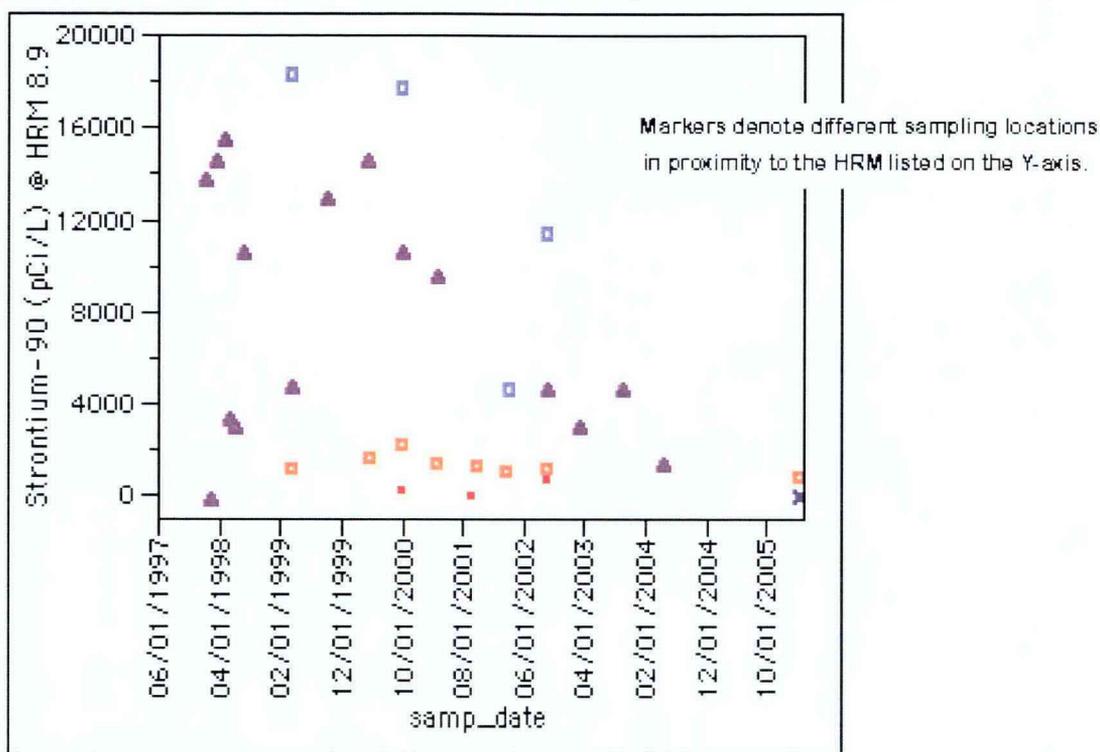


Figure 4-123. Time Series Plot of Strontium-90 (pCi/L) at HRM 9.1.

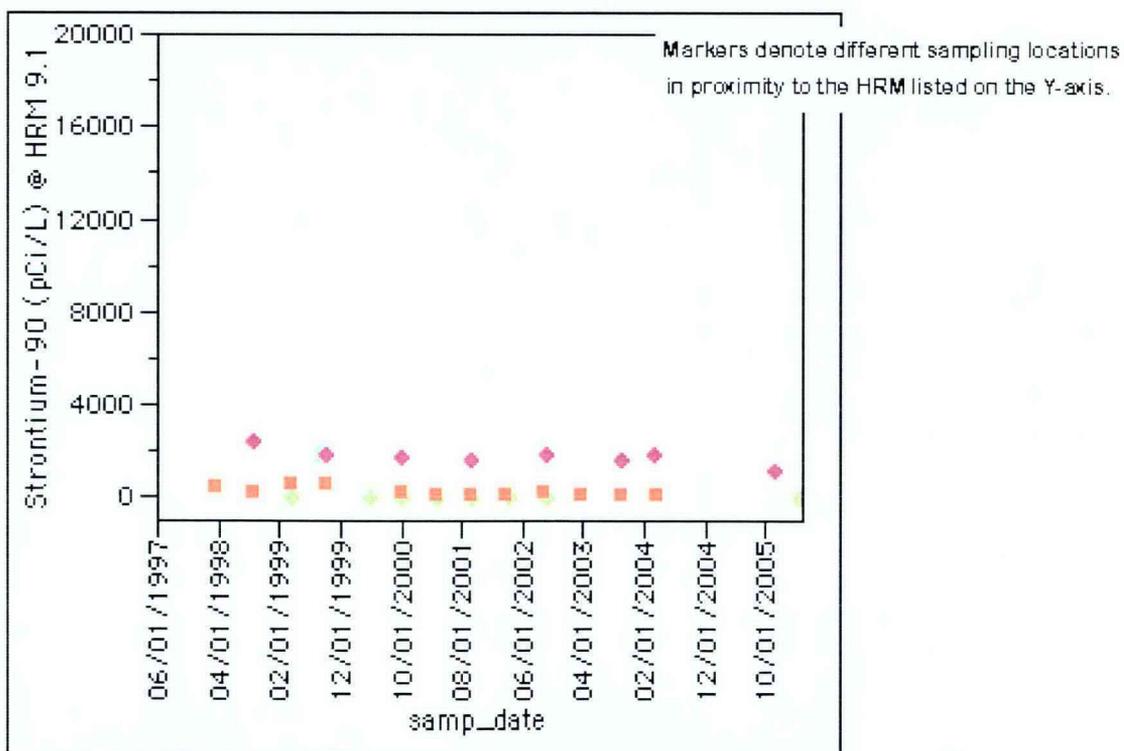


Figure 4-124. Time Series Plot of Uranium (ug/L) at HRM 42.4.

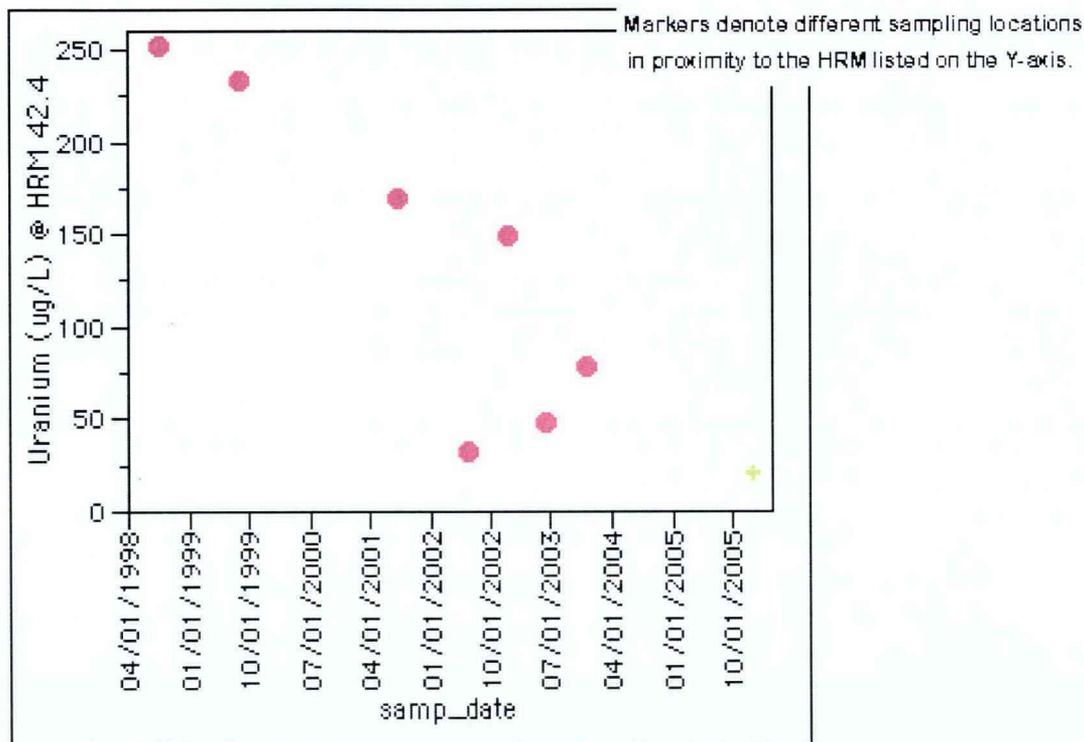


Figure 4-125. Time Series Plot of Uranium (ug/L) at HRM 42.6.

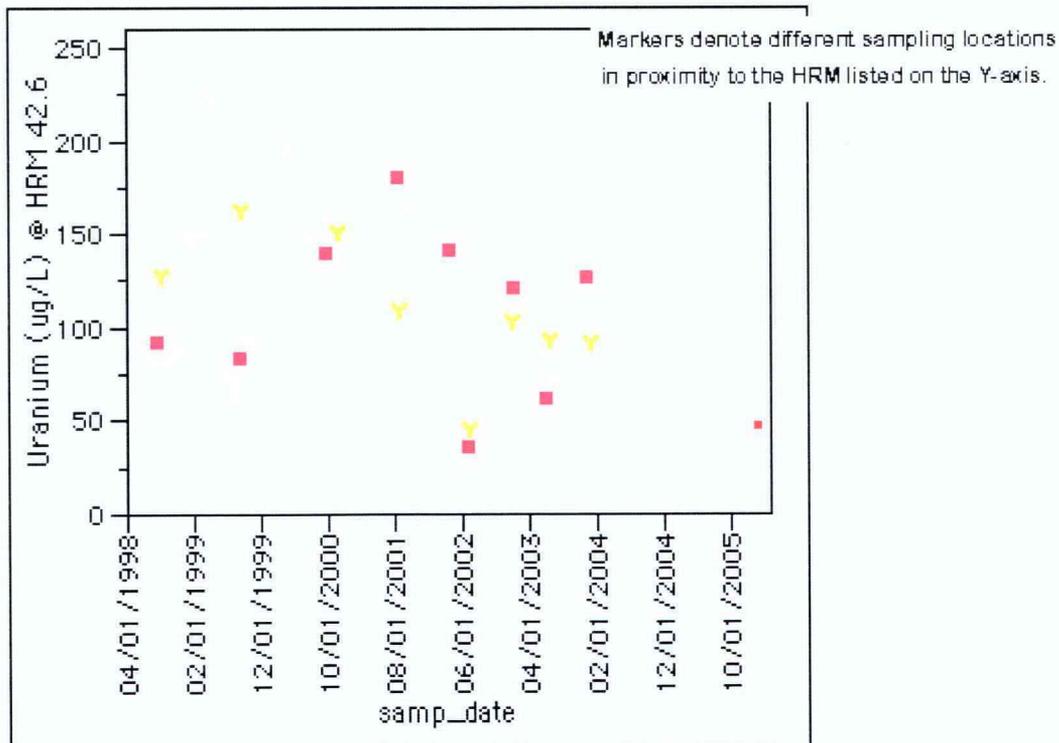


Figure 4-126. Time Series Plot of Uranium (ug/L) at HRM 42.7.

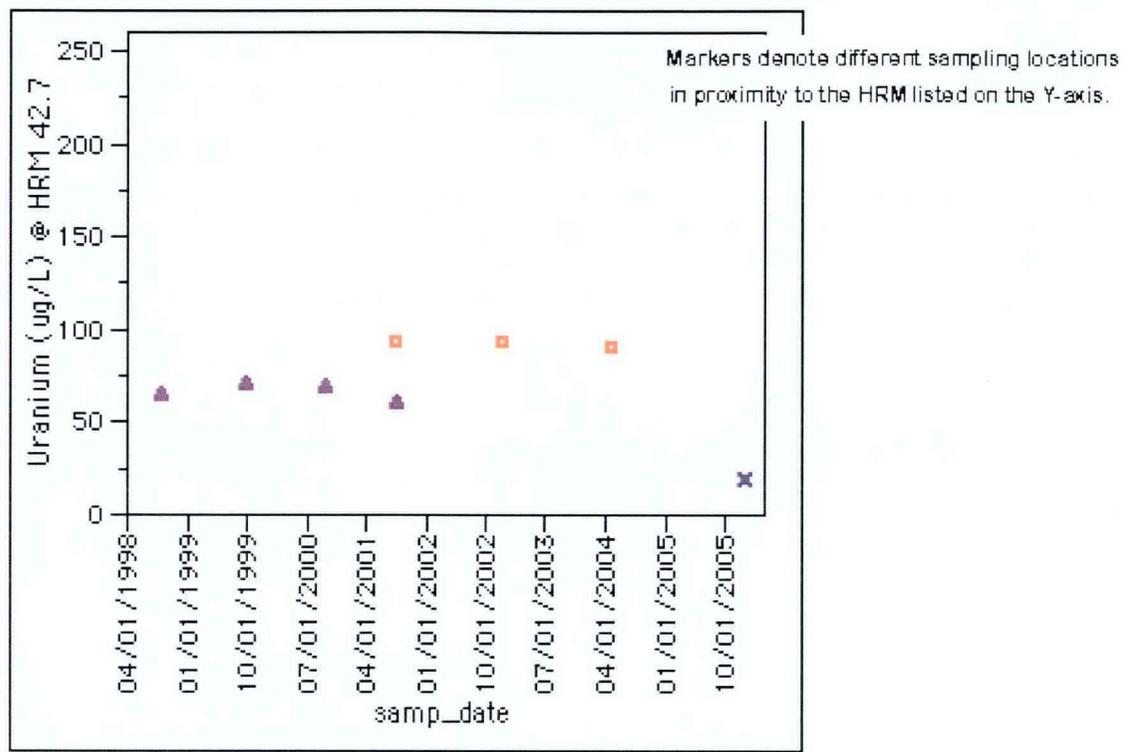


Figure 4-127. Time Series Plot of Uranium (ug/L) at HRM 42.8.

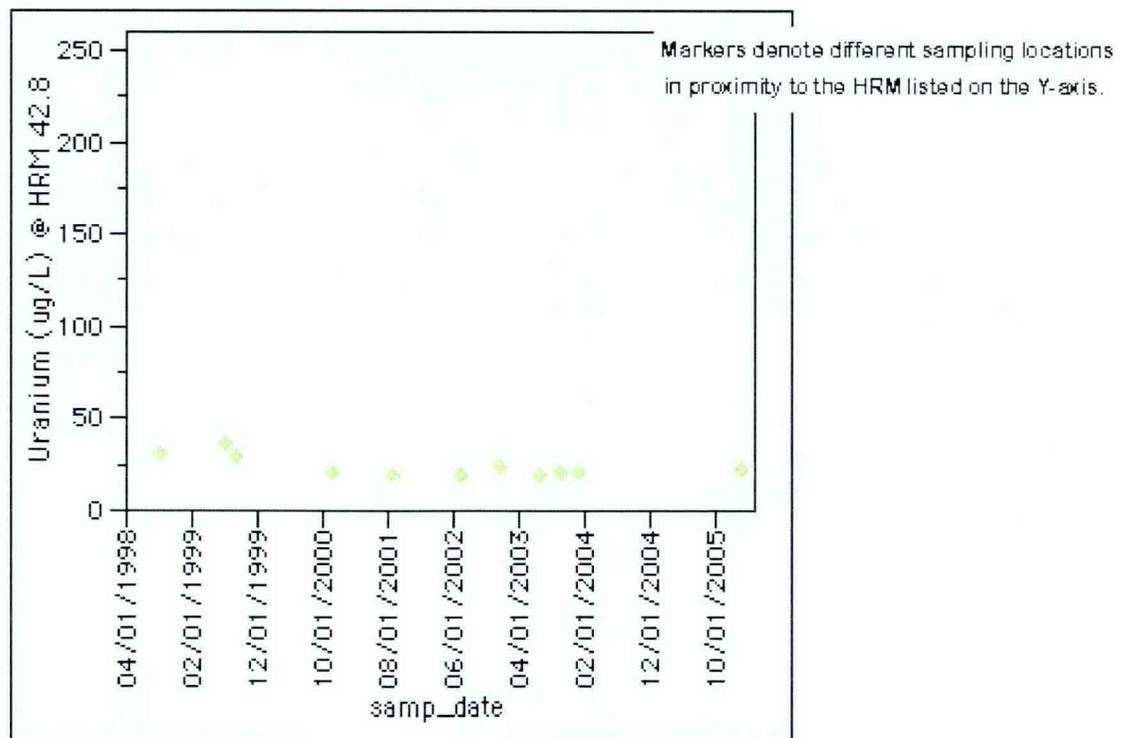


Figure 4-128. Representative Concentration UCL and Mean Calculation Decision Logic.

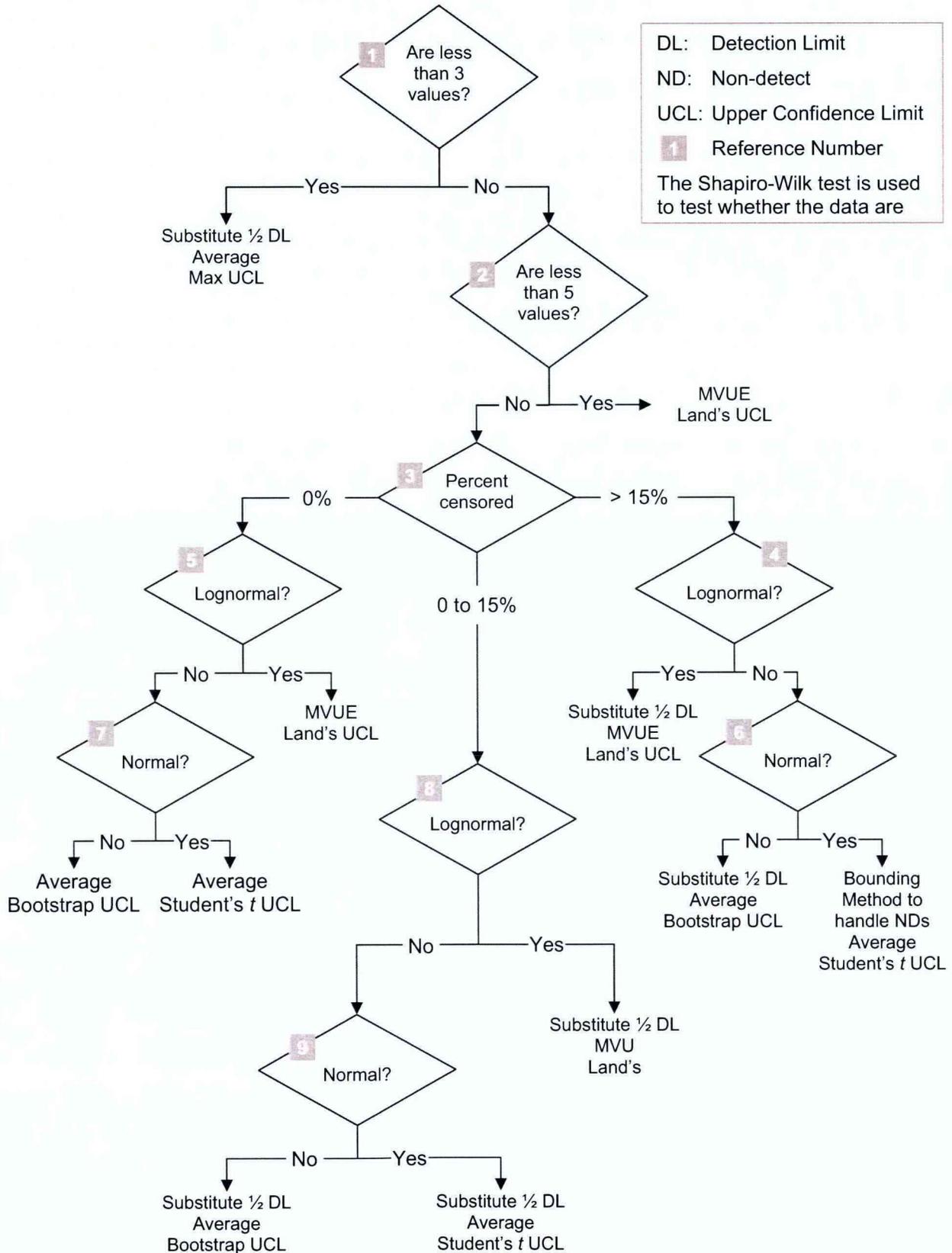
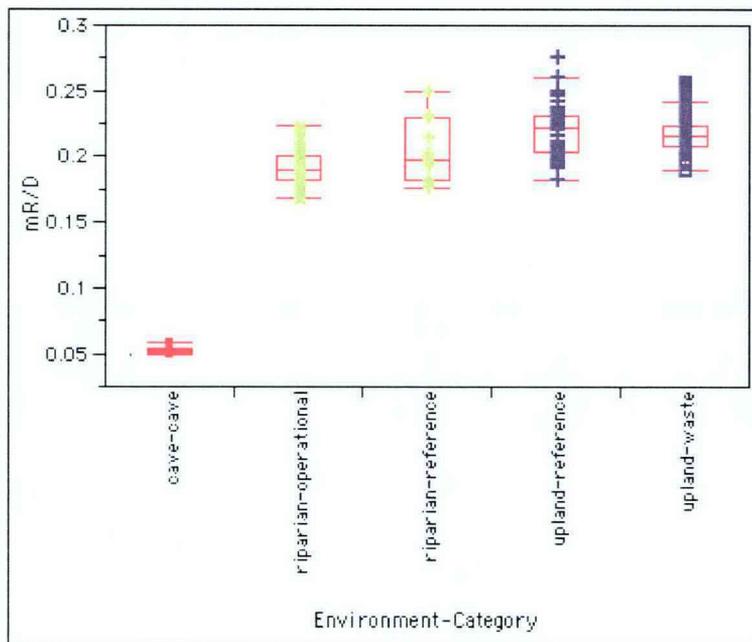
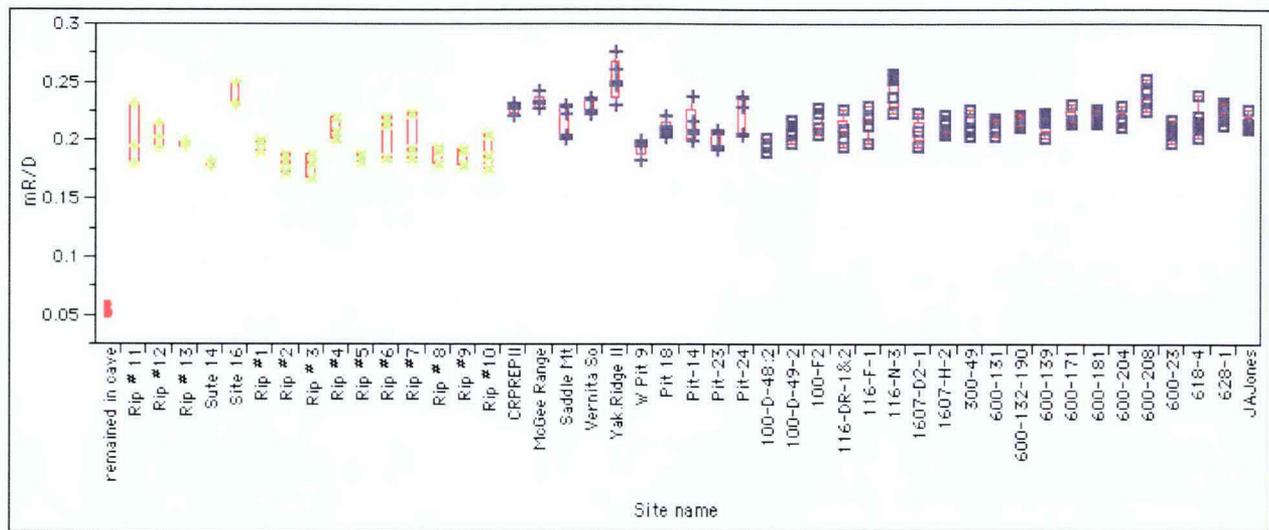


Figure 4-129. External Dosimeter Results for RCRBA Investigation Area.



Key

Category	Environment
■ cave	■ cave
+ reference	■ riparian
× operational	■ upland
■ waste	

Table 4-1. Description of Data Sources Used in the Risk Assessment. (2 Pages)

Source File	Contents
RCBRA	
a_ENRE_by_SAF_results.txt	100% data pull for RCBRA-specific data.
_RCBRA_Sup5_tab.txt	Rad, metal, organic data from supplemental pore water samples. Contains supplemental pore water sampling data.
RC-072 sample.txt RC-072_Result.txt	Discrete soil sample data. Contains soil data from T&E plant habitat discrete sampling.
tox	Contains results of toxicity testing for nematode, bluegrass, pak choi, ceriodaphnia, hyallela, and FETAX
CVP/RSVP	
CVP_RSVP_risk2_r1.txt	Contains CVP/RSVP data for 65 waste sites.
serla_result.txt	Contains verification sampling data for waste sites prior to 2004. These results were initially evaluated in the screening-level assessment as part of the DQO.
100-B/C Pilot	
Hanford_BC_Pilot_DB.txt	Contains analytical data for biotic and abiotic media used to develop the 100-B/C Pilot Project Risk Assessment Report (DOE/RL-2005-40).
100-NR-2	
tbl_100NR2_results_all.txt	Contains biotic and abiotic samples from 100-NR-2 shoreline investigation. Results table from NR2.mdb with metadata where available.
OTHER	
HEIS_aquifer_tubes.txt HEIS_biota.txt HEIS_groundwater.txt HEIS_sediment.txt HEIS_surface_water.txt	Contains biotic and abiotic data for these media from the HEIS database.
PNNL_Biota_rad_data.txt	Contains radiological tissue data for vegetation, mammals, birds, fish from 1990 to 2003.
PNNL_Sample_water.txt	Contains rad, metal, and organic data from 1990 to 2003 for seep, groundwater, surface water, and river water.
PNNL_Sediment_data.txt	Contains select rad and metal data from 1990 to 2003 for sediment ¹ .
transposedPNNLmetals.txt	Contains metals data for plants, birds, mammals, fish, invertebrates.
CRCBRA.txt	Contains data for several media collected under the Columbia River Component data compilation effort. Data in the CRCBRA file pertain only to those whose coordinates fall within the Hanford Site polygon and immediate vicinity.

Table 4-1. Description of Data Sources Used in the Risk Assessment. (2 Pages)

Source File	Contents
WARD SOIL 90-04 XY.txt WARD VEG 90-04 XY.txt WARD SOIL 05 XY.txt WARD VEG 05 XY.txt	Contains Surface Environmental Surveillance Program data from far-field surveillance of abiotic and biotic media from 1990 to 2005.
<i>Washington State Background</i>	
WA_reg_SW_mean_bkgnd.txt	Contains Washington State metals background data published by the Washington State Department of Ecology (Ecology 1994, Publication 94-115)
<i>Area Background</i>	
DOE-RL_95-55_Table_A-2_export.txt	Contains near-surface radiological soil sampling results used to calculate background in the Hanford Site vicinity. (Data published in DOE/RL-95-55)
DOERL-96-12_radvadose.txt	Contains vadose zone soil sampling results in the Hanford Site vicinity used to calculate rad background. (Data published in DOE/RL-96-12)
EQM data non-rad.txt	Contains vadose zone soil sampling results in the Hanford Site vicinity used to calculate background for metals published in DOE/RL-92-24.

- CVP = cleanup verification package
- DQO = data quality objective
- HEIS = Hanford Environmental Information System database
- PNNL = Pacific Northwest National Laboratory
- RCBRA = River Corridor Baseline Risk Assessment
- RSVP = remaining sites verification package
- T&E = threatened and endangered

Table 4-2. Summary of Biotic Tissue Samples Collected and Suite Analyses Performed for the 100 Area and 300 Area Component of the RCBRA. (2 Pages)

Analyte Group	Analytical Methods ^a	Upland/Riparian Tissues ^b										Near-Shore Aquatic Tissues ^b						Indicator Contaminants ^c
		Plants	Small mammal whole body	Small mammal liver/kidney	Soil invertebrate	Birds	Macroinvertebrate	Sculpin/Juv. Sucker whole body	Sculpin liver/kidney	Amphibians	Clam soft tissue	Clam shells	Yes	No	Yes	No	Yes	
Inorganic chemicals	6010/6020	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	Yes	No	Al, Sb, As, Ba, Be, Cd, Cr, Cu, Fe, Pb, Mn, Ni, Ag, Sn, V, Zn
	6010/6020	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	Yes	No	U
	7196A	No	No	No	No	No	No	No	No	No	No	No	No	No	No	No	No	Cr-VI
	7470/7471	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	Yes	No	Hg ^d
PCBs and pesticides	300.0	No	No	No	No	No	No	No	No	No	No	No	No	No	No	Yes	No	NO ₃
	8081A	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	No	No	No	No	No	Methoxychlor
	8082	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	No	No	No	No	No	None, but PCBs in fish have been risk drivers in previous risk assessments (EPA 2002)
Semivolatile organic chemicals	418.1	No	No	No	No	No	No	No	No	No	No	No	No	No	No	No	No	TPH
	WTPH-G and WTPH-D	No	No	No	No	No	No	No	No	No	No	No	No	No	No	No	No	TPH gasoline and diesel range
	8270A	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	No	No	No	No	No	None, but needed to evaluate risk from TPH constituents
Radionuclides	Carbon-14 ^e	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	No	No	No	No	No	C-14
	Gamma energy analysis	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	No	No	No	No	No	Cs-137, Co-60, Ra-226 ^f , Ra-228 ^f
	Isotopic thorium	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	No	No	No	No	No	Th-232
	Isotopic uranium	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	No	No	No	No	No	U-233/234, U-235, U-238

Table 4-2. Summary of Biotic Tissue Samples Collected and Suite Analyses Performed for the 100 Area and 300 Area Component of the RCBRA. (2 Pages)

Analyte Group	Analytical Methods ^a	Upland/Riparian Tissues ^b				Near-Shore Aquatic Tissues ^b						Indicator Contaminants ^c		
		Plants	Small mammal whole body	Small mammal liver/kidney	Soil invertebrate	Birds	Macroinvertebrate	Sculpin/Juv. Sucker whole body	Sculpin liver/kidney	Amphibians	Clam soft tissue		Clam shells	
	Total radioactive strontium	Yes	Yes	No	Yes	Yes	Yes	Yes	No	No	No	Yes	Yes	SI-90

^a All analytes obtained by these methods.

^b Analyses performed where analytical mass requirements met. Tissues include upland and riparian plants, small mammals, small mammal liver/kidney, soil invertebrates, benthic macroinvertebrates, sculpin, juvenile sucker, sculpin liver and kidney, amphibians, and birds.

^c Indicator contaminants were used to identify analytical suites for the risk assessment as stated in DOE/RL-2005-41, Rev.1.

^d Methods 7470 and 7471 used for mercury analysis in water and soil, respectively. Mercury was not identified as an indicator contaminant in the near-shore or riparian environment but was investigated for completeness and for human health analysis.

^e Carbon-14 measured in tissues at select locations associated with carbon-14 contamination.

^f Radium concentrations measured by gamma energy analysis.

PCB = polychlorinated biphenyl

TPH = total petroleum hydrocarbon

Table 4-3. Summary of Abiotic Samples Collected and Suite Analyses Performed for the 100 Area and 300 Area Component of the RCBRA. (2 Pages)

Analyte Group	Analytical Methods ^a	Analyzed in			Indicator Contaminants ^c
		Soil	Sediment ^b	Surface Water	
Inorganic chemicals	6010/6020	Yes	Yes	Yes	Al, Sb, As, Ba, Be, Cd, Cr, Cu, Fe, Pb, Mn, Ni, Ag, Sn, U, V, Zn
	7196A	Yes	Yes	Yes	Cr-VI
	7470/7471	Yes	Yes	Yes	Hg ^d
	300.0	Yes	No	Yes	NO ₃
PCBs and pesticides	8081A	Yes	Yes	Yes	Methoxychlor
	8082	Yes	Yes	Yes	None, but PCBs in fish have been risk drivers in previous risk assessments (EPA 2002)
Semivolatile organic chemicals	418.1	No	Yes	Yes	TPH
	WTPH-G and WTPH-D	No	Yes	Yes	TPH gasoline and diesel range
	8270A	Yes	Yes	Yes	None, but needed to evaluate risk from TPH constituents
	Gamma energy analysis	Yes	Yes	Yes	Cs-137, Co-60, Ra-226 ^e , Ra-228 ^f
Radionuclides	Isotopic thorium	Yes	Yes	Yes	Th-232
	Carbon-14 ^e	No	Yes	Yes	C-14
	Isotopic uranium	Yes	Yes	Yes	U-233/234, U-235, U-238
	Total radioactive strontium	Yes	Yes	Yes	Sr-90
Soil Toxicity	Tritium	No	No	Yes	H-3
	Plant (Sandberg's Bluegrass) Nematode (<i>Caenorhabditis elegans</i>)	Yes	No	No	Survival/growth

Table 4-3. Summary of Abiotic Samples Collected and Suite Analyses Performed for the 100 Area and 300 Area Component of the RCBRA. (2 Pages)

Analyte Group	Analytical Methods ^a	Analyzed in			Indicator Contaminants ^f
		Soil	Sediment ^b	Surface Water, Pore Water	
Sediment Toxicity	Plant (pak choi) Invertebrates (<i>Hyalella azteca</i>)	No	Yes	No	Survival/growth
Aquatic Toxicity	Amphibians (FETAX) Invertebrates (<i>Ceriodaphnia dubia</i>)	No	No	PW only	Survival/growth

^a All analytes obtained by these methods.

^b Analyses performed where analytical mass requirements met.

^c Indicator contaminants were used to identify analytical suites for the risk assessment.

^d Methods 7470 and 7471 used for mercury analysis in water and soil, respectively. Mercury was not identified as an indicator contaminant in the near-shore or riparian environment but was investigated for completeness and for human health analysis.

^e Carbon-14 measured at select locations associated with carbon-14 contamination.

^f Radium concentrations in water measured by total radium isotopic analysis. Gamma energy analysis used for all other media types.

PCB = polychlorinated biphenyl

PW = pore water

TPH = total petroleum hydrocarbon

Table 4-4. Summary of Data Resources Used for Comparison to Waste Site and 100 Area and 300 Area RCBRA Investigation Area and Near-Shore Aquatic Location Data.

Category	Resource (source code is bolded)	Description of Data Extracted
Area Background	DOE/RL-92-24, 1995, <i>Hanford Site Background: Part 1, Soil Background for Nonradioactive Analytes</i> , Rev. 4, 2 vols., U. S. Department of Energy, Richland Operations Office, Richland, Washington	Medium: Soil. Data were obtained from HEIS. Non-radionuclides only. Coordinates were not provided.
Area Background	DOE/RL-95-55, 1995, <i>Hanford Site Background: Evaluation of Existing Soil Radionuclide Data</i> , Rev. 0, U.S. Department of Energy, Richland Operations Office, Richland, Washington	Medium: Soil. Data were hand entered. Radionuclides only. Coordinates provided.
Area Background	DOE/RL-96-12, 1996, <i>Hanford Site Background: Part 2, Soil Background for Radionuclides</i> , Rev. 0, U.S. Department of Energy, Richland Operations Office, Richland, Washington	Medium: Soil. Some data were obtained from HEIS and some data were hand entered. Radionuclides only.
Washington State Background	Natural Background Soil Metals Concentrations in Washington State, Charles San Juan, Toxics Cleanup Program, Olympia, Washington, Publication 94-115, October 1994	Medium: Soil. Non-radionuclides.
Other	Surface Environmental Surveillance Program data, originating from PNNL far-facility electronic monitoring files WARD SOIL 90-04 XY.txt, WARD VEG 90-04 XY.txt, WARD SOIL 05 XY.txt, WARD VEG 05 XY.txt.	Media: Soil, vegetation (1990 to 2005). Data provided by PNNL from their database. Coordinates were provided for some locations and converted to a common coordinate system.
Other	Hanford Site-relevant data from the <i>Columbia River Component Data Evaluation Summary Report</i> , WCH-91, Rev. 0, Washington Closure Hanford, Richland Washington, July, 2006.	Media: Biota, surface water, and sediment data from a variety of sources.

HEIS = Hanford Environmental Information System

PNNL = Pacific Northwest National Laboratory

Table 4-5. Summary of Reference Site Samples, Washington State Background, Area Background for Inorganic and Radionuclides. (2 Pages)

Analyte name	Units	Reference Sites						Washington State Background			Area Background		
		Upland			Riparian			Minimum	Median	Maximum	Minimum	Median	Maximum
		Minimum mean	Median mean	Maximum mean	Minimum mean	Median mean	Maximum mean						
Calculated Total Uranium	mg/kg	0.296	0.407	0.796	1.773	2.193	2.653
Vanadium	mg/kg	42	54.04	69.28	36.84	41.58	52.18	.	.	.	24.3	52.3	105
Zinc	mg/kg	39.06	46.92	56.66	123.6	170.4	290	12	50.6	132.5	24.2	50.2	366
Americium-241	pCi/g	0.078	0.097	0.145	0.119	0.154	0.192
Cesium-137	pCi/g	0.046	0.064	0.123	0.077	0.117	0.185	.	.	.	-0.0017	0.362	1.78
Cobalt-60	pCi/g	0.035	0.048	0.069	0.059	0.06	0.065
Plutonium-238	pCi/g	0	0.005	0.023	0.009	0.013	0.018
Plutonium-239/240	pCi/g	0.002	0.012	0.039	0	0.005	0.009	.	.	.	-0.005	0.00766	0.0331
Potassium-40	pCi/g	8.216	10.56	12.14	9.382	11.14	12.76	.	.	.	0.159	14.5	38.2
Radium-226	pCi/g	0.313	0.452	0.719	0.522	0.639	0.715	.	.	.	0.298	0.643	1.16
Radium-228	pCi/g	0.559	0.765	1.061	0.7	0.809	1.191
Strontium-90	pCi/g	0.009	0.028	0.052	0.006	0.012	0.015	.	.	.	0.00797	0.0727	0.432
Thorium-228	pCi/g	0.218	0.379	0.59	0.458	0.483	0.616	.	.	.	0.529	0.986	1.58
Thorium-230	pCi/g	0.061	0.357	0.914	0.2	0.265	0.333
Thorium-232	pCi/g	0.172	0.382	0.538	0.204	0.541	0.733	.	.	.	0.468	0.9205	1.58
Uranium-233/234	pCi/g	0.073	0.151	0.287	0.686	0.981	1.19	.	.	.	0.286	0.6995	1.51
Uranium-235	pCi/g	0	0.008	0.011	0.019	0.039	0.051	.	.	.	0.00972	0.0278	0.0663
Uranium-238	pCi/g	0.096	0.136	0.265	0.585	0.731	0.878	.	.	.	0.297	0.682	1.23

Table 4-5. Summary of Reference Site Samples, Washington State Background, Area Background for Inorganic and Radionuclides. (2 Pages)

Analyte name	Units	Reference Sites						Washington State Background			Area Background		
		Upland			Riparian			Minimum	Median	Maximum	Minimum	Median	Maximum
		Minimum mean	Median mean	Maximum mean	Minimum mean	Median mean	Maximum mean						
Aluminum	mg/kg	4410	7596	11680	7834	8370	10620	5670	19402	84900	3940	7520	28800
Antimony	mg/kg	0.174	0.174	0.31	0.607	0.77	0.77	.	.	.	11.8	15.7	29.1
Arsenic	mg/kg	1.78	2.781	4.08	4.2	6.6	9.087	0.5	2.9	28.6	3	3.9	27.7
Barium	mg/kg	51.46	84.48	130.4	77.76	80.4	118	.	.	.	45.2	86.7	294
Beryllium	mg/kg	0.312	0.496	0.73	0.266	0.298	0.401	0.1	0.64	2.79	0.46	1.1	2.5
Bismuth	mg/kg	0.295	0.398	0.555	0.411	0.475	0.475
Boron	mg/kg	0.314	1.54	2.52	0.494	1.259	2.07
Cadmium	mg/kg	0.033	0.148	0.176	0.718	0.9	1.62	0.1	0.38	5	0.61	0.66	2.9
Calcium	mg/kg	2952	5200	8392	3914	4538	5226	.	.	.	2880	9060	86600
Chromium	mg/kg	5.06	9.884	13.96	15.62	20.58	22.52	2.555	18.6	235	2.9	10.7	33.2
Cobalt	mg/kg	6.08	8.6	12.18	5.98	6.72	9.227	.	.	.	5.7	11.6	26
Copper	mg/kg	10.74	15.48	21.38	16.84	24.62	33.98	4	17	243.5	8.1	14	40.3
Iron	mg/kg	17780	23110	28120	17860	18220	23674	5025	21650	112500	12300	24100	53600
Lead	mg/kg	3.82	5.918	8.72	14.78	26.96	56.51	2	7.11	207.5	1.1	5.4	74.1
Lithium	mg/kg	4.4	7.4	10.94	9.08	11.36	12.01	.	.	.	34	34	38.2
Magnesium	mg/kg	3736	4242	6110	4212	4854	5684	.	.	.	2900	4810	10500
Manganese	mg/kg	284	392.6	549.6	227.6	339.4	537.1	78	507.25	2750	196	379	1100
Mercury	mg/kg	0.012	0.015	0.015	0.036	0.054	0.087	0.004	0.0245	0.185	0.1	0.16	3.8
Molybdenum	mg/kg	0.161	0.263	0.532	0.479	0.638	0.795	.	.	.	2	2	4.3
Nickel	mg/kg	7.98	10.7	16.36	16.32	18.94	22.36	2.15	16.5	244.5	6.5	12.1	31.3
Phosphorus	mg/kg	674.6	890.2	1186	720.6	778.2	941.5
Potassium	mg/kg	840	1605	2460	828	946.2	1061	.	.	.	851	1215	7900
Selenium	mg/kg	.	.	.	0.395	0.441	0.848	0.09	0.5	7.6	0.96	5	6
Silicon	mg/kg	209	304.6	410.3	300.4	315	347.4	.	.	.	5.2	15.3	682
Silver	mg/kg	.	.	.	0.064	0.064	0.098	.	.	.	1.4	1.4	14.6
Sodium	mg/kg	97.1	169.6	227.2	148	185.6	235.8	.	.	.	101	307	6060
Strontium (elemental)	mg/kg	16.5	23.67	29.64	24.34	29.46	34.22
Thallium	mg/kg	.	.	.	0.62	0.64	0.85	.	.	.	0.61	3.7	3.7
Tin	mg/kg	0.687	0.687	0.749	0.96	2.277	4.591
Uranium (inorganic)	mg/kg	0.783	1.14	1.14	1.23	1.891	3.03

Table 4-6. Usability Codes and Reasons for Data Indicated as Not Useable in the Risk Assessment. (2 Pages)

Usability Code	RCBRA Not-Usable Reason	Count of Results
1	Missing result with detect_status = true <u>Description:</u> Result was qualified as “detected” but no value was reported.	10
2	Inappropriate analytical method <u>Description:</u> Method used for analysis was inappropriate for the analyte evaluated.	425
3	Non-standard units <u>Description:</u> Units reported were non-standard and could not be converted (e.g., pCi/sample)	124
4	Non-possible result <u>Description:</u> Result is impossible (e.g., negative result for non-radionuclide)	24
5	Bad coordinates <u>Description:</u> GPS coordinates lacking or erroneous. Insufficient locational information.	0 (Coordinates extrapolated)
6	Decayed results <u>Description:</u> Pertains to results from the 100-B/C Pilot Project that were mathematically decayed to the analysis dates for that project. Decayed results not applicable to RCBRA.	8482
7	Rejected data <u>Description:</u> Analytical data qualified as “R” by laboratory, reviewer, or validator.	207
8	Missing units <u>Description:</u> Analytical result did not report units.	6098
9	Equipment blank <u>Description:</u> Result is for laboratory equipment blank, not RCBRA investigation sample.	1029
10	Mixed media sample <u>Description:</u> Sample media is “mixed media,” such as concrete or paint chip. Mixed media not relevant to risk investigation.	3029
11	Sample has another result for same analyte using a more preferred analytical method <u>Description:</u> Analytical results for analyte generated by more than one method. Preferred method is retained in the database.	15897
12	Lab not authorized to perform this analysis for Hanford samples <u>Description:</u> Result was generated by contract laboratory outside scope of work order. Sample underwent subsequent analysis at qualified laboratory.	9

Table 4-6. Usability Codes and Reasons for Data Indicated as Not Useable in the Risk Assessment. (2 Pages)

Usability Code	RCBRA Not-Usable Reason	Count of Results
13	ISRM treatment sample <u>Description:</u> Result is reported treatment for ISRM injection well. Result not comparable to groundwater monitoring or sample data.	7672
14	Units not rcbra usable <u>Description:</u> Result reported as density.	1
15	Sodium dithionite reported in M (molar) units; not useable for risk assessment <u>Description:</u> Reported results are in molar units, therefore not useable for the risk assessment.	1013
16	Duplicate sample reported by a less preferred data source <u>Description:</u> Result was reported by more than one data source. Result from preferred data source was retained as useable in database.	7036
17	Additional screen size used for 9 samples only <u>Description:</u> Soil sieve was used inconsistently for particle size determination and therefore not considered in the analysis.	9
18	Could not determine invertebrate type from information provided <u>Description:</u> Invertebrate type as terrestrial or aquatic could not be determined from the data provided. This level of specificity is required for trophic modeling.	241
19	Uncertainty due to ongoing investigation	36
20	In-process sample - location was subsequently re-excavated	226
21	Condensate result - not useable in risk analysis	15
22	Sample has another result for same analyte which differs only in value reported - both results disqualified	4
23	Sample has another result for same analyte with more complete informaion	988
24	Sample has another identical result for same analyte	1480
25	Units not comparable to existing results	6
26	Effluent result - not useable in risk analysis	919
Total		54979

GPS = global positioning system

ISRM = In situ redox manipulation

RCBRA = River Corridor Baseline Risk Assessment

Table 4-7. Constituents Detected at Least Once in Biotic or Abiotic Media from RCBRA, 100-NR-2, or 100-B/C Pilot Project Data Sets. (2 Pages)

Aluminum	Acenaphthene	Heptachlor epoxide
Antimony	Acenaphthylene	Indeno(1,2,3-cd)pyrene
Arsenic	Acetone	Methoxychlor
Barium	Aldrin	Methylenechloride
Beryllium	Alpha-BHC	Naphthalene
Bismuth	alpha-Chlordane	Pentachlorophenol
Boron	Anthracene	Phenanthrene
Cadmium	Aroclor-1248	Phenol
Calcium	Aroclor-1254	Picloram
Calculated Total Uranium	Aroclor-1260	Pyrene
Chromium	Aroclor-1262	Toluene
Cobalt	Benzo(a)anthracene	Beryllium-7
Copper	Benzo(a)pyrene	Carbon-14
Hexavalent Chromium	Benzo(b)fluoranthene	Cesium-137
Iron	Benzo(ghi)perylene	Cobalt-60
Lead	Benzo(k)fluoranthene	Curium-242
Lithium	beta-1,2,3,4,5,6-Hexachlorocyclohexane	Curium-244
Magnesium	Bis(2-ethylhexyl) phthalate	Europium-152
Manganese	Carbazole	Europium-154
Mercury	Chrysene	Europium-155
Molybdenum	Dalapon	Gross alpha
Nickel	Delta-BHC	Gross beta
Phosphorus	Di-n-butylphthalate	Lead-212
Potassium	Dibenz[a,h]anthracene	Lead-214
Selenium	Dibenzofuran	Manganese-54
Silicon	Dicamba	Nickel-63
Silver	Dichlorodiphenyldichloroethane	Plutonium-238
Sodium	Dichlorodiphenyldichloroethylene	Plutonium-239/240
Strontium (elemental)	Dichlorodiphenyltrichloroethane	Potassium-40
Thallium	Dichloroprop	Radium-226
Tin	Dieldrin	Radium-228
Uranium (inorganic)	Diethylphthalate	Strontium-90

Table 4-7. Constituents Detected at Least Once in Biotic or Abiotic Media from RCBRA, 100-NR-2, or 100-B/C Pilot Project Data Sets. (2 Pages)

Vanadium	Endosulfan I	Technetium-99
Zinc	Endosulfan II	Thorium-228
Zirconium	Endosulfan sulfate	Thorium-230
2-(2,4,5-Trichlorophenoxy)propionic acid	Endrin	Thorium-232
2-Methylnaphthalene	Endrin aldehyde	Thorium-234
2-secButyl-4,6-dinitrophenol(DNBP)	Endrin ketone	Uranium-233/234
2,4-Dichlorophenoxyacetic acid	Fluoranthene	Uranium-235
2,4,5-Trichlorophenoxyacetic acid	Fluorene	Uranium-238
3+4 Methylphenol (cresol, m+p)	gamma-Chlordane	Zirconium/Niobium-95
4-(2,4-Dichlorophenoxy)butanoic acid	Heptachlor	

RCBRA = River Corridor Baseline Risk Assessment

Table 4-8. Detected Analytes in RCBRA Upland and Riparian Multi-increment Soil Data. (2 Pages)

2-(2,4,5-Trichlorophenoxy)propionic acid	Bis(2-ethylhexyl) phthalate	Endrin	Phenol
2,4,5-Trichlorophenoxyacetic acid	Bismuth	Endrin aldehyde	Phosphorus
2,4-Dichlorophenoxyacetic acid	Boron	Endrin ketone	Picloram
2-Methylnaphthalene	Cadmium	Europium-152	Plutonium-238
2-secButyl-4,6-dinitrophenol(DNBP)	Calcium	Europium-155	Plutonium-239/240
3+4 Methylphenol (cresol, m+p)	Carbazole	Fluoranthene	Potassium
4-(2,4-Dichlorophenoxy)butanoic acid	Carbon-14	Fluorene	Potassium-40
Acenaphthene	Cesium-137	Fluoride	Pyrene
Acenaphthylene	Chromium	gamma-Chlordane	Radium-226
Aldrin	Chrysene	Heptachlor	Radium-228
Alpha-BHC	Cobalt	Heptachlor epoxide	Selenium

**Table 4-8. Detected Analytes in RCBRA Upland and Riparian Multi-increment Soil Data.
(2 Pages)**

alpha-Chlordane	Cobalt-60	Hexavalent Chromium	Silicon
Aluminum	Copper	Indeno(1,2,3-cd)pyrene	Sodium
Anthracene	Dalapon	Iron	Strontium (elemental)
Antimony	Delta-BHC	Lead	Strontium-90
Aroclor-1248	Dibenz[a,h]anthracene	Lithium	Thallium
Aroclor-1254	Dibenzofuran	Magnesium	Thorium-228
Aroclor-1260	Dicamba	Manganese	Thorium-230
Aroclor-1262	Dichlorodiphenyldichloroethane	Mercury	Thorium-232
Arsenic	Dichlorodiphenyldichloroethylene	Methoxychlor	Thorium-234
Barium	Dichlorodiphenyltrichloroethane	Molybdenum	Tin
Benzo(a)anthracene	Dichloroprop	Naphthalene	Uranium (inorganic)
Benzo(a)pyrene	Dieldrin	Nickel	Uranium-233/234
Benzo(b)fluoranthene	Diethylphthalate	Nitrate	Uranium-235
Benzo(ghi)perylene	Di-n-butylphthalate	Nitrate	Uranium-238
Benzo(k)fluoranthene	Endosulfan I	Nitrite	Vanadium
Beryllium	Endosulfan II	Pentachlorophenol	Zinc
beta-1,2,3,4,5,6-Hexachlorocyclohexane	Endosulfan sulfate	Phenanthrene	

RCBRA = River Corridor Baseline Risk Assessment

Table 4-9. Detected Analytes in CVP/RSVP Soil Data. (2 Pages)

1,1,2,2-Tetrachloroethane	Boron	Hexadecanoic acid (9CI)	Radium-224
2-Butanone	Butylbenzylphthalate	Hexavalent Chromium	Radium-226
2-Butoxyethanol	Cadmium	Indeno(1,2,3-cd)pyrene	Radium-228
2-Methylnaphthalene	Calcium	Iron	Ruthenium-106
Acenaphthene	Carbazole	Isophorone	Selenium
Acenaphthylene	Carbon-14	Lead	Silicon
Acetone	Cesium-134	Lead-212	Sodium
Actinium-228	Cesium-137	Lead-214	Silver
Alpha-BHC	Chromium	Lithium	Strontium-90

Table 4-9. Detected Analytes in CVP/RSVP Soil Data. (2 Pages)

Aluminum	Chrysene	Magnesium	Technetium-99
Americium-241	Cobalt	Manganese	Tetrachloroethene
Anthracene	Cobalt-60	Manganese-54	Thallium
Antimony	Copper	Mercury	Thorium-228
Aroclor-1242	Dibenz[a,h]anthracene	Methylenechloride	Thorium-230
Aroclor-1248	Dibenzofuran	Molybdenum	Thorium-232
Aroclor-1254	Dichlorodiphenyldichloroethene	Naphthalene	Thorium-234
Aroclor-1260	Dieldrin	Nickel	Toluene
Arsenic	Diethyl ether	Nickel-63	TPH
Barium	Di-n-butylphthalate	Nitrate	Trichloroethene
Barium-133	Endrin aldehyde	Octacosane	Trichloromonofluoromethane
Benzene	Ethylene glycol	Octadecanoic acid	Tritium
Benzo(a)anthracene	Europium-152	Pentachlorophenol	Uranium (inorganic)
Benzo(a)pyrene	Europium-154	Phenanthrene	Uranium-233/234
Benzo(b)fluoranthene	Europium-155	Phenol	Uranium-235
Benzo(ghi)perylene	Fluoranthene	Plutonium-238	Uranium-238
Benzo(k)fluoranthene	Fluorene	Plutonium-239/240	Vanadium
Beryllium	Fluoride	Plutonium-241	Zinc
Beryllium-7	gamma-Chlordane	Potassium	
beta-1,2,3,4,5,6-Hexachlorocyclohexane	Heptachlor epoxide	Potassium-40	
Bis(2-ethylhexyl) phthalate	Heptacosane	Pyrene	

CVP = closeout verification package

RSVP = remaining sites verification package

TPH = total petroleum hydrocarbon

Table 4-10. Detected Analytes in Groundwater Well Data.

1,2-Dichloroethane	Carbon tetrachloride	Heptachlor epoxide	Silicon
1,4-Dichlorobenzene	Carbon-14	Hexavalent Chromium	Silver
1-Butanol	Chloroform	Iron	Sodium
2-Butanone	Chromium	Lithium	Sodium dithionite
2-Methylnaphthalene	cis-1,2-Dichloroethylene	Magnesium	Strontium (elemental)
Acetone	Cobalt	Manganese	Strontium-90
Alpha-BHC	Copper	Methoxychlor	Technetium-99
Aluminum	Delta-BHC	Methyl isobutyl ketone	Tetrachloroethene
Antimony	Dichlorodiphenyldichloroethane	Methylenechloride	Thallium
Aroclor-1254	Dichlorodiphenyldichloroethylene	Molybdenum	Tin
Arsenic	Dichlorodiphenyltrichloroethane	Nickel	Total petroleum hydrocarbons
Barium	Dieldrin	Nitrate	Trichloroethene
Benzo(ghi)perylene	Di-n-butylphthalate	Nitrite	Tritium
Beryllium	Endosulfan sulfate	Oil and grease	Uranium (inorganic)
Bis(2-ethylhexyl) phthalate	Endrin	Orthophosphate	Uranium-233/234
Boron	Endrin aldehyde	Phosphorus	Uranium-235
Bromide	Endrin ketone	Potassium	Uranium-238
Cadmium	Fluoride	Radium-228	Vanadium
Calcium	gamma-Chlordane	Selenium	Zinc
Carbon disulfide	Heptachlor	Silica	

Table 4-11. RCBRA Groundwater COPCs Greater than Background of Reference Site Concentrations. (2 Pages)

<i>Inorganics</i>	
Aluminum	Manganese
Antimony	Molybdenum
Arsenic	Nickel
Barium	Phosphorus
Beryllium	Selenium
Boron	Silver

**Table 4-11. RCBRA Groundwater COPCs Greater than
Background of Reference Site Concentrations. (2 Pages)**

Cadmium	Strontium (elemental)
Chromium	Thallium
Cobalt	Tin
Copper	Titanium
Hexavalent Chromium	Uranium (inorganic)
Iron	Vanadium
Lithium	Zinc
Organics	
2-Methylnaphthalene	Dichlorodiphenyltrichloroethane
Aldrin	Dieldrin
Alpha-BHC	Endosulfan sulfate
Aroclor-1254	Endrin
Aroclor-1260	Endrin aldehyde
Benzo(ghi)perylene	Endrin ketone
Bis(2-ethylhexyl) phthalate	gamma-Chlordane
Delta-BHC	Heptachlor
Di-n-butylphthalate	Heptachlor epoxide
Dichlorodiphenyldichloroethane	Methoxychlor
Dichlorodiphenyldichloroethylene	
Radionuclides	
Carbon-14	Tritium
Radium-228	Uranium-233/234
Strontium-90	Uranium-235
Thorium-228	Uranium-238
Thorium-230	
Others	
Bromide	Nitrogen in Nitrate
Chloride	Nitrogen in Nitrite
Fluoride	

COPC = contaminant of potential concern
RCBRA = River Corridor Baseline Risk Assessment

Table 4-12. Comparison of RCBRA Soils to Hanford Area Background and Washington State Background Values. (4 Pages)

Analyte Type	Analyte	Groups	Data Sets List (in Ascending Order) ^a					Mean Ranks of Data Sets ^b					Diff ^c	Conclusion ^d
			RBF	RRP	RNS	-	-							
INORGANIC	Bismuth	3	RBF	RRP	RNS	-	-	39	40.5	42	-	-	no	RBF=RRP=RNS
INORGANIC	Boron	3	RBF	RRP	RNS	-	-	31.9	37.32	52.92	-	-	yes	RBF=RRP<RNS
INORGANIC	Calculated Total Uranium	3	RNS	RBF	RRP	-	-	25.44	25.68	62.88	-	-	yes	RNS=RBF<RRP
INORGANIC	Hexavalent Chromium	3	RRP	RNS	RBF	-	-	39.71	43.66	46.94	-	-	no	all equal
INORGANIC	Phosphorus	3	RNS	RRP	RBF	-	-	32.88	36.3	53.16	-	-	yes	RNS=RRP<RBF
INORGANIC	Strontium (elemental)	3	RBF	RNS	RRP	-	-	22.7	36.34	58.8	-	-	yes	RBF<RNS<RRP
INORGANIC	Tin	3	RBF	RNS	RRP	-	-	33.86	34.14	51.33	-	-	yes	RBF=RNS<RRP
RAD	Uranium (inorganic)	3	RNS	RBF	RRP	-	-	33.58	33.92	51.75	-	-	yes	RNS=RBF<RRP
RAD	Cobalt-60	3	RBF	RNS	RRP	-	-	20.26	42.06	51.68	-	-	yes	RBF<RNS=RRP
RAD	Europium-152	3	RBF	RNS	RRP	-	-	18.42	37.22	58.36	-	-	yes	RBF<RNS<RRP
RAD	Europium-155	3	RBF	RNS	RRP	-	-	22.62	33.68	57.7	-	-	yes	RBF<RNS<RRP
RAD	Plutonium-238	3	RBF	RRP	RNS	-	-	30.06	30.15	31.08	-	-	no	all equal
RAD	Radium-228	3	RBF	RNS	RRP	-	-	24.5	44.7	44.8	-	-	yes	RBF<RNS=RRP
RAD	Thorium-230	3	RBF	RRP	RNS	-	-	29.56	34.06	50.38	-	-	yes	RBF=RRP<RNS
ORGANIC	3+4 Methylphenol (cresol, m+p)	3	RBF	RNS	RRP	-	-	36.5	37.96	39.54	-	-	no	all equal
ORGANIC	Dibenz[a,h]anthracene	3	RBF	RNS	RRP	-	-	24.04	26.96	63	-	-	yes	RBF=RNS<RRP
ORGANIC	Pyrene	3	RNS	RRP	RBF	-	-	36.4	38	39.6	-	-	yes	[RNS=(RRP)=RBF]
OTHER	Nitrogen in ammonia	3	RBF	RNS	RRP	-	-	7	7.2	9.8	-	-	no	all equal
OTHER	Nitrogen, Kjeldahl total	3	RBF	RNS	RRP	-	-	5	6.2	12.8	-	-	yes	RBF=RNS<RRP
INORGANIC	Antimony	4	RBF	RNS	RRP	HAB	-	75.5	75.5	81.57	86.36	-	no	all equal
INORGANIC	Barium	4	RBF	HAB	RRP	RNS	-	62.5	118.16	141.14	154.4	-	yes	RBF<HAB=RRP=RNS
INORGANIC	Cobalt	4	RRP	RBF	RNS	HAB	-	58.13	68.28	103.42	143.3	-	yes	[RRP=(RBF)=RNS]<HAB
INORGANIC	Lithium	4	RBF	RNS	RRP	HAB	-	42.82	66.46	88.93	89.2	-	yes	RBF<RNS<RRP=HAB
INORGANIC	Molybdenum	4	RNS	RBF	HAB	RRP	-	66.02	78.06	92.25	103.93	-	yes	RNS=RBF<HAB<RRP
INORGANIC	Silver	4	RBF	RNS	RRP	HAB	-	93	93.1	94.14	131.88	-	yes	RBF=RNS=RRP<HAB
INORGANIC	Thallium	4	RBF	RNS	RRP	HAB	-	113.42	113.52	119.83	119.86	-	no	all equal
INORGANIC	Vanadium	4	RRP	RNS	HAB	RBF	-	78.93	120	125.11	141.46	-	yes	[RRP=(RNS)=HAB=RBF]
RAD	Cesium-137	4	RNS	RBF	RRP	HAB	-	42.6	52.84	75.78	140.4	-	yes	RNS=RBF=RRP<HAB
RAD	Plutonium-239/240	4	RRP	RNS	RBF	HAB	-	47.45	50.64	59.76	112.94	-	yes	RRP=RNS=RBF<HAB
RAD	Potassium-40	4	RBF	RNS	RRP	HAB	-	38.88	79.36	88.56	230.86	-	yes	RBF=RNS=RRP<HAB
RAD	Radium-226	4	RBF	RNS	HAB	RRP	-	30.96	80.72	115.58	117.84	-	yes	RBF<[RNS<HAB=RRP]

Table 4-12. Comparison of RCBRA Soils to Hanford Area Background and Washington State Background Values. (4 Pages)

Analyte Type	Analyte	Groups	Data Sets List (in Ascending Order) ^a					Mean Ranks of Data Sets ^b					Diff ^c	Conclusion ^d
			RRP	RBF	RNS	HAB	-							
RAD	Strontium-90	4	RRP	RBF	RNS	HAB	-	42.74	46.48	55.26	101.56	-	yes	RRP=RBF=RNS<HAB
RAD	Thorium-228	4	RBF	RNS	RRP	HAB	-	18.7	49.72	51.88	96.79	-	yes	RBF<RNS=RRP<HAB
RAD	Thorium-232	4	RBF	RNS	RRP	HAB	-	23.26	48.64	52.16	96.66	-	yes	RBF<RNS=RRP<HAB
RAD	Uranium-233/234	4	RBF	RNS	HAB	RRP	-	21.02	30.68	101.95	134.26	-	yes	RBF=RNS<HAB<RRP
RAD	Uranium-235	4	RBF	RNS	RRP	HAB	-	40.26	40.86	51.3	74.12	-	yes	RBF=RNS<RRP<HAB
RAD	Uranium-238	4	RNS	RBF	HAB	RRP	-	25.54	25.9	107.08	115.64	-	yes	RNS=RBF<HAB=RRP
ORGANIC	2-Methylnaphthalene	4	RBF	RNS	RRP	HAB	-	36.5	37.96	39.54	77	-	yes	RBF=RNS=RRP<HAB
ORGANIC	Acenaphthene	4	RBF	RNS	RRP	HAB	-	24.04	26.96	63	77	-	yes	RBF=RNS<RRP<HAB
ORGANIC	Acenaphthylene	4	RBF	RNS	RRP	HAB	-	24.04	26.96	63	77	-	yes	RBF=RNS<RRP<HAB
ORGANIC	Aldrin	4	RNS	HAB	RBF	RRP	-	39.3	39.5	39.5	39.7	-	yes	[RNS=(HAB=RBF)=RRP]
ORGANIC	Alpha-BHC	4	RNS	RBF	RRP	HAB	-	33	40.5	40.5	77	-	yes	RNS<RBF=RRP<HAB
ORGANIC	Anthracene	4	RBF	HAB	RRP	RNS	-	37.12	39.5	39.5	41.88	-	no	all equal
ORGANIC	Aroclor-1248	4	RBF	RNS	RRP	HAB	-	38	38	38	77	-	yes	RBF=RNS=RRP<HAB
ORGANIC	Aroclor-1254	4	RBF	RNS	RRP	HAB	-	38	38	38	77	-	yes	RBF=RNS=RRP<HAB
ORGANIC	Aroclor-1260	4	RBF	RNS	RRP	HAB	-	38	38	38	77	-	yes	RBF=RNS=RRP<HAB
ORGANIC	Benzo(a)anthracene	4	RNS	HAB	RRP	RBF	-	38.5	39.5	39.5	40.5	-	no	all equal
ORGANIC	Benzo(a)pyrene	4	RNS	HAB	RRP	RBF	-	38.5	39.5	39.5	40.5	-	no	all equal
ORGANIC	Benzo(b)fluoranthene	4	RNS	HAB	RRP	RBF	-	38.5	39.5	39.5	40.5	-	no	all equal
ORGANIC	Benzo(ghi)perylene	4	RNS	HAB	RRP	RBF	-	39	39.5	39.5	40	-	no	all equal
ORGANIC	Benzo(k)fluoranthene	4	HAB	RBF	RNS	RRP	-	39.5	39.5	39.5	39.5	-	no	all equal
ORGANIC	Bis(2-ethylhexyl) phthalate	4	RNS	HAB	RRP	RBF	-	36.2	39.5	41.14	41.16	-	no	all equal
ORGANIC	Carbazole	4	RBF	RNS	RRP	HAB	-	36.5	37.96	39.54	77	-	yes	RBF=RNS=RRP<HAB
ORGANIC	Chrysene	4	RNS	HAB	RRP	RBF	-	37.28	39.5	39.5	41.72	-	yes	[RNS=(HAB=RRP)=RBF]
ORGANIC	Delta-BHC	4	RNS	RBF	HAB	RRP	-	38.9	39	39.5	40.6	-	no	all equal
ORGANIC	Di-n-butylphthalate	4	RNS	RBF	HAB	RRP	-	38.92	39.48	39.5	40.1	-	no	all equal
ORGANIC	Dibenzofuran	4	RBF	RNS	RRP	HAB	-	36.5	37.96	39.54	77	-	yes	RBF=RNS=RRP<HAB
ORGANIC	Dichlorodiphenyldichloroethane	4	HAB	RRP	RBF	RNS	-	38.5	38.5	40.06	40.06	-	no	all equal
ORGANIC	Dichlorodiphenyldichloroethylene	4	RNS	RBF	HAB	RRP	-	34.24	36	39.5	48.26	-	yes	[RNS=RBF=(HAB)=RRP]
ORGANIC	Dichlorodiphenyltrichloroethane	4	RNS	RBF	HAB	RRP	-	38.5	39	39.5	41	-	no	all equal
ORGANIC	Dieldrin	4	RNS	RBF	RRP	HAB	-	33	40.5	40.5	77	-	yes	RNS<RBF=RRP<HAB
ORGANIC	Diethylphthalate	4	RBF	RNS	RRP	HAB	-	37	38.44	38.56	77	-	yes	RBF=RNS=RRP<HAB
ORGANIC	Endosulfan I	4	RNS	RBF	HAB	RRP	-	37.8	38.5	39.5	42.2	-	yes	[RNS=(RBF=HAB)=RRP]

Table 4-12. Comparison of RCBRA Soils to Hanford Area Background and Washington State Background Values. (4 Pages)

Analyte Type	Analyte	Groups	Data Sets List (in Ascending Order) ^a					Mean Ranks of Data Sets ^b					Diff ^c	Conclusion ^d
			RNS	RBF	RRP	HAB	-							
ORGANIC	Endosulfan II	4	RNS	RBF	RRP	HAB	-	33	40.5	40.5	77	-	yes	RNS<RBF=RRP<HAB
ORGANIC	Endosulfan sulfate	4	RNS	HAB	RBF	RRP	-	39.4	39.5	39.5	39.6	-	no	all equal
ORGANIC	Endrin aldehyde	4	RNS	RRP	HAB	RBF	-	38.8	39.2	39.5	40.5	-	no	all equal
ORGANIC	Endrin ketone	4	RNS	RBF	RRP	HAB	-	33	40.5	40.5	77	-	yes	RNS<RBF=RRP<HAB
ORGANIC	Fluoranthene	4	RNS	HAB	RRP	RBF	-	39	39.5	39.5	40	-	no	all equal
ORGANIC	Fluorene	4	RNS	HAB	RRP	RBF	-	39.48	39.5	39.5	39.52	-	no	all equal
ORGANIC	Heptachlor	4	RNS	HAB	RRP	RBF	-	39.3	39.5	39.58	39.62	-	no	all equal
ORGANIC	Heptachlor epoxide	4	RNS	RBF	RRP	HAB	-	33	40.5	40.5	77	-	yes	RNS<RBF=RRP<HAB
ORGANIC	Indeno(1,2,3-cd)pyrene	4	RNS	HAB	RRP	RBF	-	39	39.5	39.5	40	-	no	all equal
ORGANIC	Methoxychlor	4	RBF	HAB	RNS	RRP	-	38.5	39.5	39.98	40.02	-	no	all equal
ORGANIC	Naphthalene	4	HAB	RBF	RNS	RRP	-	39.5	39.5	39.5	39.5	-	no	all equal
ORGANIC	Pentachlorophenol	4	RBF	HAB	RRP	RNS	-	39.1	39.5	39.5	39.9	-	no	all equal
ORGANIC	Phenanthrene	4	RBF	HAB	RRP	RNS	-	39.46	39.5	39.5	39.54	-	no	all equal
ORGANIC	Phenol	4	RBF	HAB	RNS	RRP	-	39.36	39.5	39.5	39.64	-	yes	all equal
ORGANIC	alpha-Chlordane	4	RNS	RBF	RRP	HAB	-	33	40.5	40.5	77	-	yes	RNS<RBF=RRP<HAB
ORGANIC	beta-1,2,3,4,5,6-Hexachlorocyclohexane	4	RNS	HAB	RBF	RRP	-	39.4	39.5	39.5	39.6	-	no	all equal
ORGANIC	gamma-Chlordane	4	RNS	HAB	RBF	RRP	-	39.1	39.5	39.5	39.9	-	yes	[RNS=(HAB=RBF)=RRP]
OTHER	Chloride	4	RBF	RNS	RRP	HAB	-	4	7.4	70.2	89.61	-	yes	[RBF=RNS=(RRP)=HAB]
OTHER	Fluoride	4	RBF	RNS	RRP	HAB	-	7.4	29.8	32.3	90	-	yes	RBF=RNS=RRP<HAB
OTHER	Nitrogen in Nitrate	4	RBF	RNS	HAB	RRP	-	75.98	93.48	111.89	196.87	-	yes	[RBF=(RNS)=HAB]<RRP
OTHER	Nitrogen in Nitrite	4	RNS	RBF	RRP	HAB	-	74.5	75.8	79.7	85.27	-	no	all equal
INORGANIC	Aluminum	5	RBF	HAB	RRP	RNS	WAB	48.26	130.89	166.27	172.06	305.17	yes	RBF<HAB=RRP=RNS<WAB
INORGANIC	Arsenic	5	RBF	WAB	RNS	HAB	RRP	64.42	156.8	171.1	215.34	321.19	yes	RBF<[WAB=(RNS)=HAB]<RRP
INORGANIC	Beryllium	5	RRP	RBF	RNS	WAB	HAB	67.4	113.04	145.54	181.18	279.48	yes	RRP=RBF=RNS=WAB<HAB
INORGANIC	Cadmium	5	RBF	RNS	HAB	WAB	RRP	123.54	144.78	171.81	217.42	348.96	yes	RBF=RNS=HAB<WAB<RRP
INORGANIC	Chromium	5	RBF	HAB	RNS	WAB	RRP	72.88	145.44	162.54	262.57	296.34	yes	RBF<HAB=RNS<WAB=RRP
INORGANIC	Copper	5	RBF	HAB	RNS	WAB	RRP	168.28	169.86	200.34	211.56	339.3	yes	RBF=HAB=RNS=WAB<RRP
INORGANIC	Iron	5	RRP	RBF	WAB	HAB	RNS	166.43	186.72	197.58	216.7	222.8	no	all equal
INORGANIC	Lead	5	RBF	HAB	WAB	RNS	RRP	109.1	165.36	212.98	216.64	382.79	yes	RBF=HAB<WAB=RNS<RRP
INORGANIC	Manganese	5	RBF	HAB	RRP	RNS	WAB	130.52	166.94	206.29	239.28	241.75	yes	[RBF=HAB=(RRP)=RNS=WAB]
INORGANIC	Mercury	5	RBF	RNS	WAB	HAB	RRP	128.2	130.18	188.38	221.12	252.74	yes	RBF=RNS<WAB<HAB=RRP
INORGANIC	Nickel	5	RBF	HAB	RNS	WAB	RRP	70.82	164.86	186.68	238.7	310.24	yes	RBF<[HAB=(RNS)=WAB]<RRP

Table 4-12. Comparison of RCBRA Soils to Hanford Area Background and Washington State Background Values. (4 Pages)

Analyte Type	Analyte	Groups	Data Sets List (in Ascending Order) ^a					Mean Ranks of Data Sets ^b					Diff ^c	Conclusion ^d
INORGANIC	Selenium	5	RBF	RNS	RRP	WAB	HAB	162.9	165.76	177.94	195.12	195.25	yes	[RBF=RNS=(RRP)=WAB=HAB)
INORGANIC	Zinc	5	RBF	RNS	HAB	WAB	RRP	137.18	171.96	189.52	192.39	387.84	yes	RBF=RNS=HAB=WAB<RRP

- = not applicable; no group

^adata sets list (in ascending order) = codes for datasets listed in ascending order of mean rank analyte concentration

Background and Reference Dataset Groups:

HAB = Hanford Area Background

WAB = Washington Background

RBF = RCBRA Backfill

RNS = RCBRA Native Soil

RRP = RCBRA Riparian

^bThe mean of ranks of the data sets - group identity matches order in data sets list

^cdiff=yes if there is at least one significant difference between groups; = no, otherwise

^dconclusion based on a Tukey multiple comparison of data ranks

= - groups not significantly different

< group significantly different - group(s) on left has significantly smaller concentration than group(s) on right

[], () - brackets enclose groups not significantly different; occurs when multiple comparisons result in overlapping conclusions

eg: Arsenic conclusion RBF<[WAB=(RNS)=HAB]<RRP implies:

RBF is significantly smaller than all other groups

WAB is not significantly different from RNS, but is significantly smaller than HAB

RNS is significantly larger than WAB, but is not significantly different from HAB

RRP is significantly larger than all other groups

RCBRA = River Corridor Baseline Risk Assessment

Table 4-13. RCBRA Soil COPCs Greater than Background of
Reference Site Concentrations. (2 Pages)

<i>Inorganics</i>	
Aluminum	Lead
Antimony	Lithium
Arsenic	Mercury
Barium	Molybdenum
Beryllium ^a	Nickel
Bismuth	Phosphorus
Boron	Selenium ^a
Cadmium	Silver
Calculated Total Uranium	Strontium (elemental)
Chromium	Thallium
Cobalt ^a	Tin
Copper	Uranium (inorganic)
Hexavalent Chromium	Zinc
<i>Organics</i>	
2-(2,4,5-Trichlorophenoxy)propionic acid	Dibenz[a,h]anthracene
2,4,5-Trichlorophenoxyacetic acid	Dibenzofuran
2,4-Dichlorophenoxyacetic acid	Dicamba ^a
2-Methylnaphthalene	Dichlorodiphenyldichloroethane
2-secButyl-4,6-dinitrophenol(DNBP)	Dichlorodiphenyldichloroethylene
3+4 Methylphenol (cresol, m+p)	Dichlorodiphenyltrichloroethane
4-(2,4-Dichlorophenoxy)butanoic acid	Dichloroprop
Acenaphthene	Dieldrin
Acenaphthylene	Diethylphthalate ^a
Aldrin ^a	Di-n-butylphthalate
Alpha-BHC	Endosulfan I
alpha-Chlordane	Endosulfan II
Anthracene	Endosulfan sulfate
Aroclor-1248	Endrin aldehyde
Aroclor-1254	Endrin ketone
Aroclor-1260	Fluoranthene
Aroclor-1262 ^b	Fluorene

**Table 4-13. RCBRA Soil COPCs Greater than Background of
Reference Site Concentrations. (2 Pages)**

Benzo(a)anthracene	gamma-Chlordane
Benzo(a)pyrene	Heptachlor
Benzo(b)fluoranthene	Heptachlor epoxide
Benzo(ghi)perylene	Indeno(1,2,3-cd)pyrene
Benzo(k)fluoranthene	Methoxychlor
beta-1,2,3,4,5,6-Hexachlorocyclohexane	Naphthalene
Bis(2-ethylhexyl) phthalate	Pentachlorophenol ^a
Carbazole	Phenanthrene
Chrysene	Phenol ^a
Dalapon	Picloram ^a
Delta-BHC ^a	Pyrene
Radionuclides	
Carbon-14 ^b	Radium-228
Cesium-137 ^a	Strontium-90 ^a
Cobalt-60	Thorium-228 ^a
Europium-152	Thorium-230
Europium-155	Thorium-232 ^a
Plutonium-238 ^a	Uranium-233/234
Plutonium-239/240 ^a	Uranium-235 ^a
Potassium-40	Uranium-238
Radium-226	
Other	
Chloride ^a	Nitrogen in Nitrite and Nitrate ^b
Fluoride	Nitrogen in ammonia ^a
Nitrogen in Nitrate	Nitrogen, Kjeldahl total ^a
Nitrogen in Nitrite	

^aUncertainty analysis (low detect rate comparison) concludes indistinguishable from background.

^bNo background data for comparison.

COPC = contaminant of potential concern

RCBRA = River Corridor Baseline Risk Assessment

Table 4-14. CVP/RSVP Soil COPCs Greater than Background of Reference Site Concentrations. (2 Pages)

<i>Inorganics</i>	
Aluminum	Lead
Antimony	Lithium
Arsenic	Manganese ^a
Barium	Mercury
Beryllium	Molybdenum
Boron ^b	Nickel
Cadmium	Selenium
Calculated Total Uranium	Silver
Chromium	Thallium
Cobalt ^a	Uranium (inorganic) ^b
Copper	Vanadium ^a
Hexavalent Chromium ^b	Zinc
<i>Organics</i>	
1,1,2,2-Tetrachloroethane ^c	Dichlorodipenyldichloroethylene ^c
2-Butanone ^c	Dichlorodiphenyltrichloroethane ^c
2-Butoxyethanol ^b	Dieldrin ^c
2-Methylnaphthalene ^c	Diethyl ether ^b
Acenaphthene ^c	Endrin aldehyde ^c
Acenaphthylene ^c	Ethylene glycol ^b
Acetone ^c	Fluoranthene ^c
Alpha-BHC ^c	Fluorene ^c
Anthracene ^c	Heptachlor epoxide ^c
Aroclor-1242 ^c	Heptacosane ^b
Aroclor-1248 ^c	Hexadecanoic acid (9CI) ^b
Aroclor-1254 ^c	Indeno(1,2,3-cd)pyrene ^c
Aroclor-1260 ^c	Isophorone ^c
Benzene ^c	Methylenechloride ^c
Benzo(a)anthracene ^c	Naphthalene ^c
Benzo(a)pyrene ^c	Octacosane ^b
Benzo(b)fluoranthene ^c	Octadecanoic acid ^b
Benzo(ghi)perylene ^c	Pentachlorophenol ^c

Table 4-14. CVP/RSVP Soil COPCs Greater than Background of Reference Site Concentrations. (2 Pages)

Benzo(k)fluoranthene ^c	Phenanthrene ^c
Bis(2-ethylhexyl) phthalate ^c	Phenol ^c
Butylbenzylphthalate ^c	Pyrene ^b
Carbazole ^c	Tetrachloroethene ^c
Chrysene ^c	Toluene
Di-n-butylphthalate ^c	Trichloroethene ^c
Di-n-octylphthalate ^c	Trichloromonofluoromethane ^b
Dibenz[a,h]anthracene ^b	beta-1,2,3,4,5,6-Hexachlorocyclohexane ^c
Dibenzofuran ^c	gamma-Chlordane ^c
Radionuclides	
Americium-241 ^b	Plutonium-241 ^b
Carbon-14 ^b	Potassium-40
Cesium-137	Radium-226
Cobalt-60 ^b	Radium-228 ^b
Europium-152 ^b	Strontium-90
Europium-154 ^b	Technetium-99 ^b
Europium-155 ^b	Tritium ^b
Nickel-63 ^b	Uranium-233/234
Plutonium-238 ^b	Uranium-235
Plutonium-239/240	Uranium-238
Other	
Fluoride	TPHs - diesel range ^b
Phosphate ^a	TPHs - kerosene range ^b
TPH ^b	TPHs - motor oil (high boiling) ^b

^aUncertainty analysis (low detect rate comparison) concludes indistinguishable from background.

^bNo background data for comparison.

^cBackground data consists of just three samples, all reported as nondetects.

COPC = contaminant of potential concern

CVP = closeout verification package

RSVP = remaining sites verification package

TPH = total petroleum hydrocarbon

**Table 4-15. 100-B/C Pilot Project Soil COPCs Greater than
Background of Reference Site Concentrations.**

<i>Inorganics</i>	
Antimony ^a	Molybdenum
Arsenic	Nickel
Boron	Phosphorus
Cadmium	Selenium
Calculated Total Uranium	Silver ^a
Chromium	Strontium (elemental)
Copper	Thallium
Hexavalent Chromium	Tin
Lithium	Uranium (inorganic)
Mercury	Zinc
<i>Organics</i>	
Acetone	Chrysene
Anthracene	Dichlorodiphenyltrichloroethane
Aroclor-1254	Dieldrin
Benzo(a)anthracene	Di-n-butylphthalate
Benzo(a)pyrene	Fluoranthene
Benzo(b)fluoranthene	Indeno(1,2,3-cd)pyrene
Benzo(ghi)perylene	Methylenechloride
Benzo(k)fluoranthene	Phenanthrene
beta-1,2,3,4,5,6-Hexachlorocyclohex	Pyrene
Bis(2-ethylhexyl) phthalate	Toluene ^a
Carbazole	
<i>Radionuclides</i>	
Cobalt-60	Plutonium-238
Curium-244	Plutonium-239/240
Europium-152	Radium-228
Europium-154	Strontium-90
Europium-155	Technetium-99
Nickel-63	Uranium-235 ^a

^aUncertainty analysis (low detect rate comparison) concludes indistinguishable from background.
COPC = contaminant of potential concern

**Table 4-16. 100-NR-2 Project Soil COPCs Greater than
Background of Reference Site Concentrations.**

<i>Inorganics</i>	
Antimony	Manganese
Arsenic	Mercury
Barium	Nickel
Cadmium	Selenium
Calculated Total Uranium	Uranium (inorganic)
Chromium	Vanadium
Lead	Zinc
<i>Radionuclides</i>	
Cobalt-60	Plutonium-238
Europium-152	Strontium-90
Europium-154	Zirconium/Niobium-95
Europium-155	

COPC = contaminant of potential concern

Table 4-17. COPCs Identified for RCBRA Pore Water.

<i>Inorganics</i>	
Aluminum ^a	Nickel
Arsenic ^a	Phosphorus ^a
Cadmium ^a	Selenium
Chromium	Thallium ^a
Cobalt	Tin
Copper ^a	Uranium (inorganic)
Hexavalent Chromium	Vanadium ^a
Lead ^a	
<i>Organics</i>	
2-Nitrophenol	Endrin aldehyde
3+4 Methylphenol (cresol, m+p)	Methoxychlor
Di-n-butylphthalate ^a	Phenol
Dimethyl phthalate	
Radionuclides	
Radium-226	Tritium ^a
Strontium-90	Uranium (radionuclide) ^b
Thorium-232	Uranium-235
<i>Others</i>	
Chlorine ^a	Nitrogen, Kjeldahl total ^a
Nitrogen in ammonia ^a	TPH - diesel range

^aUncertainty analysis (low detect rate comparison) concludes indistinguishable from background

^bNo background data for comparison

COPC = contaminant of potential concern

RCBRA= River Corridor Baseline Risk Assessment

TPH = total petroleum hydrocarbons

Table 4-18. COPCs Identified for RCBRA Sediment.

<i>Inorganics</i>	
Beryllium	Tin ^a
Mercury ^a	Titanium
Selenium	Uranium (inorganic)
Silver	
<i>Organics</i>	
Anthracene	Dichlorodiphenyldichloroethane
Aroclor-1254	Dichlorodiphenyldichloroethylene ^a
Aroclor-1260	Dichlorodiphenyltrichloroethane ^a
Benzo(a)anthracene	Di-n-butylphthalate ^a
Benzo(a)pyrene	Endosulfan I ^a
Benzo(b)fluoranthene	Fluoranthene
Benzo(ghi)perylene	gamma-Chlordane
Benzo(k)fluoranthene	Indeno(1,2,3-cd)pyrene
Chrysene	Phenanthrene
Delta-BHC	Pyrene
Dibenz[a,h]anthracene	
<i>Radionuclides</i>	
Carbon-14 ^b	Europium-154
Cesium-137	Strontium-90 ^a
Cobalt-60 ^a	Uranium-235 ^a
Europium-152 ^a	
<i>Others</i>	
Fluoride	Nitrogen in Nitrite
Nitrogen in ammonia	TPHs - gasoline range

^aUncertainty analysis (low detect rate comparison) concludes indistinguishable from background.

^bNo background data for comparison.

COPC = contaminant of potential concern

RCBRA = River Corridor Baseline Risk Assessment

TPH = total petroleum hydrocarbon

Table 4-19. COPCs Identified for 100-B/C Pilot Project Sediment.

<i>Inorganics</i>	
Antimony	Manganese ^a
Arsenic ^a	Mercury
Barium	Nickel ^a
Beryllium	Selenium
Cadmium	Silver
Chromium	Thallium
Copper ^a	Uranium (inorganic)
Lead ^a	Zinc ^a
<i>Radionuclides</i>	
Cesium-137	Technetium-99 ^b
Potassium-40	

^aUncertainty analysis (low detect rate comparison) concludes indistinguishable from background.

^bNo background data for comparison.

COPC = contaminant of potential concern

Table 4-20. COPCs Identified for 100-NR-2 Project Sediment.

<i>Inorganics</i>	
Antimony	Selenium
Mercury	Uranium (inorganic)
<i>Radionuclides</i>	
Cesium-137	Potassium-40 ^a
Cobalt-60	Strontium-90

^aUncertainty analysis (low detect rate comparison) concludes indistinguishable from background.

COPC = contaminant of potential concern

**Table 4-21. Frequency of Detection of Total and Hexavalent Chromium in
Near-Shore Aquatic Media.**

Constituent	Media	Total Number Results	Number Non-Detect Results	Number Detected Results	Frequency of Detection
Chromium	AQUATIC MACROINVERTEBRATE	105	0	105	100.0%
Chromium	AQUATIC VEGETATION	33	0	33	100.0%
Chromium	AQUIFER TUBE	79	2	77	97.5%
Chromium	CLAM	339	8	331	97.6%
Chromium	CRAYFISH	10	0	10	100.0%
Chromium	FISH	664	71	593	89.3%
Chromium	MUSSEL	3	0	3	100.0%
Chromium	PORE WATER	49	38	11	22.4%
Chromium	SEDIMENT	125	5	120	96.0%
Chromium	SEEP	102	5	97	95.1%
Chromium	SURFACE WATER	919	236	683	74.3%
Hexavalent Chromium	AQUIFER TUBE	221	11	210	95.0%
Hexavalent Chromium	PORE WATER	60	32	28	46.7%
Hexavalent Chromium	SEDIMENT	1	1	0	0.0%
Hexavalent Chromium	SEEP	16	0	16	100.0%
Hexavalent Chromium	SURFACE WATER	41	29	12	29.3%

Table 4-22. Frequency of Detection of Strontium-90 in Near-Shore Aquatic Media.

Constituent	Media	Number Results	Number Non-Detect Results	Number Detected Results	Frequency of Detection
Strontium-90	AQUATIC MACROINVERTEBRATE	43	29	14	32.6%
Strontium-90	AQUATIC VEGETATION	50	19	31	62.0%
Strontium-90	AQUIFER TUBE	87	51	36	41.4%
Strontium-90	CLAM	193	31	162	83.9%
Strontium-90	CRAYFISH	8	0	8	100.0%
Strontium-90	FISH	865	356	509	58.8%
Strontium-90	MUSSEL	1	0	1	100.0%
Strontium-90	PORE WATER	49	46	3	6.1%
Strontium-90	SEDIMENT	250	150	100	40.0%
Strontium-90	SEEP	190	74	114	60.0%
Strontium-90	SURFACE WATER	1179	245	934	79.2%

Table 4-23. Frequency of Detection of Total and Isotopic Uranium in Near-Shore Aquatic Media. (3 Pages)

Constituent	Media	Number Results	Number Non-Detect Results	Number Detected Results	Frequency of Detection
Calculated Total Uranium	AQUATIC MACROINVERTEBRATE	14	12	2	14.3%
Calculated Total Uranium	AQUATIC VEGETATION	20	0	20	100.0%
Calculated Total Uranium	AQUIFER TUBE	10	0	10	100.0%
Calculated Total Uranium	CLAM	18	12	6	33.3%
Calculated Total Uranium	FISH	101	73	28	27.7%
Calculated Total Uranium	PORE WATER	49	19	30	61.2%
Calculated Total Uranium	SEDIMENT	171	1	170	99.4%

**Table 4-23. Frequency of Detection of Total and Isotopic Uranium in
Near-Shore Aquatic Media. (3 Pages)**

Constituent	Media	Number Results	Number Non-Detect Results	Number Detected Results	Frequency of Detection
Calculated Total Uranium	SEEP	86	0	86	100.0%
Calculated Total Uranium	SURFACE WATER	811	16	795	98.0%
Uranium-233/234	AQUATIC MACROINVERTEBRATE	14	12	2	14.3%
Uranium-233/234	AQUATIC VEGETATION	20	0	20	100.0%
Uranium-233/234	AQUIFER TUBE	20	0	20	100.0%
Uranium-233/234	CLAM	20	12	8	40.0%
Uranium-233/234	FISH	101	73	28	27.7%
Uranium-233/234	PORE WATER	49	14	35	71.4%
Uranium-233/234	SEDIMENT	197	1	196	99.5%
Uranium-233/234	SEEP	174	0	174	100.0%
Uranium-233/234	SURFACE WATER	1197	13	1184	98.9%
Uranium-235	AQUATIC MACROINVERTEBRATE	14	12	2	14.3%
Uranium-235	AQUATIC VEGETATION	20	0	20	100.0%
Uranium-235	AQUIFER TUBE	23	3	20	87.0%
Uranium-235	CLAM	18	12	6	33.3%
Uranium-235	FISH	118	108	10	8.5%
Uranium-235	PORE WATER	49	44	5	10.2%
Uranium-235	SEDIMENT	189	88	101	53.4%
Uranium-235	SEEP	172	18	154	89.5%
Uranium-235	SURFACE WATER	1197	954	243	20.3%
Uranium-238	AQUATIC MACROINVERTEBRATE	14	12	2	14.3%
Uranium-238	AQUATIC VEGETATION	20	0	20	100.0%

**Table 4-23. Frequency of Detection of Total and Isotopic Uranium in
Near-Shore Aquatic Media. (3 Pages)**

Constituent	Media	Number Results	Number Non-Detect Results	Number Detected Results	Frequency of Detection
Uranium-238	AQUIFER TUBE	23	3	20	87.0%
Uranium-238	CLAM	20	12	8	40.0%
Uranium-238	FISH	104	76	28	26.9%
Uranium-238	PORE WATER	49	19	30	61.2%
Uranium-238	SEDIMENT	202	5	197	97.5%
Uranium-238	SEEP	174	0	174	100.0%
Uranium-238	SURFACE WATER	1198	18	1180	98.5%

**Table 4-24. Summary of Statistically Significant Regressions and Positive Relationships
for Soil to Biotic Tissue Concentrations.**

Analyte	Media	intercept	slope	r.squared	slope.p	n
Arsenic	TERRESTRIAL INVERTEBRATE	0.697	0.105	0.294	0.0009	34
Cadmium	MAMMAL	0.100	0.0621	0.265	0.006	27
Chromium	MAMMAL	0.363	0.0014	0.120	0.020	45
Copper	TERRESTRIAL INVERTEBRATE	2.59	0.318	0.211	0.0020	43
Mercury	TERRESTRIAL INVERTEBRATE	-0.322	6.26	>0.999	0.014	3
Tin	MAMMAL	1.09	0.0093	0.301	0.023	17
Zinc	TERRESTRIAL INVERTEBRATE	35.9	0.0788	0.181	0.0044	43

Note that 7 of 333 are <0.05, which is less than a 2% rate or less than the 5% significance level used.

**Table 4-25. Summary of Statistically Significant Regressions and Positive Relationships
for Sediment to Biotic Tissue Concentrations.**

Analyte	Media	intercept	slope	r.squared	slope.p	n
Potassium	AQUATIC MACROINVERTEBRATE	1329	0.440	0.184	0.029	26
Tin	AQUATIC MACROINVERTEBRATE	0.246	0.915	0.959	0.021	4

Note that 2 of 142 are <0.05, which is about a 1% rate or less than the 5% significance level used.

Table 4-26. Summary of Statistically Significant Regressions and Positive Relationships for Pore Water to Biotic Tissue Concentrations.

Analyte	Media	intercept	slope	r.squared	slope.p	n
Iron	CLAM	89.2	0.0116	0.328	0.041	13
Potassium	AQUATIC MACROINVERTEBRATE	1344	0.270	0.294	0.0062	24

Note that 2 of 90 are <0.05, which is about a 2% rate or less than the 5% significance level used.

Table 4-27. Analytes for Which Representative Concentrations were Calculated Using Half the Practical Quantification Limit Regardless of Whether Any Detects were Recorded.

Analyte	Comment
Americium-241	
Cesium-137	
Cobalt-60	
Europium-152	
Europium-154	
Europium-155	
Uranium-235	
Uranium-238	
Uranium-233/234	
Strontium-90	
Carbon-14	focused sampling for 100-300 rcbra
Tritium	
Iodine-129	no 100-300 rcbra data, measured in gw
Nickel-63	no 100-300 rcbra data, measured in cvp
Technetium-99	no 100-300 rcbra data, measured in cvp
Barium-133	Not a reactor fission product, only 49 total 100-300 RCBRA sample results
Plutonium-238	
Plutonium-239/240	
Aroclor-1016	no detects for 613 samples
Aroclor-1221	no detects for 613 samples
Aroclor-1232	no detects for 613 samples
Aroclor-1242	no detects for 613 samples
Aroclor-1248	1 detect for 613 samples
Aroclor-1254	29 detects for 613 samples
Aroclor-1260	27 detects for 613 samples

**Table 4-27. Analytes for Which Representative Concentrations were Calculated
Using Half the Practical Quantification Limit Regardless of Whether
Any Detects were Recorded.**

Analyte	Comment
Aroclor-1262	5 detects for 5 samples
Aroclor-1268	no detects for 5 samples

Table 4-28. The Number of Representative Concentrations for Each Data Source and Statistical Method. (4 Pages)

Data Source	Media	Value	All Detects [N<3]	All Detects [N=3,4]	All Detects [N>=5]	All Non-detects [N<5]	All Non-detects [N>=5]	Non-detects < 15%	Non-detects > 15%	Some detects [N<5]	
RCBRA	BIOTA	All Non-detects	0	0	0	8317	542	0	0	0	
		Bootstrap	0	0	130	0	0	4	30	0	
		Lognormal: Lands H	0	341	187	0	0	19	28	182	
		Max	4737	129	2	0	0	0	239	329	
		Normal: Bounded Students t	0	0	0	0	0	0	4	0	
		Normal: Students t	0	0	32	0	0	5	0	0	
	SEDIMENT	All Non-detects	0	0	0	3277	0	0	0	0	0
		Bootstrap	0	0	0	0	0	0	0	0	0
		Lognormal: Lands H	0	0	0	0	0	0	0	0	0
		Max	1773	0	0	0	0	0	0	0	2
		Normal: Bounded Students t	0	0	0	0	0	0	0	0	0
		Normal: Students t	0	0	0	0	0	0	0	0	0
	SOIL	All Non-detects	0	0	0	86	3916	0	0	0	0
		Bootstrap	0	0	1206	0	0	14	99	0	
		Lognormal: Lands H	0	56	858	0	0	2	3	16	
		Max	0	10	361	0	0	0	632	13	
		Normal: Bounded Students t	0	0	0	0	0	0	0	0	
		Normal: Students t	0	0	73	0	0	3	0	0	
	WATER	All Non-detects	0	0	0	14457	197	0	0	0	0
		Bootstrap	0	0	42	0	0	2	8	0	
		Lognormal: Lands H	0	41	22	0	0	3	2	8	
		Max	5588	26	2	0	0	0	17	85	
		Normal: Bounded Students t	0	0	0	0	0	0	0	0	
		Normal: Students t	0	0	1	0	0	0	0	0	
CVP/RSVP	BIOTA	All Non-detects	0	0	0	0	0	0	0	0	
		Bootstrap	0	0	0	0	0	0	0	0	
		Lognormal: Lands H	0	0	0	0	0	0	0	0	
		Max	0	0	0	0	0	0	0	0	
		Normal: Bounded Students t	0	0	0	0	0	0	0	0	
		Normal: Students t	0	0	0	0	0	0	0	0	

Table 4-28. The Number of Representative Concentrations for Each Data Source and Statistical Method. (4 Pages)

Data Source	Media	Value	All Detects [N<3]	All Detects [N=3,4]	All Detects [N>=5]	All Non-detects [N<5]	All Non-detects [N>=5]	Non-detects < 15%	Non-detects > 15%	Some detects [N<5]	
	SEDIMENT	All Non-detects	0	0	0	0	0	0	0	0	
		Bootstrap	0	0	0	0	0	0	0	0	0
		Lognormal: Lands H	0	0	0	0	0	0	0	0	0
		Max	0	0	0	0	0	0	0	0	0
		Normal: Bounded Students t	0	0	0	0	0	0	0	0	0
		Normal: Students t	0	0	0	0	0	0	0	0	0
	SOIL	All Non-detects	0	0	0	2534	1242	0	0	0	0
		Bootstrap	0	0	679	0	0	9	45	0	0
		Lognormal: Lands H	0	943	872	0	0	4	18	53	0
		Max	676	287	54	0	0	0	188	101	0
		Normal: Bounded Students t	0	0	0	0	0	0	2	0	0
		Normal: Students t	0	0	97	0	0	4	0	0	0
	WATER	All Non-detects	0	0	0	0	0	0	0	0	0
		Bootstrap	0	0	0	0	0	0	0	0	0
		Lognormal: Lands H	0	0	0	0	0	0	0	0	0
		Max	0	0	0	0	0	0	0	0	0
		Normal: Bounded Students t	0	0	0	0	0	0	0	0	0
		Normal: Students t	0	0	0	0	0	0	0	0	0
100-NR-2	BIOTA	All Non-detects	0	0	0	9	1	0	0	0	0
		Bootstrap	0	0	16	0	0	1	3	0	0
		Lognormal: Lands H	0	1	40	0	0	0	0	0	0
		Max	22	1	1	0	0	0	3	0	0
		Normal: Bounded Students t	0	0	0	0	0	0	1	0	0
		Normal: Students t	0	0	0	0	0	0	0	0	0
	SEDIMENT	All Non-detects	0	0	0	3	0	0	0	0	0
		Bootstrap	0	0	0	0	0	0	0	0	0
		Lognormal: Lands H	0	12	0	0	0	0	0	0	1
		Max	5	0	0	0	0	0	0	0	1
		Normal: Bounded Students t	0	0	0	0	0	0	0	0	0
		Normal: Students t	0	0	1	0	0	0	0	0	0

Table 4-28. The Number of Representative Concentrations for Each Data Source and Statistical Method. (4 Pages)

Data Source	Media	Value	All Detects [N<3]	All Detects [N=3,4]	All Detects [N>=5]	All Non-detects [N<5]	All Non-detects [N>=5]	Non-detects < 15%	Non-detects > 15%	Some detects [N<5]
	SOIL	All Non-detects	0	0	0	5	0	0	0	0
		Bootstrap	0	0	8	0	0	0	0	0
		Lognormal: Lands H	0	2	6	0	0	0	0	0
		Max	8	5	0	0	0	0	0	2
		Normal: Bounded Students t	0	0	0	0	0	0	0	0
		Normal: Students t	0	0	1	0	0	0	0	0
	WATER	All Non-detects	0	0	0	3	2	0	0	0
		Bootstrap	0	0	3	0	0	0	0	0
		Lognormal: Lands H	0	12	5	0	0	0	0	1
		Max	2	2	2	0	0	0	3	1
		Normal: Bounded Students t	0	0	0	0	0	0	0	0
		Normal: Students t	0	0	0	0	0	0	0	0
100-B/C PILOT	BIOTA	All Non-detects	0	0	0	33	34	0	0	0
		Bootstrap	0	0	32	0	0	1	6	0
		Lognormal: Lands H	0	21	216	0	0	7	20	1
		Max	64	2	17	0	0	0	31	2
		Normal: Bounded Students t	0	0	0	0	0	0	3	0
		Normal: Students t	0	0	13	0	0	2	0	0
	SEDIMENT	All Non-detects	0	0	0	5	0	0	0	0
		Bootstrap	0	0	0	0	0	0	0	0
		Lognormal: Lands H	0	6	0	0	0	0	0	0
		Max	33	4	0	0	0	0	0	0
		Normal: Bounded Students t	0	0	0	0	0	0	0	0
		Normal: Students t	0	0	0	0	0	0	0	0
	SOIL	All Non-detects	0	0	0	139	15	0	0	0
		Bootstrap	0	0	8	0	0	0	3	0
		Lognormal: Lands H	0	0	39	0	0	1	0	2
		Max	65	0	7	0	0	0	4	6
		Normal: Bounded Students t	0	0	0	0	0	0	0	0
		Normal: Students t	0	0	2	0	0	0	0	0

Table 4-28. The Number of Representative Concentrations for Each Data Source and Statistical Method. (4 Pages)

Data Source	Media	Value	All Detects [N<3]	All Detects [N=3,4]	All Detects [N>=5]	All Non-detects [N<5]	All Non-detects [N>=5]	Non-detects < 15%	Non-detects > 15%	Some detects [N<5]
	WATER	All Non-detects	0	0	0	29	45	0	0	0
		Bootstrap	0	0	23	0	0	4	14	0
		Lognormal: Lands H	0	0	13	0	0	1	5	0
		Max	9	5	0	0	0	0	12	0
		Normal: Bounded Students t	0	0	0	0	0	0	1	0
		Normal: Students t	0	0	2	0	0	0	0	0

5.0 HUMAN HEALTH RISK ASSESSMENT

5.1 INTRODUCTION

The human health risk assessment section of the 100 Area and 300 Area Component of the RCBRA provides the methods and results that quantify human exposure to contaminants in environmental media and associated health effects. A baseline human health risk assessment is generally structured in four steps (EPA 1989):

1. Data Collection and Evaluation
2. Exposure Assessment
3. Toxicity Assessment
4. Risk Characterization.

The human health risk assessment within this document focuses on Steps 2, 3, and 4 of the risk assessment process, since Step 1 is in common with the ecological risk assessment and is presented in Section 4.0 of this document. A brief summary of the CEM, derived from the CSM in Section 2.0, is provided in Section 5.2 of this risk assessment. The principal aspects of the exposure assessment (Step 2) are the calculation of exposure point concentrations in each exposure medium and the calculation of chemical intake. These are described in Sections 5.3 and 5.4 of this risk assessment, respectively. The toxicity assessment (Step 3), described in Section 5.5, discusses the toxicity models used to characterize potential adverse human health effects related to contaminant exposure. The sources of toxicity criteria, which represent the chemical-specific results of the toxicity models employed by the EPA, are also discussed. The risk characterization (Step 4) is described in Section 5.6. The risk characterization discusses how intake estimates and toxicity criteria are used to assess four measures of potential health effects: chemical hazard, chemical cancer risk, radionuclide cancer risk, and radionuclide dose. The results of the human health risk assessment, including uncertainty analyses, are presented in Sections 5.7 and 5.8. A summary of the 100 Area River Effluent Pipelines Assessment is provided in Section 5.9. Section 5.10 presents the conclusions of the human health risk assessment.

Electronic files of the representative concentrations used to calculate exposure point concentrations in the various exposure media, the exposure point concentrations, and the human health risk assessment results are provided in electronic format in Appendices F-6 and G-1. Additionally, the computer code used to perform the risk calculations using the representative concentrations input file and a primer on the use of the code and interpretation of the output text files are contained in Appendices F-6 and G-1. A Microsoft[®] Excel workbook that can be used to calculate risk assessment results for any combination of exposure scenario and COPCs is also included in Appendix G-1. The full text, as well as references, tables, and figures, of the 100 Area River Effluent Pipelines Assessment is provided in Appendix G-2.

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5.2 EXPOSURE SCENARIOS AND RECEPTORS

5.2.1 Summary of the Human Health Conceptual Exposure Model

As described in Section 2.4, the nature of potential exposure scenarios in the 100 Area and 300 Area Component of the RCBRA has been the subject of numerous workshops and other discussions among various parties. One outcome of the early discussions was a decision to implement a pilot human health and ecological risk assessment for the 100-B/C Area. The draft pilot assessment is documented in *100-B/C Pilot Project Risk Assessment Report* (DOE/RL-2005-40). The 100-B/C Pilot Project risk assessment employed the following exposure scenarios in the human health risk assessment:

- Hypothetical Native American User Scenario (CTUIR)
- Hypothetical Rural-Residential Scenario
- Hypothetical Resident Monument Worker Scenario
- Hypothetical Industrial/Commercial Worker Scenario
- Hypothetical Recreational Use Scenarios: Avid Hunter, Avid Angler, and Casual User applications.

Among these exposure scenarios, contaminant exposure and potential health effects were quantified in the 100-B/C Pilot Project Risk Assessment for all except the Native American User scenario developed by the CTUIR. Local and regional Tribes having ancestral ties to the Hanford Reach of the Columbia River and surrounding lands were requested by DOE to provide an exposure scenario(s) reflecting their traditional activities (DOE/RL-2005-40). At this time, only the CTUIR have submitted an exposure scenario report to DOE (Harris and Harper 2004). Exposure via this scenario is evaluated in this report.¹

It is important to recognize the hypothetical nature of these exposure scenarios relative to future conditions in the 100 Area and 300 Area. The order of the exposure scenarios in the bullets above reflects a range of high-intensity to low-intensity exposure conditions. The purpose of assessing potential risks under such a range of conditions is to provide risk managers with information on how potential risks may vary as a function of exposure intensity under a variety of exposure assumptions. The use of these scenarios in this risk assessment does not imply any endorsement of either the scenarios or the underlying assumptions by DOE or other stakeholders with respect to future land use. In particular, risks related to the Rural-Residential and CTUIR scenarios are not representative of potential future exposures when DOE maintains its anticipated land use and institutional controls.

Potentially complete exposure pathways, and associated exposure media, for these exposure scenarios are summarized in Table 5-1. The bases of these scenarios are discussed in detail in the CSM. Estimation of contaminant concentrations in these media for use in the human health

¹ Additional hypothetical Native American scenarios may be provided in the future by the Yakama tribe, the Wanapum, or other groups.

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risk assessment, and calculation of chemical intake for the receptors and scenarios shown in Table 5-1, is the subject of subsequent sections of this exposure methodology.

5.2.2 Implementation of the Conceptual Site Model

This RCBRA encompasses the six reactor areas that make up the 100 Area, the 300 Area, and various waste sites within the 100-IU-2 and 100-IU-6 OUs. The Hanford Site covers a total of 1,450 km² (560 mi²), of which the 100 Area comprises about 52 km² (20 mi²) and the 300 Area (with surrounding areas used for solid and liquid waste disposal) comprises about 2.6 km² (1 mi²). Across the various exposure scenarios and pathways, chronic exposures may be highly localized (e.g., ingestion of produce grown in a home garden) or occur over a very broad area (e.g., ingestion of meat from cattle or game that have grazed over many square miles). Therefore, delineation of appropriate locations and spatial scales for application of the exposure scenarios is critical.

Conceptually, the environmental data described in Section 4.0 can be grouped into data that pertain to individual waste sites and data that characterize contaminant concentrations in environmental media over larger areas and across multiple potential sources. Individual waste site data primarily include the CVP/RSVP data. Riparian and near-shore biota, soil, and sediment data; surface water data; and environmental surveillance tissue data are examples of data that characterize contaminant concentrations in the environment over larger areas. The MIS soil data and tissue data collected under the SAP (DOE/RL-2005-42) at specific waste sites occupies a middle ground between these data types. Although they are collected at specific sites they are intended to represent the range of potential concentrations across all remediated sites. Groundwater well data may in some cases reflect contamination from a particular liquid waste disposal site, and other times reflect contamination from multiple 100 Area, 300 Area, and/or 200 Area sources.

The human health risk assessment calculations will be conducted, depending on exposure scenario, on either a "local area" scale, a "broad area" scale, or using combinations of both local and broad area scales for specific exposure pathways. The "local area" scale relates to contaminant concentrations associated with an individual waste site, while the "broad area" scale relates to contaminant concentrations across an individual reactor area or the 300 Area.

A summary of soil-related exposure pathways, organized according to spatial scale, is provided in Table 5-2. The presence of foodstuff pathways in Table 5-2 reflects the fact that (with the exception of aquatic foodstuffs) soil is the primary environmental medium harboring contaminants that may migrate to these foodstuffs.

For the Resident Monument Worker, soil-related exposures are fractionated between the Local Area and the Broad Area as a function of the length of time spent at a residence and working, respectively. For the Rural-Residential and CTUIR scenarios, exposure and associated health effects are evaluated for both a purely local exposure scenario (Local Only) or with a combination of Local Area and Broad Area exposures. In the latter case, the Broad Area soil-related pathways shown for these scenarios are substituted for the Local Area equivalents.

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The CTUIR scenario implemented in this assessment employs primarily Local Area soil exposure (around the residence), with Broad Area soil exposure assessed for biotically mediated soil exposure pathways. Harris and Harper (2004) describe the application of the subsistence lifestyle scenario across five separate age groups with numerous activity categories and associated pathways. This assessment incorporates the exposure pathways and contact rate parameter values described in Harris and Harper (2004). However, rather than attempting to apportion time across numerous potential activities and locations for different individuals, this assessment considers an individual who spends essentially all of their time in and around their residence. This assumption results in a maximally exposed individual with respect to residual contamination associated with an individual waste site.

The purpose of calculating risks related to groundwater is primarily to provide an approximate measure of the relative significance of soil and groundwater as exposure media in the 100 Area and 300 Area. It is desirable in principle to sum risks related to groundwater exposure with risks for other media in order to understand all potential risks under hypothetical residential conditions and to assess the relative importance of exposures via soil and groundwater. However, it is also appropriate to recognize that the purpose of this human health assessment is primarily to evaluate the adequacy of soil remediation efforts at individual waste sites. Protection of groundwater from residual soil contamination was addressed in the development of existing waste site soil remediation criteria for the 100 Area and 300 Area, and groundwater is being addressed via a program instituted in parallel with waste site remediation. It is also significant that residual groundwater contamination in the investigation areas has been impacted by releases outside of these areas, and that groundwater contaminant concentrations are dynamic and have not necessarily peaked for all combinations of contaminants and locations in the 100 Area and 300 Area. Pathway-specific health risks related to groundwater contamination for the Rural-Residential, Monument Worker, and CTUIR exposure scenarios (see Table 5-1) will be calculated for each monitoring well in the 100 Area and 300 Area. The range of potential groundwater-related risks across the monitoring wells in the 100 Area and 300 Area will be described for each of these exposure scenarios in Section 5.8.

The point in time when the hypothetical future exposure scenarios are applied may be an important consideration. As described in Section 2.1, all but two of the 100 Area reactors (i.e., 105-B and 105-N) are being placed in ISS for up to 75 years while cobalt-60 (5.3-year half-life) and cesium-137 (30.1-year half-life) decay to lower activity levels. Because no definite date has been identified as a time when any particular scenario may be realized, radiological sample data have been protectively used as measured and have not been decayed to any specific future time.

5.3 EXPOSURE ASSESSMENT: CALCULATION OF EXPOSURE POINT CONCENTRATIONS

5.3.1 Overview

An exposure point concentration is the concentration of a contaminant in an exposure medium at the time and location where a receptor may contact that medium. Exposure point concentrations

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are calculated based on representative concentrations in sampled media, as described in Section 4.0. The distinction between exposure point concentrations and representative concentrations is that representative concentrations are limited to individual sample media, whereas exposure point concentrations also encompass modeled concentrations in other exposure media. A discussion of the estimation of exposure point concentrations for each combination of exposure medium and scenario described in Table 5-1 is provided below. Text files of the representative concentrations used to calculate exposure point concentrations in the various exposure media, and the exposure point concentrations, are provided in electronic format in Appendices F-5 and G-1.

The primary data sets used for calculating representative concentrations include the following:

1. Post-remediation soil data of the waste sites addressed in this risk assessment. These data are applied to risk calculations on the scale of an individual waste site.
2. MIS soil data collected in upland and riparian environments in accord with the SAP (DOE/RL-2005-42). The locations of sample collection are depicted in figures provided in Appendix B of this report.
3. Columbia River sediment data and sculpin tissue data collected in the near-shore environment in accord with the SAP (DOE/RL-2005-42). The locations of sample collection are depicted in figures provided in Appendix B of this report.

Additional data sources, primarily employed in the uncertainty analysis to qualify exposure point concentrations calculated using the primary data sets identified above, are discussed in Sections 5.3.2 through 5.3.15 where applicable.

Exposure point concentrations for both the mean and the 95% UCL are discussed in the following subsections. The mean is used to calculate potential risks using central tendency estimates of possible exposure. The UCL is used when calculating risk related to RME assumptions.

5.3.2 Upland Surface and Subsurface Soil

There are five types of environmental data that are applicable to the calculation of representative concentrations for upland areas. Evaluation of these data for this purpose is described in Section 4.0.

CVP and RSVP data: CVP/RSVP are post-remediation soil samples. Representative concentrations for the mean and the UCL of each waste site, based on the 0- to 4.6-m (0- to 15-ft) sample interval, have been calculated according to methods described in Section 4.0. Where CVP data are available for the >4.6-m (15-ft) sample interval, representative concentrations for the mean and UCL of each waste site in this interval will be calculated in like manner as the 0- to 4.6-m (0- to 15-ft) interval. For any specific site, there may exist shallow-zone data (0 to 4.6 m [0 to 15 ft]), deep-zone data (>4.6 m [15 ft]), both, or either one or the other.

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MIS Data from the Operational Area: MIS soil data have been collected at waste sites identified in the SAP (DOE/RL-2005-42). Representative concentrations for the mean and UCL of operational area soils have been calculated according to methods described in Section 4.0.

MIS Data from the Reference Area: Reference site data are employed to calculate background risks related to the soils characterized by the MIS investigation areas. Representative concentrations for the mean and UCL of reference area soils have been calculated according to methods described in Section 4.0.

100-B/C Pilot Project Data: Selected waste disposal sites at the 100-B/C Area were evaluated as part of the 100-B/C Pilot Project risk assessment (DOE/RL-2005-40). As stated in the SAP (DOE/RL-2005-42), data from the 100-B/C Pilot Project sampling is included within the risk assessment for the 100 Area and 300 Area Component of the RCBRA. These upland soil data have been integrated with the MIS soil data in the calculation of representative concentrations for the mean and UCL.

Background Data: Hanford Site background data are used to represent concentrations of metals and radionuclides in waste site soils, as discussed in the following subsection. Mean values are available in the Hanford Site background reports described in Section 4.0. A 90th percentile of the lognormal distribution is used to represent an upper-bound value for Hanford Site soil background. Background comparisons to identify analytes present at concentrations exceeding those measured in background samples are performed according to methods described in Section 4.0.

5.3.2.1 Upland Exposure Point Concentrations for the Rural-Residential, CTUIR, Resident Monument Worker, and Industrial/Commercial Worker Scenarios. As discussed in Section 5.2.2, exposure point concentrations for Upland areas are calculated on a scenario-specific and pathway-specific basis for exposures that are envisioned to occur within both relatively small (“local area”) and relatively large (“broad area”) chronic exposure areas.

“Local Area” Upland Soil Exposure Point Concentrations

“Local area” exposure point concentrations are associated with the CTUIR, Rural-Residential, Industrial/Commercial, and the residential portion of the Resident Monument Worker exposure scenarios (see Table 5-2). Sites known to have significant levels of contamination and therefore targeted for remediation have traditionally been referred to as CVP sites. Waste sites where levels of residual contamination were uncertain, and which have often had minimal or no subsequent removal actions, have often been referred to as RSVP sites (i.e., “remaining sites” or “native soil sites”).

At excavated (CVP) sites, soil data reflect residual contamination in soils adjacent to, or below, a volume of excavated soil that has been replaced with clean backfill. The residual contamination on the sidewalls of such an excavation has been colloquially referred to as a “bathtub ring,” reflecting an assumption that the verification data collected from these sidewalls represents a relatively thin layer of affected soils enclosing the clean backfill. Residual contamination is also generally expected in the vadose zone beneath a site, particularly if the site was associated with

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disposal of liquid wastes. In principle, residual contamination levels in the CVP and RSVP verification samples should be below interim cleanup levels that were applied to shallow zone (< 4.6 m [15 ft] bgs) and deep zone (> 4.6 m [15 ft]) soils.

Because CVP data from excavated sites are related to subsurface soils with the configuration described above, it is necessary to hypothesize some process by which these soils may serve as a chronic human exposure medium. For this purpose, a basement excavation model is employed to generate a reasonable worst-case hypothetical exposure term for the Rural-Residential, CTUIR, Resident Monument Worker, and Industrial/Commercial Worker exposure scenarios. The output of the basement excavation model is a hypothetical surface soil exposure area that contains contamination that is representative of both the soil data characterized by the CVP verification data as well as the backfill that exists at the site.

The basic attributes of the basement excavation model include the following:

- An assumed 4.6-m (15-ft) basement excavation depth (corresponds to the thickness of the shallow zone data from which shallow zone CVP samples were collected)
- A basement area of approximately 50 m² (based on assumed dimensions of 5 m by 10 m [16 ft by 33 ft], where the 5-m [16-ft] width accommodates the intersection of an excavation side wall with the 1.5:1 slope that was commonly employed)
- Incorporation of drill cuttings from a water supply well (the volume of cuttings relate to a well with an assumed diameter of 15 cm and a depth equal to the thickness of the vadose zone below the remedial excavation at the waste site)².

An excavation with dimensions of 4.6 m by 5 m by 10 m (15 ft by 16 ft by 33 ft) yields an excavated soil volume of approximately 230 m³. This material is assumed to be distributed onto the ground surface, where it covers an area of approximately 1,500 m² to a depth of approximately 15 cm. An area of this size is associated with an effectively infinite source area for several exposure pathways in the RESRAD computer code developed to support radioactive site assessment under DOE Order 5400.5. These pathways include external irradiation, inadvertent soil ingestion, and growing of garden produce (ANL-EAD-4). A plan view of the basement excavation model showing the excavation oriented along a sidewall where residual contamination may be encountered and a hypothetical 1,500 m² exposure area is shown in Figure 5-1.

The presumed location of the basement with respect to the sidewalls of an excavated site is an important characteristic of the basement excavation model. Because backfill was used to replace excavated soils, the location of a basement in the middle of an excavated site will result in no contact with residual contaminants if the excavation depth is 4.6 m (15 ft) or greater. However, if the basement were oriented with the long axis parallel to an excavated sidewall, approximately 50% of the excavation would consist of soils below the slope of the sidewall. If it is protectively

² If information on the thickness of the vadose zone below the remedial excavation was unavailable for a particular waste site, a thickness was estimated based on information from other sites within the operational area.

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assumed that all soil to a depth of 4.6 m (15 ft) beneath the sidewall is characterized by the shallow-zone verification samples, then the basement excavation model yields approximately a 1:1 mixing of "contaminated" soil with backfill when the excavation depth is 4.6 m (15 ft). A graphical depiction of this scenario, including the presence of a presumed drinking water well through a vadose zone characterized by deep-zone verification samples, is shown in Figure 5-2. As shown in Figure 5-2, levels of soil contamination will decrease with lateral distance from the sidewalls.

As the excavation depth decreases from 4.6 m (15 ft), the proportion of excavated materials made up of soils from below the sidewall progressively decreases. At an excavation depth of approximately 3.2 m (10.5 ft) or less, it is more protective to assume that the basement is situated in the middle of the excavation, where the material below 3.2 m (10.5 ft) is characterized by the CVP samples. At a 3.2-m (10.5-ft) waste site excavation depth, approximately 33% of the basement materials are composed of potentially contaminated soil. Exposure point concentrations are calculated based on the proportion of mixing of shallow-zone and backfill soils; for unexcavated sites the proportion of backfill is zero. Deep-zone residual contamination is incorporated via the well cuttings term. Because the volume of deep-zone soils is negligible relative to the remaining terms, the fractional quantity of this soil is not accounted for in the calculation of the local area exposure point concentration. The calculation is performed according to:

$$(F_{sz} \times C_{sz}) + (F_{bf} \times C_{bf}) + [(V_w / V_b) \times C_{dz}]$$

where F_{sz} = fraction of basement excavation represented by potentially contaminated shallow zone soil (unitless)

C_{sz} = contaminant concentration in shallow zone soil (mg/kg or pCi/g)

F_{bf} = fraction of basement excavation represented by backfill soil (unitless)

C_{bf} = contaminant concentration in backfill soil (mg/kg or pCi/g)³

V_w = Volume of potentially contaminated well cuttings (m³)

V_b = Volume of basement excavation (m³)

C_{dz} = contaminant concentration in deep zone soil (mg/kg or pCi/g).

When the site excavation depth is 3.2 m (10.5 ft) or less, the fraction of excavated material represented by potentially contaminated shallow zone soil (F_{sz}) is calculated as,

$$(4.6 \text{ m} - \text{excavation depth}) / 4.6 \text{ m}$$

At excavation depths greater than 3.2 m, F_{sz} is calculated as,

$$0.33 + [0.17 \times (\text{ABS}(3.2 \text{ m} - \text{excavation depth}) / (4.6 \text{ m} - 3.2 \text{ m}))]$$

where ABS indicates the absolute value of (3.2 m – excavation depth), and with a maximum value of 0.5 for F_{sz} .

³ Represented by the MIS upland sites soil data; see explanation below.

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The exposure point concentrations calculated using the basement excavation model are assigned to an approximately 1,500 m² hypothetical source term with a depth of potentially contaminated soil equal to 15 cm. The results of these assessments for excavated waste sites remediated to date may be extrapolated to still-unremediated sites to draw conclusions regarding the adequacy of cleanup levels identified in the interim action RODs for protecting human health.

Two issues must be addressed to allow computation of the local area exposure point concentrations for this hypothetical source term. The first is how to represent constituent concentrations in backfill. In theory, these might be represented using either background data or the MIS soil data collected to characterize upland soils. The MIS soil data was used for this purpose because the origin of the backfill that was used during site remediation is often uncertain. However, it is possible that constituent concentrations in backfill may be overestimated by the use of these data. The upland MIS soil data represent concentrations in the 0- to 15-cm (0- to 6-in.) surface layer, whereas the data are used to here to represent backfill that is commonly much thicker than 15 cm (6 in.).

The second issue is how to represent the concentrations of constituents that were not analyzed in the CVP samples. The analyte list for these samples was focused to those chemicals and radionuclides related to historical operations, whereas the MIS soil data include a broad suite of analytes. Hanford Site background data have been selected to represent the concentrations of metals and radionuclides that were not analyzed for in shallow- and deep-zone verification samples. In lieu of a UCL value, which is not defined in the Hanford Site background reports, a 90th percentile of the lognormal distribution (which is tabulated in these reports) is used.

Lastly, the question of how to represent background exposure point concentrations for the calculation of background risks must be considered. Local area exposure point concentrations for the hypothetical source term are calculated using values from four data sets: CVP and RSVP shallow-zone data, CVP deep-zone data, MIS upland soil data, and Hanford Site background data. Because the waste site soil CVP data are only one input for computing these exposure point concentrations, it is not feasible to compute background risks for individual waste sites based on the specific analytes detected in the verification data. Background risks are calculated using both the broad area upland soil data and the reference area upland soil data. Background risk calculations related to the former may be conceptually understood as pertaining to a “no excavation” model, where a residence or commercial structure is located at some distance from a waste site or otherwise does not intrude into subsurface soils represented by the CVP soil data. The background risk calculations using the reference area upland soil data reflect background risks across the soil constituents comprising the MIS analytical suite in areas unaffected by Hanford Site operations.

“Broad Area” Upland Soil Exposure Point Concentrations

“Broad area” exposure point concentrations in the upland environment are associated with the residential portion of the Resident Monument Worker and certain exposure pathways in the CTUIR and Rural-Residential exposure scenarios (see Table 5-2). The MIS and 100-B/C Pilot Project upland soil data are used to calculate “broad area” exposure point concentrations, which

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describe contaminant concentrations that are averaged over the scale of an entire operational area or larger. These data are considered to be representative of present-day surface soil contamination in potentially affected regions of an operational area.

MIS sampling data from 20 upland locations representing 20 operational area sites are used for calculating "broad area" exposure point concentrations in upland soils. In addition, upland operational area soil samples from the 100-B/C Pilot Project (DOE/RL-2005-40) are incorporated in the calculation of "broad area" upland exposure point concentrations. There are one to three samples in this data set for metals and semivolatile organic chemicals, and approximately six samples for radionuclides. Exploratory data analysis has been conducted to determine whether significant differences in contaminant concentrations exist across the 20 operational area MIS samples and the 100-B/C Pilot Project samples. Based on these analyses (see Section 4.0), representative concentrations in upland soils for operational areas have been calculated by pooling the soil data across the 20 MIS sampling locations and the 100-B/C Pilot Project samples. The MIS soil data from the 10 upland reference area sites will be used to calculate representative concentrations associated with background in upland areas. Representative concentrations for the mean and UCL have been calculated according to methods described in Section 4.0.

5.3.2.2 Upland Exposure Point Concentrations for the Recreational Scenario. The upland exposure point concentrations for the Avid Hunter variation of the Recreational scenario are calculated in a manner identical to the "broad area" exposure point concentrations for the Rural-Residential, Resident Monument Worker, and CTUIR scenarios described above.

5.3.3 Riparian Soil

"Broad area" exposure point concentrations in the riparian environment are associated with the Casual User recreational exposure scenario, and the inhalation exposure route for the Avid Angler exposure scenario (see Table 5-2). MIS sampling data from 10 operational area riparian locations are used for calculating exposure point concentrations in riparian soil. Additionally, the riparian and near-shore areas at the 100-B/C and 100-N Areas were evaluated as part of the 100-B/C Pilot Project (DOE/RL-2005-40) and the 100-NR-2 shoreline investigation (DOE/RL-2005-22). Sampling in these areas was not be duplicated under the SAP (DOE/RL-2005-42). The data from these projects are also included within this risk assessment, as stated in the SAP (DOE/RL-2005-42). There are approximately 16 samples for metals and radionuclides and 3 samples for semivolatile organic chemicals in the 100-B/C Pilot Project riparian soil data set. In the 100-NR-2 riparian soil data set, there are between 13 and 17 samples for metals, strontium-90, and technetium-99, and between 1 and 4 samples for certain other radionuclides.

Exploratory data analysis has been conducted to determine whether riparian soils have different concentrations of COPCs than the upland soils and to determine the appropriate spatial scale for the aggregation of these soil data. Based on these analyses (see Section 4.0), riparian and upland soils are differentiated as unique potential exposure media. Riparian soil data have been binned in two groups: an operational area group and a reference site group. The MIS data from the

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10 riparian operational area sites, along with the 100-B/C and 100-NR-2 data, will be used to calculate representative concentrations associated with operational riparian areas. The MIS soil data from the five riparian reference area sites will be used to calculate representative concentrations associated with background in riparian areas.

Representative concentrations for the mean and UCL have been calculated according to methods described in Section 4.0. The riparian soil data are only employed in the risk assessment as a source term for the Casual User exposure scenario and the dust inhalation exposure pathway in the Avid Angler variation of the Recreational exposure scenario. For this latter use, the analytes evaluated in the risk calculations are those which are also included in the risk calculations for sediments. For other exposure pathways in the Avid Angler exposure scenario (inadvertent soil ingestion, dermal absorption, and external irradiation), direct exposure to sediments is assessed. Inhalation exposure is not a feasible exposure pathway for submerged sediments because these sediments are unavailable as a source for suspended particulates in air.

5.3.4 Near-Shore Sediment

As part of the SAP (DOE/RL-2005-42), sediment data were collected from 37 near-shore aquatic sites (including 7 reference locations) along the Hanford Reach of the Columbia River. These sites were selected based on the locations of known groundwater plumes, the results of a 2005 conductivity survey for identifying areas of groundwater discharge to the river, and the results of past biota sampling locations and results indicating areas of potential contamination. Hence, these sediment data provide a protectively biased estimate of general sediment concentrations along the Hanford Reach. The sampling sites associated with known groundwater plumes were located within the 300 Area (uranium plume), 100-D and 100-K Areas (chromium plumes), and 100-N Area (strontium plume). Sediment data from the 100-B/C Pilot Project (DOE/RL-2005-40) and 100-NR-2 investigation (DOE/RL-2005-22) are also included within this risk assessment, as stated in the SAP (DOE/RL-2005-42). There are either two or three sediment samples for metals and radionuclides in the 100-B/C Pilot Project sediment data set. In the 100-NR-2 sediment data set, there are also either two or three samples for metals and most radionuclides and six samples for strontium-90.

Background comparisons between RCBRA sediment samples collected under the SAP (DOE/RL-2005-42) within the Hanford Reach and reference area locations indicate that, among analytes with adequate detection frequency to conduct statistical tests, only concentrations of ammonia nitrogen, titanium, and beryllium may exceed reference area concentrations. However, when integrating the other data sources described in Section 4.2, it is evident that concentrations of uranium in certain 300 Area sediments and strontium-90 concentrations in samples collected during the 100-NR-2 investigation (DOE/RL-2005-22) are elevated relative to other operational area sediments. As discussed in Section 4.4, elevated concentrations of strontium-90 were correlated in sediment and clam tissues in the 100-N Area. Some correlation of inorganic and isotopic uranium concentrations in sediment and aquatic tissue samples was also observed at the 300 Area.

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On the basis of the observed differences in certain analyte concentrations in the 300 Area and 100-N Area, relative to remaining operational areas in the 100 Area, risk calculations for sediments are conducted independently for these three data groups. Concentrations of certain analytes also appear elevated in sediments collected during the 100-B/C Pilot investigation, though this may be a function of differences in sample preparation and/or collection protocol relative to the RCBRA samples. Because there are already three data groupings based on apparent differences in sediment analyte concentrations, the 100-B/C Pilot sediment data are also evaluated independently rather than being integrated with one of the other data sets. A total of four data groups are evaluated for sediment exposures:

1. 100 Area (inclusive of all 100 Area data)
2. 300 Area
3. 100-N Area (data collected under the 100-NR-2 investigation)
4. 100-B/C Area (data collected under the 100-B/C Pilot investigation).

As indicated in Figure 5-1, the sediment data are employed in the risk assessment only as a source term for the Avid Angler exposure scenario. Evaluation of exposure via fish ingestion is also organized according to these data groups, as discussed in Section 5.3.15.

5.3.5 Dust in Ambient Air

Exposure point concentrations related to windborne soil particulates in ambient air (e.g., dust) are estimated using a screening-level model that relates soil concentrations to the concentration of respirable particles in air. The specific model that is used for these calculations is EPA's particulate emission factor (PEF) model. The PEF model for wind erosion can be used to estimate annual average concentrations of respirable particulates (approximately 10 μm and less) in ambient air (EPA/540/R-95/128, EPA 2002b). The PEF model has two components. The first component is an atmospheric dispersion term (Q/C_{wind}) that relates air concentrations to particulate emissions. The second component is a particulate emission model related to wind erosion.

The ratio of the concentration of respirable particulates in air to the particle flux from the ground is represented in the PEF model by the Q/C_{wind} term, which is defined as the inverse of this ratio. The Q/C_{wind} term is derived from EPA modeling using the industrial source complex air dispersion model in short-term mode for a variety of source sizes and meteorological conditions. For the local-area risk calculations related to excavated waste sites, a Q/C_{wind} value of 84 $\text{g}/\text{m}^2\text{-sec}$ per kg/m is calculated for the hypothetical exposure area (1,500 m^2 or 0.37 acres; see Section 5.3.2). Because the actual area associated with unexcavated RSVP sites was unavailable at the time this report was prepared, an area of 1,500 m^2 has also been uniformly applied to these sites. For the broad-area risk calculations, a Q/C_{wind} value of 28 $\text{g}/\text{m}^2\text{-sec}$ per kg/m is calculated, corresponding to an area of 2,023,500 m^2 (500 acres)⁴.

⁴This is the largest area in the range of sites used by EPA (2002, Appendix D) for developing the model for the Q/C_{wind} term.

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The Q/C_{wind} calculations are based on a least squares curve fit of site size and dust concentration performed by EPA for 29 sites and documented in Appendix D of EPA (2002b). The resulting equation, provided as Exhibit D-1 of EPA (2002b) is:

$$Q/C_{\text{wind}} = A \times \exp\left[\frac{(\ln A_{\text{site}} - B)^2}{C}\right]$$

where Q/C_{wind} = inverse of the mean particulate concentration at the center of a square source area per unit particulate flux ($\text{g}/\text{m}^2\text{-sec}$ per kg/m)

A_{site} = area of site (acres)

A, B, C = curve fitting constants

exp = the exponent applied to the base of the natural logarithm "e."

The average values for the constants A, B, and C cities in a climatic zone that includes the cities of Boise, Idaho; Winnemucca, Nevada; Salt Lake City, Utah; Casper, Wyoming; and Denver, Colorado are used to represent the Hanford region. These averages, calculated from the individual values in Exhibit D-2 of EPA (2002b), are as follows:

A: 11.1906

B: 21.4867

C: 250.8165

The wind erosion component of the PEF model is composed of the remaining terms in the PEF equation. The form of the PEF model is obtained from EPA's Soil Screening Guidance (EPA/540/R-95/128, EPA 2002b). The derivation of the interim term for equivalent threshold value of windspeed at 7 m height was obtained from Appendix D of the same guidance.

$$\text{PEF}_{\text{wind}} = Q/C_{\text{wind}} \times \frac{3600 \text{ sec/hr}}{0.036 \times (1-v) \times (U_m/U_{t-7})^3 \times F(x)}$$

where PEF_{wind} = particulate emission factor for wind-generated erosion (m^3/kg)

Q/C_{wind} = inverse of the mean particulate concentration at the center of a square source area per unit particulate flux ($\text{g}/\text{m}^2\text{-sec}$ per kg/m^3)

U_m = mean annual windspeed (m/sec)

v = fraction of vegetative cover (dimensionless)

$F(x)$ = function dependent on U_m/U_{t-7} (dimensionless).

and

$$U_{t-7} = U_t/0.4 \times \ln(700 \text{ cm}/z_0)$$

where U_{t-7} = equivalent threshold value of windspeed at 7-m height (m/sec)

U_t = threshold friction velocity (m/sec)

Z_0 = surface roughness height (cm).

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A mean annual wind speed of 3.4 m/s for the Hanford Site is used. The other site-specific inputs are fraction of vegetative cover and surface roughness height. The fraction of vegetative cover, estimated from information obtained for the ecological risk assessment component of the RCBRA, is 30%. A value of 3, which is an average value for “grassland” (Cowherd et al. 1984, Figure 3-6), is used for surface roughness height.

Values of $F(x)$ will be estimated based on the function shown in Figure 4-3 of Cowherd et al. (1984). The value of x is calculated as,

$$0.886 \times (U_{t.7} / U_m)$$

and the function $F(x)$ is approximated using the following equations (equations fit by visual approximation to the graphic in Figure 4-3 of Cowherd et al. (1984).

$$\begin{aligned} \text{when } x < 1, F(x) &= (6 - x^3)/\pi \\ \text{when } x \geq 1 \text{ and } < 2, F(x) &= (-1.3 \times x) + 2.89 \\ \text{when } x \geq 2, F(x) &= [(8 \times x^3) + (12 \times x)] \times e^{-(x^2)} \end{aligned}$$

A value of 0.625 m/s at the ground surface is applied for U_t (EPA/540/R-95/128, Appendix D).

Exposure point concentrations of each contaminant in air are calculated by dividing the soil concentration by the PEF value as follows:

$$EPC_{\text{air}} (\text{mg}/\text{m}^3) = C_{\text{soil}} (\text{mg}/\text{kg}) / \text{PEF} (\text{m}^3/\text{kg})$$

The calculation is analogous, with unit corrections, for radionuclides. Because any residual VOCs in soil can be assumed to be largely dissipated by the process of exhuming waste soils and distributing them on the ground surface (see Section 5.3.2), risk calculations for the dust inhalation pathway do not address VOCs.

Inhalation risks based solely on the local area soil source term are used in the local area risk calculations for the Rural-Residential and CTUIR exposure scenarios in order to focus the inhalation risk calculation on residual contamination for the specific waste site. Because airborne dust surrounding a residence may include contributions from surrounding areas, the broad area risk calculations for the Rural-Residential and CTUIR exposure scenarios include a sum of exposure to both local area and broad area airborne dusts.

5.3.6 Groundwater

Exposure point concentrations in groundwater are calculated for each individual monitoring well sampled under the SAP (DOE/RL-2005-42). As discussed in Section 5.2.2, pathway-specific health risks related to groundwater contamination for the Rural-Residential, Resident Monument Worker, and CTUIR exposure scenarios will be calculated for each monitoring well in the reactor areas and 300 Area. The range of potential groundwater-related risks across the monitoring wells in the reactor areas and 300 Area will be described for these exposure scenarios in Section 5.8. Unfiltered RCBRA groundwater data are used to represent current groundwater

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constituent concentrations. Background calculations for groundwater are performed using the summarized groundwater concentration information available for metals and radionuclides in the executive summary of *Hanford Site Background: Part 3, Groundwater Background* (DOE/RL-96-91).

5.3.7 Seeps and River Water

As indicated in Table 5-1, exposure to chemicals in seeps and river water is potentially associated with sweat lodge exposure pathways in the CTUIR exposure scenario. Seeps at the river bank are indicated by areas of thicker vegetation in the shoreline area rather than as free-flowing sources of water, and regular collection of such water for sweat lodge use is unlikely given seasonal constraints on flow and the close proximity of the Columbia River for readily obtaining surface water. With respect to surface water, evaluation of water data from regions where groundwater plumes emerge at the Columbia River (see Section 4.4) indicate only a few instances where surface water results appear to be impacted. As described in Section 4.2, most surface water data were collected by PNNL for special characterization purposes and likely do not reflect normal mixing that would rapidly dilute emerging groundwater. For these reasons, and because groundwater data are available to protectively characterize subsurface water that may be expressed at the ground surface, only groundwater data are used to calculate human health risks via exposure to water.

5.3.8 Indoor Air (VOCs and Domestic Water Use)

Indoor air concentrations of VOCs related to domestic uses of contaminated groundwater can be estimated using models described in EPA's *Volatilization Rates from Groundwater to Indoor Air, Phase II* (EPA 600/R-00/096). This guidance describes the development of models for estimating chemical emissions from washing machines, dishwashers, showers, and bathtubs. The models make use of source- and chemical-specific mass transfer coefficients, and air exchange rates for the shower and appliances, to estimate VOC releases from water to air. However, detected organic chemicals in the groundwater samples collected under the SAP (DOE/RL-2005-42) were limited primarily to pesticides and phthalates. VOCs were not among the detected organic chemicals, although other groundwater sampling has indicated the presence of VOCs at certain locations in the 100 Area and 300 Area. Therefore, while the protocol described in the following paragraphs is established for estimating VOC concentrations in indoor air, it has not been implemented in this risk assessment.

In *Volatilization Rates from Groundwater to Indoor Air, Phase II* (EPA 600/R-00/096), mass transfer coefficients and air exchange rates were measured for 113 experiments involving 5 tracer chemicals (acetone, ethyl acetate, toluene, ethylbenzene, and cyclohexane) and 4 sources (showers, bathtubs, washing machines, and dishwashers). Experimental results included chemical stripping efficiencies for each source, mass transfer coefficients (overall, liquid-phase, gas-phase), and an assessment of the importance of gas-phase resistance to mass transfer. Stripping efficiencies ranged from 6.3% to 80% for showers, 2.6% to 69% for bathtubs, 18% to 100% for dishwashers, and 3.8% to 100% for washing machines. Acetone and cyclohexane always defined the lower and upper bounds, respectively, of these ranges.

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A set of protocols for estimating emission rates for other chemicals is defined in *Volatilization Rates from Groundwater to Indoor Air, Phase II* (EPA 600/R-00/096) for each of the four sources. However, some of the necessary chemical-specific inputs to implement these protocols are unavailable. Also, variability related to the construction of washing machines and dishwashers, frequency of events for each source, water temperature, and house and room ventilation rates can greatly affect final VOC air concentrations (EPA 600/R-00/096). For these reasons, semiquantitative estimates of VOC indoor air concentrations will be generated to provide a screening-level evaluation of potential exposure and health effects via this pathway.

For bathing, VOC stripping efficiencies for showers representing the upper end of the experimental ranges measured for the five chemicals evaluated in EPA 600/R-00/096 (Table 9-1) are as follows:

- Acetone – 16% (Henry's constant = 0.00159)
- Ethyl acetate – 36% (Henry's constant = 0.00574)
- Ethylbenzene – 75% (Henry's constant = 0.323)
- Toluene – 77% (Henry's constant = 0.272)
- Cyclohexane – 80% (Henry's constant = 8.2).

VOC Henry's constants, obtained from a companion database to EPA's *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA 2005), are summarized in Table 5-3. When specific VOCs were not included in this database, Henry's constants were researched in the peer-reviewed Hazardous Substances Data Base managed by the National Library of Medicine in their Toxicology Data Network (NLM 2007).

For VOCs other than the five studied in EPA 600/R-00/096, stripping efficiencies will be assigned based on Henry's constant from interpolation within this range. A review of the relationship between Henry's constant and VOC stripping efficiency, plotted for these five VOCs in Figure 5-3, indicates that the data are inadequate to derive a statistical regression. For values of Henry's constant above approximately 0.25, the stripping efficiency appears to have reached a ceiling of approximately 75% to 80%. Unfortunately, between Henry's constant of about 0.005 to 0.25, there are no data for stripping efficiency. Based on the available data, the following bins will be used to assign VOC stripping efficiency to calculate indoor air concentrations:

Henry's constant < 0.0025:	25% efficiency
Henry's constant < 0.25 and > 0.0025:	50% efficiency
Henry's constant > 0.25:	80% efficiency

The calculation of stripping efficiency and indoor air exposure to chemicals in groundwater is limited to VOCs because the available data for stripping efficiencies is limited to this class of compounds. Furthermore, assignment of a 25% stripping efficiency to chemicals with very low volatility is liable to grossly overestimate potential inhalation exposures. Consistent with EPA screening protocol, VOCs are defined as organic chemicals having a Henry's constant larger than 1×10^{-5} atmosphere/m³ – mol and a molecular weight below 200 g/mol.

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The steady-state VOC emission rate during showering is calculated as:

$$\text{Emission}_{\text{VOC}} = C_w \times Q_{\text{H}_2\text{O}} \times \text{Eff}_{\text{strip}}$$

where $\text{Emission}_{\text{VOC}}$ = emission rate of VOC into shower stall (mg/hr)

C_w = concentration of VOC in domestic well water (mg/L)

$Q_{\text{H}_2\text{O}}$ = volumetric water flow rate (L/hr)

$\text{Eff}_{\text{strip}}$ = chemical-specific VOC stripping efficiency (unitless)

A volumetric flow rate of 9 L/min (540 L/hr) is used based on the high-end flow rate employed during the shower stall experiments described in EPA (2001b).

Steady-state exposure point concentrations in shower stall air are calculated according to:

$$\text{EPC}_{\text{shower}} = \text{Emission}_{\text{VOC}} / (\text{ACH} \times V)$$

where $\text{EPC}_{\text{shower}}$ = exposure point concentration in shower stall air (mg/m³)

ACH = air exchange rate in shower (1/hr)

V = shower stall volume (m³)

An air exchange rate of 12 per hour and shower stall volume of 1.7 m³ are used based on the measured values of these parameters for the shower stall experiments described in EPA (2001b).

In addition to exposure to VOCs in domestic water while bathing, potential exposures related to other water uses (such as dishwashers and washing machines) may contribute to indoor VOC exposures. For reasons described above, quantification of exposure via these other sources is subject to a high degree of uncertainty.

5.3.9 Sweat Lodge Air (Surface or Groundwater)

Appendix 4 of Harris and Harper (2004) provides an exposure assessment methodology for calculating concentrations of volatile and nonvolatile chemicals in air associated with the use of contaminated water in a sweat lodge. Contaminants are assumed to be introduced into the sweat lodge predominately through the water poured over heated rocks that is used to create steam. Equations are provided for both volatile and nonvolatile contaminants. For volatile and SVOCs, Equation 7 of Harris and Harper (2004) provides the following method for calculating air-phase concentrations in the sweat lodge:

$$\text{EPC}_{\text{air}} = C_w \cdot \left(\frac{V_{w,\text{total}}}{2} \right) \cdot \left(\frac{1}{\frac{2}{3} \cdot \pi \cdot r^3} \right)$$

where EPC_{air} = exposure point concentration in air (mg/m³)

C_w = concentration of volatile or semivolatile compound in water (mg/L)

$V_{w,\text{total}}$ = total volume of water used to create steam during the lodge (L)

r = radius of a hemispherical lodge (m).

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The last term in the equation represents the internal volume of the sweat lodge. Values for $V_{w,\text{total}}$ (4 L) and r (1 m) were obtained from Table 1 of Appendix 4 of Harris and Harper (2004). The equation for volatile and semivolatile chemicals will be used for all organic chemicals.

For nonvolatile chemicals, including metals and radionuclides (except tritium), Appendix 4 of Harris and Harper (2004) provides a different approach to estimating air-phase contaminant concentrations. In this approach, the quantity of nonvolatile constituents in the air phase is limited to that which is carried by the volume of liquid water used to create saturated conditions in the lodge rather than by the total volume of water used. The volume of water needed to saturate the volume of air in the lodge is calculated using the Ideal Gas Law. Harris and Harper (2004) provides the full derivation. The resulting method from Equation 14 of Harris and Harper (2004) is:

$$EPC_{\text{air}} = C_w \cdot \left(\frac{MW_w}{R \cdot T \cdot \rho_w} \right) \cdot \ln(p^*)$$

where EPC_{air} = exposure point concentration in air (mg/m^3)
 C_w = concentration of nonvolatile contaminant in water (mg/L)
 MW_w = molecular weight of water (18.0 g/gmole)
 R = ideal gas law constant (0.06237 ($\text{mmHg} \cdot \text{m}^3$)/($\text{gmole} \cdot \text{K}$))
 T = temperature of the sweat lodge (K)
 ρ_w = density of liquid water (1000 g/L)
 p^* = partial pressure of water at temperature T (mmHg)

and, using the Antoine equation for estimating the vapor pressure of water at lodge temperature,

$$\ln(p^*) = \left(18.3036 - \frac{3816.44}{T - 46.13} \right)$$

The temperature of the sweat lodge is set at 150 °F (339 °K)⁵, from Table 2 of Appendix 4 of Harris and Harper (2004).

5.3.10 Garden Produce

As footnoted in Table 5-1, exposure point concentrations in garden produce cannot be directly represented by representative concentrations. Contaminant concentrations in garden produce (e.g., fruits and vegetables) are estimated from the “local area” surface soil exposure point concentrations described above.

Exposure point concentrations in plant tissues are calculated based on root uptake using plant-soil concentration ratios (K_{p-s}) obtained from two published sources. For radionuclides and metals, suggested values of K_{p-s} for composite plants were obtained from supporting documentation for the RESRAD computer code (Wang et al. 1993, Table 9). These values, standardized as dry weight using a wet-to-dry weight conversion factor of 0.15 (EPA/600/P-95/002Fb, Table 9-27), are reproduced for convenience in Table 5-4. If a

⁵ Table 2 of Appendix 4 of Harris and Harper (2004) has a typographical error showing 389 °K.

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radionuclide or metal was unlisted in Table 9 of Wang et al. values of K_{p-s} were obtained directly from Version 6.3 of RESRAD or, if the element is not included in the RESRAD library, from Table 2.1 of Baes et al. (1984).

For organic chemicals, above-ground plant tissue concentrations are calculated as a function of a chemical's octanol-water partition coefficient based on the methodology described in Sections 5.3.3 and A-2 of EPA's *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA/530-R-05-006). The regression for calculating K_{p-s} for an organic chemical, published as Equation A-2-17 of EPA/530-R-05-006 is:

$$\log K_{p-s} = 1.588 - 0.578 (\log K_{ow})$$

Values of the octanol-water partition coefficient for organic chemicals, obtained from the companion database to EPA/530-R-05-006, are provided in Table 5-5. When specific organic chemicals were not included in this database, $\log K_{ow}$ values were researched in the peer-reviewed Hazardous Substances Data Base managed by the National Library of Medicine in their Toxicology Data Network (NLM 2007). Organic chemical K_{p-s} values are provided in Table 5-4. Additionally, the ratio of soil concentrations and plant tissue concentrations collected from the MIS investigation areas may be employed in the uncertainty analysis to qualify the calculated K_{p-s} values.

Plant-soil ratios are used to calculate produce exposure point concentrations as follows:

$$EPC_{\text{produce}} \text{ (mg/kg wet produce)} = (C_{\text{soil}} \times K_{p-s}) \times CF$$

where C_{soil} = contaminant concentration in soil (mg/kg soil)

K_{p-s} = plant-soil concentration ratio (mg/kg dry plant per mg/kg soil)

CF = wet to dry weight conversion factor (unitless)

In order for the wet-weight ingestion rate values described in *Exposure Factors Handbook, Volume II – Food Ingestion Factors* (EPA/600/P-95/002Fb) to be consistent with the dry-weight concentrations represented by K_{p-s} , the dry-weight concentrations must be converted to a wet-weight basis. A wet-to-dry weight conversion factor of 0.15 is used for this purpose, based on moisture content information for common fruits and vegetables (EPA/600/P-95/002Fb, Table 9-27).

Plant-soil concentration ratios reflect an assumption that there is a linear and unchanging relationship between soil and plant tissue concentrations. For this reason, K_{p-s} values are liable to overestimate plant tissue concentrations when soil concentrations are high. If this is suspected of contributing significant uncertainty in the risk assessment results, regression models to relate plant tissue metal concentrations to soil metal concentrations may be employed to estimate the degree of potential bias. Soil-plant regression models have been developed by various authors. For example, in *Empirical Models for the Uptake of Inorganic Chemicals by Plants* (BJC/OR-133), regressions were published for eight metals: arsenic, cadmium, copper, lead, mercury, nickel, and selenium. The general form of the BJC/OR-133 regression models is:

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$$\ln(C_{\text{plant}}) = [\ln(C_{\text{soil}}) \times B_1] + B_0$$

where C_{plant} = concentration in plant (mg/kg dry plant)
 C_{soil} = concentration in soil (mg/kg dry soil)
 B_1 and B_0 = regression fitting parameters (unitless).

Evaluation of uncertainty in estimated plant tissue concentrations using such regression analyses may be incorporated into the uncertainty analysis if plant ingestion is determined to be a potentially significant exposure pathway.

Exposure point concentrations for garden produce are related only to soil exposure point concentrations and do not account for potential contribution from the use of contaminated groundwater for irrigation. A protective screening of the potential impacts to groundwater related to waste site contamination was incorporated into the interim remedial action criteria used for waste site remediation (DOE/RL-96-17). The potential significance of existing groundwater contamination for exposure via domestic uses of groundwater will be evaluated in the risk assessment to provide information to decision makers on the relative importance of soil and groundwater sources of contamination for a hypothetical future residential user. However, dynamic modeling of soil and biota contaminant concentrations over time related to agricultural uses of groundwater is beyond the scope of this assessment. The potential significance of agricultural uses of groundwater will be qualitatively addressed in the uncertainty analyses of the risk assessment for each operational area.

5.3.11 Native Plants

Exposure point concentrations for contaminants in the tissues of native plants will be calculated directly from the RCBRA plant data, as described in Section 4.0. Additional sources of information relating to contaminant concentrations in wild plants have been summarized in Appendix C of the *Risk Assessment Work Plan for the 100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment* (DOE/RL-2004-37). Plant tissue data collected under the SESP include a variety of upland and riparian vegetation and fruit, as well as the aquatic plant milfoil. These data may not be specifically associated with known areas of residual contamination, and in some cases may reflect levels of contamination that represent pre-remediation rather than current conditions. However, they are still likely to be informative in that they provide a time series of measured plant concentrations with which to benchmark the "snapshot" data collected under the SAP (DOE/RL-2005-42).

Plant-soil ratios will be used to calculate exposure point concentrations in wild plants as follows:

$$\text{EPC}_{\text{plant}} \text{ (mg/kg wet plant)} = C_{\text{plant}} \times \text{CF}$$

where C_{plant} = contaminant concentration in plant (mg / kg dry weight)
 CF = wet weight conversion factor (dry weight / wet weight)

Exposure point concentrations in plants are expressed on a dry-weight basis in the data base. In order for the wet-weight ingestion rate values described in Section 5.4 to be consistent with these

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dry-weight concentrations, the dry-weight concentrations must be converted to a wet-weight basis. A wet weight conversion factor of 0.15 kg dry weight per kg wet weight is used for this purpose, based on moisture content information for common fruits and vegetables (EPA/600/P-95/002Fb, Table 9-27).

5.3.12 Poultry and Eggs

Evaluation of human exposure to environmental contaminants via animal products has not traditionally been common in Superfund risk assessments. Relatively recent EPA guidance for conducting human health risk assessment for hazardous waste incinerators addresses transfer factors for chicken tissue and eggs for organic chemicals but for only a limited number of metals including mercury compounds, cadmium, selenium, and zinc (EPA/530-R-05-006). Evaluation of exposure via animal products (particularly beef and milk) has long been routine in radiation dose assessment. Because it is desirable to use a common protocol and reference for specifying metal and radionuclide transfer factors, values for poultry and egg transfer factors for both metals and radionuclides were identified in guidance for radionuclide dose assessment. Specifically, exposure point concentrations for metals and radionuclides in poultry meat and eggs will be modeled from soil exposure point concentrations using transfer factors published in Table 6.18 of *Residual Radioactive Contamination from Decommissioning – Technical Basis for Translating Contamination Levels to Annual Total Effective Dose Equivalent: Final Report* (NUREG/CR-5512-V1). These values are reproduced for convenience in Table 5-6.

For organic chemicals, poultry and egg transfer factors will be calculated as a function of a chemical's octanol-water partition coefficient based on the methodology described in Appendix A2 of EPA's *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA/530-R-05-006). This methodology employs general transfer factors that relate to fat tissue in a variety of animals, which are then made specific based on the defined fat content of a particular animal or tissue. The regression for calculating a poultry biotransfer factor ($B_{a(p)}$) for an organic chemical, published as Equations A-2-27 and A-2-28 of EPA/530-R-05-006 is:

$$B_{a(p)} = 10^{\log B_{\text{fat}}} \times \text{fat content.}$$

where B_{fat} = biotransfer factor for fat tissue (mg / kg fat per mg/day)

and

$$\log B_{\text{fat}} = -0.099 (\log K_{\text{ow}})^2 + 1.07 \log K_{\text{ow}} - 3.56.$$

The equation for $\log B_{\text{fat}}$ is published as Equation A-2-21 of EPA/530-R-05-006. The fat content of chicken and eggs used in this equation is 14% and 8%, respectively (EPA/530-R-05-006, A2-2.13.3). Values of the octanol-water partition coefficient for organic chemicals, obtained from the companion database to EPA/530-R-05-006, are provided in Table 5-5. $B_{a(p)}$ values for organic chemicals are provided in Table 5-6.

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Transfer factors describe the relationship between intake of a contaminant in feed and the associated tissue concentration in an animal. In this risk assessment, the poultry transfer factors will be applied to uptake of metals and radionuclides in soil rather than feed. It is assumed that chicken feed is store-bought, rather than produced from grain grown on-site, and that exposure to soil contaminants for free-ranged chickens is a result solely of their foraging habits. Because the relative bioavailability of specific metals may be expected to vary between soil and feed, and there is limited chemical-specific information to define soil bioavailability, the use of feed-based transfer factors is a source of uncertainty. Of equal concern is that transfer factors, like the K_{p-s} values described above, reflect an assumption that tissue concentrations increase linearly with soil concentrations. They are therefore also likely to overestimate tissue concentrations when soil metal or radionuclide concentrations are high. Finally, transfer factors are applicable only when tissue concentrations have equilibrated with intake rates (Ward and Johnson 1986) – the time needed for such equilibration in chicken meat and eggs has not been determined.

The chicken meat and egg transfer factors are used to calculate exposure point concentrations according to the methodology described in Section 5.6 of EPA (1995b). Because poultry uptake is limited to soil ingestion, the calculation of fresh-weight exposure point concentrations in poultry (meat and eggs) can be expressed as:

$$EPC_{\text{poultry}} \text{ (mg/kg)} = C_{\text{soil}} \times B_{a(p)} \times (UR_{fd} \times F_{s,fd} \times B_s)$$

where C_{soil} = concentration of contaminant in soil (mg/kg)

$B_{a(p)}$ = feed-chicken or feed-egg transfer factor for poultry (mg/kg fresh chicken meat per mg/day, or mg/kg fresh egg per mg/day)

UR_{fd} = uptake rate of dry feed (kg feed/day)

$F_{s,fd}$ = fraction of soil in diet (kg soil/kg feed)

B_s = soil bioavailability factor (unitless).

An uptake rate of feed for laying hens of 0.1 kg/day (Ng et al. 1982) will be used in the risk assessment. A different value of 0.2 kg/day is published by EPA/530-R-05-006, but it is judged more appropriate to use the value cited in Ng et al. (1982) because this reference is also the original source of many of the poultry transfer factors cited in NUREG/CR-5512-V1. A value of 10% (0.1) will be used for the fraction of soil in the diet based on a recommendation in EPA/530-R-05-006 (Section 5.6). In the absence of chemical-specific data, values of B_s will be assumed to be 1.0 as recommended in EPA/530-R-05-006. (See Section 5.4.2 for information on B_s values.) The origin of the $B_{a(p)}$ values is described above.

As discussed in relation to the calculation of exposure point concentrations for garden produce, the potential significance of agricultural uses of contaminated groundwater (in this case, the use of well water for watering poultry) will be qualitatively addressed in the uncertainty analyses of the risk assessment for each operational area.

5.3.13 Beef and Milk

The modeling of exposure point concentrations in beef and milk is conceptually similar to that described for poultry and eggs. In both cases, transfer factors are used to describe the

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relationship between intake of a contaminant in feed and the associated tissue concentration in an animal. In the case of poultry, all contaminant intake was assumed to occur due to ingestion of soil particulates. Cattle may also be exposed to soil contaminants via direct ingestion of soil while grazing. Additionally, cattle may be exposed via ingestion of plants growing on contaminated soils.

Two variations of these exposure pathways are evaluated in the risk assessment. In the first case, beef cattle or milk cows are assumed to be free-ranged over an operational area. They may be exposed to soil contaminants via direct soil ingestion while grazing and via plants growing on contaminated soil. This corresponds to the "broad area" assessment of these exposure pathways shown in Table 5-2. In the second variation, beef cattle or milk cows are assumed to be penned near a homestead. They may be exposed to soil contaminants via ingestion of home-grown fodder. The primary source of soil contaminants is assumed to be from redistribution to a 1,500-m² area of the ground surface during basement excavation for the homestead (see Section 5.3.2). This corresponds to the "local area" assessment of these exposure pathways shown in Table 5-2. The area within which soil concentrations will be averaged for penned cattle is 2 ha (20,000 m²), which is associated with an area factor of 1.0 for the beef and milk ingestion pathways in the RESRAD computer code (ANL-EAD-4, Section D.2.1.2). Soil contaminant concentrations in the cattle enclosure are thus the area-averaged sum of local and broad area values, as follows:

$$EPC_{\text{pen}} = [EPC_{\text{local}} \times (1500 \text{ m}^2 / 20000 \text{ m}^2)] + [EPC_{\text{broad}} \times (1 - (1500 \text{ m}^2 / 20000 \text{ m}^2))]$$

Suggested values of the feed-beef and feed-milk transfer factors for metals and radionuclides are obtained from supporting documentation for the RESRAD computer code (Wang et al. 1993, Tables 11 and 12). If a radionuclide or metal was unlisted in Tables 11 and 12 of Wang et al. transfer factor values were obtained directly from Version 6.3 of RESRAD or, if the element is not included in the RESRAD library, from Tables 2.24 and 2.25 of Baes et al. (1984). For organic chemicals, beef and milk transfer factors will be calculated as a function of a chemical's octanol-water partition coefficient based on the methodology described in Appendix A2 of EPA's *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA/530-R-05-006). Both the methodologies and uncertainties related to these transfer factors are similar to those described for the transfer factors employed for poultry. However, the fat content of beef and milk used in the equation described in Section 5.3.12 is 19% and 4%, respectively (EPA/530-R-05-006, A2-2.13.1). Values of the beef and milk biotransfer factors for metals, radionuclides, and organic chemicals are provided in Table 5-7.

The beef and milk transfer factors are used to calculate exposure point concentrations according to the methodology described in Section 5.4 of EPA (1995b). For free-ranging cattle, the calculation of fresh-weight exposure point concentrations in beef and milk can be expressed as:

$$EPC_{\text{cattle}} \text{ (mg/kg)} = C_{\text{soil}} \times B_{a(c)} \times [(UR_g \times K_{g-s}) + (UR_s \times B_s)] \times MF$$

where C_{soil} = concentration of contaminant in soil (mg/kg)

$B_{a(c)}$ = feed-beef or feed-milk transfer factor for cattle (mg/kg fresh beef per mg/day, or mg/kg milk per mg/day)

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UR_g = uptake rate of forage plants (grass) by cattle, dry weight (kg/day)

K_{g-s} = grass-soil concentration ratio (mg/kg dry grass per mg/kg soil)

UR_s = uptake rate of soil by cattle (kg soil/d)

B_s = soil bioavailability factor (unitless)

MF = metabolism factor (unitless).

For penned livestock, the calculation of exposure point concentrations in beef and milk can be expressed as:

$$EPC_{\text{cattle}} \text{ (mg/kg)} = C_{\text{soil}} \times B_{a(c)} \times (UR_g \times K_{g-s}) \times MF.$$

EPA/530-R-05-006 provides food uptake rates for cattle separately for forage, grain, and silage. For simplification, all feed for free-ranged cattle will be assumed to be obtained from forage. For penned cattle, 100% of feed will be assumed to consist of grasses such as alfalfa grown by the farmer. The EPA/530-R-05-006 recommended value of 12 kg dry weight of feed per day will be used in the risk assessment. For metals and radionuclides, suggested values of K_{g-s} will be obtained from supporting documentation for the RESRAD computer code (Wang et al. 1993, Table 10). Values of K_{g-s} for organic chemicals are identical for vegetables and forage (EPA/530-R-05-006, A2-2.12.3). A cattle soil uptake of 0.5 kg/day will be used based on EPA recommendation (EPA 2005). In the absence of chemical-specific data, values of B_s and MF will be assumed to be 1.0 as recommended in EPA/530-R-05-006. (*See Section 5.4.2 for information on B_s values.*) The origin of the $B_{a(c)}$ values is described above.

As discussed in relation to the calculation of exposure point concentrations for garden produce, the potential significance of agricultural uses of contaminated groundwater (in this case, the use of well water for watering cattle and growing fodder) will be qualitatively addressed in the uncertainty analyses of the risk assessment for each operational area.

5.3.14 Wild Game

Estimation of exposure point concentrations in wild game, such as mule deer or elk, can be approached in a manner analogous to that described for free-range cattle in Section 5.3.13. The tissue concentrations of contaminants for free-ranged cattle, modeled from the soil data described in Sections 5.3.2 and 5.3.3, will also be employed as estimates for mule deer and elk.

Existing potential sources of information relating to contaminant concentrations in wild game have been summarized in Appendix C of the *Risk Assessment Work Plan for the 100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment* (DOE/RL-2004-37). Animal tissue data collected under the SESP include mule deer and cottontail rabbit. Data are also available for game birds such as pheasant and quail. These data may not be specifically associated with known areas of residual contamination, and in some cases may reflect levels of contamination that represent pre-remediation rather than current conditions. However, these data are useful because they provide some information on actual contaminant tissue burdens in hunted species such as rabbit, mule deer, and game birds. In the case of mule deer, the environmental surveillance data are used in the uncertainty analysis to benchmark the modeled tissue concentrations in free-ranged beef cattle.

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5.3.15 Food Fish

As part of the SAP (DOE/RL-2005-42), tissue data from sculpin, macroinvertebrates (including crayfish), and asiatic clams were obtained from areas along the Hanford Reach of the Columbia River. Additionally, tissue data collected during the 100-B/C Pilot Project (DOE/RL-2005-40) and the 100-NR-2 investigation (DOE/RL-2005-22) are integrated in this assessment. Analytical data in fish tissue for the 100-B/C Pilot Project are limited to strontium-90, technetium-99, Aroclor-1254, and Aroclor-1260. Analytical data in fish tissue for the 100-NR-2 investigation are limited to strontium-90 and technetium-99. For a broader spatial scale and historical perspective on contaminant levels in fish tissue, analyte concentrations described in SESP reports and the EPA report *Columbia River Basin Fish Contaminant Survey 1996-1998* (EPA 910-R-02-006) are also discussed in the uncertainty analysis.

RCBRA sampling sites are located within the 300 Area (uranium plume), 100-D and 100-K Areas (chromium plumes), 100-N Area (strontium plume), and 100-B/C Area. The sites sampled under the SAP (DOE/RL-2005-42) were selected based on the locations of known groundwater plumes, the results of a 2005 Columbia River conductivity survey for identifying areas of groundwater discharge to the river, and the results of past biota sampling locations and results. Hence, these data provide protectively biased estimates of tissue concentrations in localized aquatic species.

The RCBRA tissue data from each of the three types of aquatic biota described above (sculpin, macroinvertebrates, and clams) have been employed to calculate exposure point concentrations for the fish ingestion exposure pathway. In the case of sculpin, which are the primary data used in the risk assessment, the tissue data are employed as a protective surrogate for more commonly fished resident species. Fish ingestion risks calculated using data for macroinvertebrates (crayfish) and clams are discussed in the uncertainty analysis. Representative concentrations for the mean and UCL of fish tissue data were calculated according to methods described in Section 4.0.

As described in Section 4.4 and discussed in the context of the human health risk assessment in Section 5.3.4, concentrations of certain analytes in aquatic tissue samples were higher in the 100-N Area (strontium-90) and 300 Area (total and isotopic uranium) than in other operational areas sampled. Exposure point concentrations are calculated separately for these locations, which represent unique potential exposure areas for fishing by virtue of differences in the exposure point concentrations. Across both sediment and fish samples, separate risk calculations are conducted for the following data groups:

1. 100 Area (inclusive of all 100 Area sculpin fish tissue data)
2. 300 Area
3. 100-N Area (data collected under the 100-NR-2 investigation)
4. 100-B/C Area (data collected under the 100-B/C Pilot investigation).

Existing potential sources of information relating to contaminant concentrations in fish have been summarized in Appendix C of the *Risk Assessment Work Plan for the 100 Area and*

300 Area Component of the River Corridor Baseline Risk Assessment (DOE/RL-2004-37). Fish tissue data exist for bass, carp, sculpin, sucker, whitefish, clams, and crayfish. These data may or may not be associated with known areas of groundwater impacts to the Columbia River. However, even if such association cannot be determined, these data are useful because they represent a time series of concentrations collected under differing river conditions. In particular, a number of the same species are represented that were collected under the SAP (DOE/RL-2005-42). Therefore, the range of analytical results in these data are discussed relative to the data used in the risk calculations in the uncertainty analysis of this risk assessment.

5.4 EXPOSURE ASSESSMENT: CALCULATION OF INTAKE

5.4.1 Overview

As discussed in *Risk Assessment Guidance for Superfund, Human Health Evaluation Manual, Part A* (EPA 1989), the RME estimate is generally the principal basis for evaluating potential risks at a Superfund site. Later EPA guidance (EPA 1992c, 1995b) recommended including CTE estimates in addition to RME estimates of risk. In general, an RME estimate of risk is at the high end of a risk distribution (90th to 99.9th percentiles), whereas the CTE estimate is associated with the mean or 50th percentile of a risk distribution (*Risk Assessment Guidance for Superfund: Volume 3 – Part A, Process for Conducting Probabilistic Risk Assessment* [EPA 540-R-02-002]). An RME scenario assesses risk to individuals whose behavioral characteristics may result in much higher potential exposure than seen in the average individual. A CTE scenario assesses potential risk to an average member of the population. The inclusion of both RME and CTE calculations provides a semiquantitative measure of the range of expected risks that may occur under a particular exposure scenario.

The basic structure of the exposure equations used in this assessment were obtained from *Risk Assessment Guidance for Superfund, Human Health Evaluation Manual, Part A* (EPA 1989). The general intake equation for chemicals that serves as the basis for the pathway-specific equations is:

$$\text{Chemical Intake} = \frac{C_i \times CR \times EF \times ED}{BW \times AT}$$

where Intake = rate of chemical available for uptake at an exchange boundary (mg/kg body weight/day)

C_i = concentration of chemical i at exposure point (e.g., mg /kg soil, mg/L water)

CR = contact rate with the environmental medium (e.g., mg soil ingestion /day; L water ingestion per day)

EF = exposure frequency (days/yr)

ED = exposure duration (year)

BW = body weight (kg)

AT = averaging time for toxicological effects (days).

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Intake is specified as the rate at which a chemical becomes available for uptake at an exchange boundary, such as the walls of the gastrointestinal tract or the skin. Hence, it is not equivalent to an absorbed dose, which is the amount of chemical actually entering the bloodstream across an exchange boundary.

Separate intake calculations are performed for adults and children when evaluating noncarcinogenic effects, because the averaging time over which effects are assessed is equal to the exposure duration (EPA 1989). However, because cancer risk is expressed as a probability averaged over a lifetime, exposure as a child and adult is integrated in these intake calculations.

Intake for radiation risk and dose is calculated in a somewhat different manner than either cancer risk or radiation hazard. As described in Chapter 10 of EPA (1989), the general intake equation for radiation dose is analogous to that for chemical exposures, except that averaging time and body weight are omitted. These terms are effectively incorporated within the radionuclide cancer slope factors and dose conversion factors used to evaluate radiation dose (see Sections 5.4 and 5.5).

Pathway-specific intake equations and associated exposure parameter values are described for each exposure pathway in the following subsections. Instead of chemical mass, radionuclide activity (e.g., pCi) is used to quantify the amount of a radionuclide in an environmental medium. The general intake equation for radiation dose or cancer risk that serves as the basis for the pathway-specific equations is:

$$\text{Radionuclide Intake} = C_i \times CR \times EF \times ED$$

where Intake = rate of radionuclide available for uptake at an exchange boundary (pCi)
 C_i = concentration of radionuclide i at exposure point (e.g., pCi/g soil, pCi/L water)
 CR = contact rate with the environmental medium (e.g., mg soil ingestion /day; L water ingestion per day)
 EF = exposure frequency (days/yr)
 ED = exposure duration (year).

In practice, radionuclide dose is generally compared with dose thresholds that are specified on an annual basis. Therefore, the exposure duration term is generally omitted from the radionuclide equation when it is applied to radiation dose, resulting in intake expressed in units of pCi/yr.

5.4.2 Soil or Sediment Ingestion

Chemical intake via soil ingestion is calculated using the following equation. This equation must be modified when calculating intake for carcinogenic chemicals or radionuclide cancer risk by summing child and adult body-weight averaged intakes, expressed as $[(IR_s \times EF \times ED) / BW]$. For radionuclide intake, the equation would be further modified to exclude body weight and averaging time, as indicated in Section 5.4.1.

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$$\text{Intake (mg/kg} \cdot \text{d)} = \frac{C_{s,i} \times B_s \times IR_{s,d} \times EF \times ED \times 10^{-6} \text{ kg / mg}}{BW \times AT}$$

where $C_{s,i}$ = concentration of contaminant i in exposure area soil (mg/kg soil)
 B_s = relative bioavailability from soil in the gastrointestinal tract (unitless)
 $IR_{s,d}$ = daily soil ingestion rate (mg of soil/day)
 EF = exposure frequency (day/yr)
 ED = exposure duration (year)
 BW = body weight (kg)
 AT = averaging time (day).

The daily soil ingestion rate (IR_d), which is generally published as a daily rate, may be modified on a scenario-specific basis to account for the fraction of waking hours assumed to be spent within the exposure area. This site-specific soil ingestion rate (IR_s) is calculated as:

$$IR_s = IR_{s,d} \times [T_{\text{site}} / (24 \text{ hr} - T_{\text{sleep}})]$$

where IR_s = site-specific soil ingestion rate (mg of soil/day)
 T_{site} = daily time spent in exposure area (hr)
 T_{sleep} = daily time spent sleeping (hr)

and the term $[T_{\text{site}} / (24 \text{ hr} - T_{\text{sleep}})]$ is constrained to a value of 1.0 or less.

For scenarios that integrate exposure across multiple exposure areas within a single operational area (e.g., Resident Monument Worker), intake is apportioned on a time-dependent basis depending on the fraction of time spent in different exposure areas.

RME values for each exposure parameter, for each combination of land-use scenario and receptor, are provided in Table 5-8. Values for the CTE calculations are provided in Table 5-9. The basis of the parameter values related to soil and sediment ingestion are discussed below.

Relative Bioavailability from Soil (B_s). The relative bioavailability of a chemical represents the fraction of ingested chemical on soil that is absorbed in the gastrointestinal tract relative to the fraction that is absorbed from food or water. This term is introduced to the intake calculation for soil and sediment ingestion because bioavailability of many chemicals on soil tends to be less than that of the chemical when it is administered in food or water, which is generally how a chemical is administered in toxicological studies. In the absence of site-specific data related to bioavailability, the bioavailability of all chemicals in soil has been protectively assumed in the risk assessment to be equivalent to that in the administered dose.

Exposure Frequency (EF). An exposure frequency of 350 days/yr is used for the Rural-Residential scenario and the residential component of the Resident Monument Worker scenario, based on EPA guidance (EPA 1991c). In both cases assuming 2 weeks of vacation or other time spent at a separate location. The exposure frequency of 250 days/yr for the Industrial/Commercial Worker scenario and the occupational component of the Resident Monument

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Worker scenario is derived from EPA guidance (EPA 2002b) and reflects a 5-day work week for 50 weeks/yr. These values are used for both RME and CTE calculations. An exposure frequency of 365 days/yr is used for the CTUIR scenario (Harris and Harper 2004).

Exposure frequency values for the Recreational scenarios are inherently subjective. For both upland and riparian hunting, which are seasonal activities, proposed RME and CTE values are 30 and 10 days, respectively. These values will be apportioned equally to upland and riparian areas. Bank fishing for salmon, steelhead, and other species may occur over a greater length of time than hunting. Proposed RME and CTE values for the Avid Angler are 60 and 30 days/yr, respectively. For the Casual Use recreational scenario, the same seasonal exposure frequencies described for upland and riparian hunting are used. Identical recreational exposure frequencies are applied for adult and child receptors.

Exposure Duration (ED). A value of 6 years is used as the exposure duration for children in both the RME and CTE calculations for all scenarios that include children (EPA 1991c). An RME total exposure duration of 30 years is used for the Rural-Residential and Recreational scenarios (EPA/600/P-95/002Fc, Table 15-176), which represents the 95th percentile of population mobility from the 1993 U.S. Census. The CTE total exposure duration of 9 years for these scenarios is recommended in EPA exposure assessment guidance (EPA 1997c, Table 15-176) and represents the average of population mobility from the 1993 U.S. Census. An occupational exposure duration of 25 years for the Industrial/Commercial Worker and Resident Monument Worker scenarios is used for the RME calculations (EPA 1991c). For the CTE calculation, the median occupational tenure of the U.S. working population (6.6 years) is used. An exposure duration of 70 years is used for the CTUIR scenario (Harris and Harper 2004).

For the intake calculations for carcinogens, where exposure is summed across adult and child exposure periods, total exposure durations are based on combined childhood and adult exposure in order that the total exposure period remain consistent. For example, in the Rural-Residential and Recreational scenarios, 24 years and 3 years are used for adult exposure duration in the RME and CTE calculations, respectively.

Body Weight (BW). The body weight of children (16.6 kg) is based on the mean body weight of male and female children, ages 1 to 6 (EPA/600/P-95/002Fa, Table 7-3). For adults, a mean body weight of 70 kg is used (EPA 1991c). For the Avid Angler and Hunter scenarios, exposure of children is described in the CSM (see Section 2.0). A mean body weight of 34.6 kg is used based on the mean weight for children ages 7 to 12 years (EPA/600/P-95/002Fa, Table 7-3).

Averaging Time. The averaging time for noncarcinogenic effects is set equal to the exposure duration (EPA 1989). For carcinogenic effects, where effects are averaged over a lifetime, the averaging time is equal to human lifespan. An average human lifespan of 70 years for the general population is used (EPA 1991c).

Daily Soil Ingestion Rate (IR_{s,d}). Daily soil ingestion rates are based on EPA recommendations for risk assessment. A value of 200 mg/day used for the RME Rural-Residential and Recreational calculations for children is a “conservative estimate of the mean” (EPA/600/P-95/002Fa, Table 4-23). The 100 mg/day child value used in the CTE calculations

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instead represents a mean value (EPA/600/P-95/002Fa, Table 4-23). An adult value of 50 mg/day, which represents a mean value, is used for the CTE calculation for all adult receptors except in the CTUIR scenario (EPA/600/P-95/002Fa, Table 4-23). Because no other estimate is provided to represent upper-bound or protective estimates, an earlier EPA recommendation of 100 mg/day is used for the RME calculation (EPA 1991c). A daily soil ingestion rate of 400 mg/day is used for both child and adult receptors for the CTUIR scenario (Harris and Harper 2004).

Because the actual fraction of daily soil ingestion from any location is plausibly also a function of the amount of time spent there, the daily values are modified based on time spent in the exposure area.

Time On-Site (T_{site}). Rural Resident: 24 hours for a child receptor is used for both RME and CTE calculations. An adult value of 21 hours, the recommended value for time spent indoors in all locations for an adult over age 12, is used for both RME and CTE calculations (EPA/600/P-95/002Fc, Section 15.4.1). CTUIR: An on-site exposure period of 24 hours is used for receptors in this scenario. Although individuals may engage in activities away from the primary residence, an RME exposure is more likely to involve continuous exposure at a residence located near an upland waste site because local soil contaminant concentrations are likely highest in these areas. Industrial/Commercial and Resident Monument Worker: An 8-hour work day is assumed for chronic exposure for both the RME and CTE calculations. Recreational: Avid Hunter – A value of 8 hours is proposed for both RME and CTE calculations. Avid Angler – RME and CTE of 8 hours and 5 hours, respectively (PNNL-13840, Table 4.10). Casual User – RME and CTE of 6 hours and 3 hours, respectively, based on surveys of swimmers (PNNL-13840, Table 4.9).

Time Spent Sleeping (T_{sleep}). The daily time spent sleeping for children (11.5 hours) is the age-weighted 50th percentile of time spent sleeping or napping for a child age 1 to 6 years (EPA/600/P-95/002Fc, Table 15-83). For adults, the daily time spent sleeping (8 hours) is the 50th percentile of time spent sleeping or napping for an adult age 5–64 years (EPA/600/P-95/002Fc, Table 15-83). These values are used for both RME and CTE calculations.

5.4.3 Inhalation

Contaminant intake via inhalation is calculated using the following equation. This equation must be modified when calculating intake for carcinogenic chemicals by summing child and adult body-weight averaged intakes, expressed as $[(InhR \times ET \times EF \times ED) / BW]$. For radionuclide intake, the equation would be modified as indicated in Section 5.4.1.

$$\text{Intake (mg/kg} \cdot \text{d)} = \frac{C_{a,i} \times InhR \times ET \times EF \times ED}{BW \times AT}$$

where $C_{a,i}$ = concentration of contaminant i in exposure area air (mg/m³) (see Section 5.3.5)

InhR = site-specific inhalation rate (m³/hr)

ET = exposure time on site (hr/day)

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EF = exposure frequency (day/yr)
 ED = exposure duration (year)
 BW = body weight (kg)
 AT = averaging time (day).

For scenarios that have a residential component related to a waste site (e.g., Rural-Residential, CTUIR and Resident Monument Worker), values of $C_{a,i}$ calculated using chemical concentrations representing the operational area will also be calculated. If risks are higher using the broad area values of $C_{a,i}$, these inhalation pathway results will be used for these scenarios. For all other exposure scenarios, values of $C_{a,i}$ are always calculated using chemical concentrations representing the operational area.

RME values for each exposure parameter, for each combination of land use scenario and receptor, are provided in Table 5-8. Values for the CTE calculations are provided in Table 5-9. The basis of the parameter values related to dust inhalation are discussed below.

Inhalation Rate (InhR). A value of $0.63 \text{ m}^3/\text{hr}$ is used for adults in the Rural-Residential scenario for both RME and CTE calculations and for the CTE calculation in the Industrial scenario. It is the hourly equivalent of the recommended mean value for adult men (EPA/600/P-95/002Fc, Table 5-23). A value $1.0 \text{ m}^3/\text{hr}$, the recommended short-term exposure value for light activity (EPA/600/P-95/002Fc, Table 5-23), is used for the RME calculation for the Industrial/Commercial Worker scenario. The adult Recreational scenario values of 1.0 and $1.6 \text{ m}^3/\text{hr}$ shown in Tables 5-8 and 5-9 are the recommended short-term exposure values for light and moderate activities, respectively (EPA/600/P-95/002Fc, Table 5-23). An adult daily inhalation rate of 30 m^3 , corresponding to an hourly rate of $1.25 \text{ m}^3/\text{hr}$, is used for all applications for adult receptors in the CTUIR scenario (Harris and Harper 2004).

The RME and CTE value of $0.34 \text{ m}^3/\text{hr}$ for children in the Rural-Residential and CTUIR scenarios was calculated according to age-weighted rates provided in Table 7-14 (EPA-600-P00-002B) for children age 1 to 6 ($2/6 \times 6.8 \text{ m}^3/\text{day} + 3/6 \times 8.3 \text{ m}^3/\text{day} + 1/6 \times 10 \text{ m}^3/\text{day}$). The child Recreational scenario values of 1.0 and $1.2 \text{ m}^3/\text{hr}$ shown in Tables 5-7 and 5-8 are the recommended short-term exposure values for light and moderate activities, respectively (EPA-600-P00-002B, Table 7-14). These short-term values pertain to children ages 18 years and under. They are therefore likely to provide a protective bias when applied to children ages 1 to 6 or 7 to 12 years of age.

Exposure Time (ET). The value for exposure time on site is equivalent to the value of T_{site} , but expressed with units of daily rate (hr/day). For the sweat lodge exposure pathway, an event duration of 1 hour is used (Harris and Harper 2004).

Values for exposure frequency, exposure duration, body weight, and averaging time are identical to those described for the soil ingestion exposure pathway in Section 5.4.2. For inhalation in the sweat lodge, an exposure frequency (EF_{sweat}) of 365 days/yr is used (Harris and Harper 2004).

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5.4.4 Dermal Absorption from Soil and Sediment

Contaminant intake from soil via dermal contact is calculated using the following equation. This equation must be modified when calculating intake for carcinogenic chemicals by summing child and adult body-weight averaged intakes, expressed as $[(DA_{\text{event}} \times SA \times EF \times ED) / BW]$.

$$DAD = \frac{DA_{\text{event},i} \times SA_s \times EF \times ED}{BW \times AT}$$

where DAD = dermally absorbed dose (mg/kg-day)
 $DA_{\text{event},i}$ = absorbed dose per event, contaminant i (mg/cm²-event)
 SA_s = skin surface area exposed to soil (cm²)
 AF = soil adherence factor (mg/event)
 EF = exposure frequency for soil contact (event/yr)
 ED = exposure duration (year)
 BW = body weight (kg)
 AT = averaging time (day)

and

$$DA_{\text{event},i} = C_{s,i} \times ABS_{d,i} \times AF \times CF$$

where $DA_{\text{event},i}$ = absorbed dose per event, contaminant i (mg/cm²-event)
 $C_{s,i}$ = concentration of chemical i in soil (mg/kg)
 $ABS_{d,i}$ = dermal absorption fraction, contaminant i (unitless)
 AF = soil adherence factor (mg/event)
 CF = conversion factor (10⁻⁶ kg/mg).

RME values for each exposure parameter, for each combination of land use scenario and receptor, are provided in Table 5-8. Values for the CTE calculations are provided in Table 5-9. The basis of the parameter values related to soil and sediment ingestion are discussed below.

Dermal Absorption Fraction (ABS_d). The dermal absorption fraction characterizes the fraction of contaminant in soil that is absorbed into the skin during an exposure event. Values of ABS were obtained from Exhibit 3-4 of EPA's *Supplemental Guidance for Dermal Risk Assessment* (EPA/540/R/99/005). In Section 3.2.2.4 of this guidance, EPA recommends that absorption of metals from soil not be quantified with generic ABS_d values if a metal-specific value is unavailable. Therefore, dermal absorption from soil is only quantified for those metals for which EPA provides a value in Exhibit 3-4. Also in accord with EPA guidance (EPA/540/R/99/005), dermal absorption of VOCs from soil is not quantified (but absorption of semivolatile compounds is evaluated). Radiation dose and risk via dermal absorption will not be quantified as it is likely to be negligible compared with other exposure pathways of radiation exposure (EPA 1989, Section 10.5.5). Values of ABS_d for specific metals and organic chemicals from Exhibit 3-4 of EPA/540/R/99/005 are summarized in Table 5-11.

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Skin Surface Area Exposed to Soil (SA_s). The adult value (5,700 cm²) used for the Rural-Residential, CTUIR, Avid Angler, and Casual Use scenarios is the recommended value in EPA/540/R/99/005. The adult exposed skin surface area corresponds to skin revealed when dressed in short sleeves, short pants, and shoes. The child value (2,800 cm²) used for the Rural-Residential, CTUIR, and Casual Use scenarios is also the recommended value from EPA/540/R/99/005. The child exposed skin surface area corresponds to a <1 to <6 year old child with skin revealed when dressed in short sleeves and short pants, and no shoes. The adult value of 3,300 cm² used for the occupational portion of the Resident Monument Worker, the Industrial/Commercial Worker, and the Avid Hunter scenarios is the recommended value in EPA/540/R/99/005; Section 3.2) and corresponds to the head, hands, and forearms. These values are used for both RME and CTE calculations.

For the older child associated with the Avid Hunter and Avid Angler scenarios, surface area values were obtained from Exhibit C-1 of EPA/540/R/99/005. For the Avid Hunter scenario, the older child (7 to 12 years) was assumed to be exposed over the face, hands, and forearms in the same manner as an adult. The mean values for these body parts for age groups 7<8 through 12<13 were summed to obtain a surface area value of 1,500 cm². For the older child in the Avid Angler scenario, the areas of the lower legs and feet were added to obtain a surface area value of 3,400 cm².

Soil Adherence Factor (AF). The soil adherence factor defines the mass of soil that adheres on a unit area of skin. Adherence factors are affected by soil qualities (such as particle size and moisture content) and body part. The RME adult adherence factor for Rural-Residential and Recreational scenarios (0.07 mg/cm²-event) is the high-end recommended value in EPA/540/R/99/005, which is based on the 50th percentile for a high-exposure activity (gardening). A somewhat higher value of 0.1 mg/cm²-event is proposed for the CTUIR scenario, consistent with measurements for farmers (EPA/540/R/99/005, Exhibit 3-3). The adult CTE value (0.04 mg/cm²-event) for these scenarios is the recommended value in EPA/540/R/99/005; Exhibit 3-3) for the 50th percentile exposure intensity that reflects the geometric mean for landscapers. While the adherence factor related to gardening is more applicable to a Rural-Residential scenario, the lower value reflects a time-averaged chronic exposure period that includes days when gardening or other outdoor activities are not engaged in. For the Industrial/Commercial Worker scenario, which envisions a more sedentary (indoor) exposure, RME (0.04 mg/cm²-event) and CTE (0.01 mg/cm²-event) values are proposed (EPA/540/R/99/005, Exhibit 3-3). All values pertain to soil adherence on the face, hands, forearms, and lower legs.

The RME child adherence factor for all scenarios (0.2 mg/cm²-event) is the recommended value for high-end exposure activities (EPA/540/R/99/005). The value pertains to both the 95th percentile for day-care children (average activity) and the 50th percentile for children playing in wet soil (high-exposure activity). The child CTE value (0.04 mg/cm²-event) is the value in EPA/540/R/99/005 (Exhibit 3-3) for exposure intensity that reflects the geometric mean for children ages 1 to 12 playing in dry soil or in an indoor/outdoor daycare environment. All values pertain to soil adherence on the face, hands, forearms, lower legs, and feet.

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Exposure Frequency for Soil Dermal Contact (EF). Although the units of exposure frequency differ for soil dermal contact (events/yr) compared to the definition of exposure frequency in Section 5.4.2 (days/yr), the same exposure frequency values described in Section 5.4.2 may be applied to the soil dermal contact exposure pathway.

Values for exposure duration, body weight, and averaging time are identical to those described for the soil ingestion exposure pathway in Section 5.4.2.

5.4.5 Dermal Absorption from Water; Residential Uses

The methodology for assessing dermal uptake of metals and organic chemicals from water will follow the approach described in EPA's *Supplemental Guidance for Dermal Risk Assessment* (EPA/540/R/99/005). The absorbed dose per dermal exposure event (DA_{event}) for metals in water is a function of the soil concentration, exposure time, and chemical-specific dermal permeability coefficient. Calculation of DA_{event} for organic chemicals in water is more complex because absorption is also a function of the absorption and desquamation kinetics in the skin. One of two equations is used for this calculation, depending on the length of time required to achieve a steady-state skin concentration relative to the length of an exposure event. If the exposure event time (T_{event}) is shorter than the time to reach steady state (i.e., the length of time required for a chemical to be absorbed into the viable epidermis; t^*), the nonsteady-state model is used (EPA/540/R/99/005). If T_{event} is shorter than t^* , the pseudo-steady-state model is used (EPA/540/R/99/005).

Contaminant uptake via dermal absorption from water is calculated using the following equation. This equation is modified when calculating dermal uptake for carcinogenic chemicals by summing child and adult body-weight averaged exposures, expressed as $[(SA_w \times EV_f \times EF \times ED) / BW]$.

$$DAD = \frac{DA_{event,i} \times SA_w \times EV_{f,bath} \times EF \times ED}{BW \times AT}$$

where DAD = dermally absorbed dose (mg/kg-day)
 $DA_{event,i}$ = absorbed dose per event, contaminant i (mg/cm^2 -event)
 SA_w = skin surface area exposed to water (cm^2)
 $EV_{f,bath}$ = event frequency for bathing (events/day)
 EF = exposure frequency (days/yr)
 ED = exposure duration (years)
 BW = body weight (kg)
 AT = averaging time (days).

The following steady-state equation is used to estimate DA_{event} for metals:

$$DA_{event,i} = K_p \times C_{w,i} \times t_{event,bath}$$

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where $DA_{event,i}$ = absorbed dose per event, contaminant i ($\text{mg}/\text{cm}^2\text{-event}$)

$t_{event,bath}$ = duration of bathing event (hr/event)

K_p = permeability coefficient from water through skin (cm/hr)

$C_{w,i}$ = concentration of chemical i in water (mg/L).

For organic chemicals, the following equations are used to calculate DA_{event} :

$$\text{If } t_{event,bath} < t^*, \text{ then } DA_{event} = 2 \times FA \times K_p \times C_{w,i} \times CF \times \sqrt{\frac{6 \times \tau \times t_{event}}{\pi}}$$

$$\text{If } t_{event,bath} > t^*, \text{ then } DA_{event} = FA \times K_p \times C_{w,i} \times CF \times \left[\frac{t_{event}}{1+B} + 2 \times \tau \times \left(\frac{1+3B+3B^2}{(1+B)^2} \right) \right]$$

where t^* = time it takes to reach steady-state conditions in skin (hr)

FA = fraction of chemical dissolved in skin that is absorbed (unitless)

K_p = permeability coefficient from water through skin (cm/hr)

$C_{w,i}$ = concentration of chemical i in water (mg/L)

CF = conversion factor ($0.001 \text{ L}/\text{cm}^3$)

τ = lag time (hr/event)

π = pi; constant (unitless)

B = ratio of the permeability coefficient through the stratum corneum to the permeability coefficient across the viable epidermis (unitless).

Values for the chemical-specific parameters (t^* , FA , K_p , τ , and B) are obtained from Appendix B of EPA/540/R/99/005 and are reproduced in Table 5-12. Consistent with EPA guidance (EPA 1989, Section 10.5.5), radiation dose and risk via dermal absorption will not be quantified as it is likely to be negligible compared with other exposure pathways of radiation exposure.

As discussed in Appendix A of EPA/540/R/99/005, values of K_p are probably the most uncertain of the parameters in the dermal dose equation, with measured values having an uncertainty of plus or minus a half order of magnitude. Therefore, the final dose and risk estimates for dermal absorption are highly uncertain. In order to focus the dermal absorption pathway on chemicals that are potentially significant contributors to exposure via this route, the dermal absorption pathway for domestic water uses is first subjected to a screening evaluation as described in Section A.4 of EPA/540/R/99/005. The objective of the screening evaluation is to identify chemicals that may contribute via dermal absorption during bathing more than 10% of the drinking water intake. Using residential exposure assumptions for drinking water ingestion rate and bathing duration, organic chemicals are screened according to Equation A.16 of EPA/540/R/99/005:

$$\text{Dermal/Ingestion} > 10\% \text{ when, } FA \times K_p \times \sqrt{t_{event}} > 0.005$$

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Metals are screened according to Equation A.17 of EPA/540/R/99/005:

$$\text{Dermal/Ingestion} > 10\% \text{ when, } K_p > \text{ABS}_{\text{GI}}$$

Organic and inorganic chemicals which are not identified as potentially contributing greater than 10% of the ingestion dose via dermal absorption during bathing will not be evaluated for this exposure route in the Rural-Residential and CTUIR exposure scenarios.

Skin Surface Area Exposed to Water (SA_w). The adult value (18,000 cm²) used for the Rural-Residential (bathing) and CTUIR (bathing) scenarios is the recommended value in Exhibit 3-2 of EPA/540/R/99/005. The child value (6,600 cm²) used for the Rural-Residential and CTUIR scenarios is also the recommended value from EPA/540/R/99/005. The exposed skin surface areas corresponds to the total body surface area.

Event Frequency for Bathing (EV_{f,bath}). An event frequency for bathing or showering of one event per day is used for both RME and CTE calculations for the Rural-Residential and CTUIR scenarios (EPA/540/R/99/005).

Duration of Bathing Exposure Event (t_{event,bath}). Bathing event duration values for adults and children were obtained from Exhibit 3-2 of EPA/540/R/99/005. RME and CTE estimates for adults are 0.58 hr/event and 0.25 hr/event, respectively. For children, RME and CTE estimates are 1.0 hr/event and 0.33 hr/event, respectively.

Scenario-specific values for exposure frequency, exposure duration, body weight, and averaging time for those scenarios having dermal exposure to water are identical to those described for the soil ingestion exposure pathway in Section 5.4.2.

5.4.6 Dermal Absorption from Water; Sweat Lodge

Appendix 4 of Harris and Harper (2004) provides an exposure assessment methodology for calculating dermal absorption of chemicals present in water used during a sweat lodge. Contaminants are assumed to be introduced into the sweat lodge predominantly through the water poured over heated rocks that are used to create steam. Volatile and semivolatile chemicals are assumed to be 100% volatilized and to contribute to dermal uptake solely through air-phase contact with the skin (Harris and Harper 2004). By contrast, nonvolatile chemicals that become airborne as aerosols during vaporization of water are assumed to deposit on skin with condensing water. Total exposure to nonvolatile chemicals is expressed as the sum of contribution from air-phase and water-phase chemical concentrations. Metals will be treated as nonvolatile chemicals and all organic chemicals will be assumed to be either volatile or semivolatile for these calculations. Consistent with EPA guidance (EPA 1989, Section 10.5.5), radiation dose and risk via dermal absorption will not be quantified as it is likely to be negligible compared with other exposure pathways of radiation exposure.

For all contaminants in water, Appendix 4 of Harris and Harper (2004) indicates that the equation of absorbed dose per event (DA_{event}) described in Section 5.4.5 be modified in the

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following manner for calculating dermal absorption related to air-phase concentrations in the sweat lodge:

$$DA_{event} = C_{a,i} \times K_p \times t_{event,sweat} \times CF$$

where DA_{event} = absorbed dose per event (mg/cm^2 -event)
 $C_{a,i}$ = concentration of chemical i in sweat lodge air (mg/m^3) (see Section 5.3.9)
 K_p = permeability coefficient from water through skin (cm/hr)
 $t_{event,sweat}$ = duration of sweat event (hr/event)
 CF = conversion factor ($10^{-6} \text{ m}^3/\text{cm}^3$).

For nonvolatile chemicals (but not volatile and semivolatile chemicals), a second DA_{event} term is calculated to account for deposition of a chemical on skin with condensing water. In this case, the concentration of chemical in condensed water on the skin is presumed to be equal to that in the water used to create steam in the lodge (Harris and Harper 2004). The same equation described in EPA/540/R/99/005 for calculating DA_{event} for metals is recommended for nonvolatile chemical absorption from water in Harris and Harper (2004):

$$DA_{event} = K_p \times C_{w,i} \times t_{event,sweat} \times CF$$

where DA_{event} = absorbed dose per event (mg/cm^2 -event)
 $C_{w,i}$ = concentration of chemical i in sweat lodge water (mg/L)
 K_p = permeability coefficient from water through skin (cm/hr)
 $t_{event,sweat}$ = duration of sweat event (hr/event)
 CF = conversion factor ($10^{-3} \text{ L}/\text{cm}^3$).

For nonvolatile chemicals, total DA_{event} is thus the sum of two DA_{event} terms, one for vapor-phase and one for water-phase exposure.

DA_{event} is introduced into the equation for dermally absorbed dose, as defined in Section 5.4.5. The values of EF_{sweat} and ED_{sweat} , 365 d/yr and 68 yr, respectively, are taken from Appendix 4 of Harris and Harper (2004).

$$DAD = \frac{DA_{event} \times SA_w \times EF_{sweat} \times ED_{sweat}}{BW \times AT}$$

5.4.7 External Irradiation from Soil or Sediment

The general radionuclide intake equation described in Section 5.4.1 may also be used to characterize external radiation dose and risk from soil and sediment exposure. However, "contact" is simply a function of exposure time, and the equation reduces to:

$$\text{External Radiation Exposure} = C_{s,i} \times ((ET_{in} \times GSF) + ET_{out}) \times EF \times ED$$

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where Exposure = rate of external exposure (pCi-hr/g soil);
 $C_{s,i}$ = concentration of radionuclide i in exposure area soil (pCi/g)
 ET_{in} = indoor exposure time on site (hr/day)
 ET_{out} = outdoor exposure time on site (hr/day)
 GSF = gamma shielding factor for indoor exposure
 EF = exposure frequency (days/yr)
 ED = exposure duration (year)

Time Spent Outdoors (ET_{out}). For the Rural-Residential and CTUIR scenarios, the RME child value of 3 hours is approximately the 75th percentile of time spent at home in the yard for a child age 1 to 6 years (EPA/600/P-95/002Fb, Table 15-120). The adult RME residential value of 3 hours is also approximately the 75th percentile of time spent at home in the yard for adult age categories. The child CTE residential value of 1.7 hours is the age-weighted 50th percentile of time spent at home in the yard for a child age 1 to 6 years (EPA/600/P-95/002Fb, Table 15-120). Likewise, the adult CTE value of 1.5 hours is also approximately the 50th percentile of time spent at home in the yard for adult age categories. The amount of time spent outdoors is equal to the total time spent in the exposure area (see Section 5.4.2) for the remaining exposure scenarios.

Time Spent Indoors (ET_{in}). The amount of time spent indoors is calculated as the total time spent in the exposure area minus the time spent outdoors ($T_{site} - ET_{out}$).

Gamma Shielding Factor (GSF). The gamma shielding factor accounts for attenuation of external irradiation in the indoor environment from the shielding effects of the residence. The value of the gamma shielding factor may be expected to vary as a function of building construction methods, the geometry of the source term, and the nuclide-specific energy of the gamma emission. A value of 0.4 for the gamma shielding factor is employed based on EPA recommendation for developing soil screening guidelines (EPA/540-R-00-007, Equation 4).

If radiation dose is expressed on an annual basis, as is the case in this risk assessment, the term ED is omitted from both the internal and external exposure equations. Values for exposure frequency and ED are identical to those described in Sections 5.4.2 through 5.4.4.

5.4.8 Ingestion of Water

Chemical intake via water ingestion is calculated using the following equation. This equation must be modified when calculating intake for carcinogenic chemicals by summing child and adult body-weight averaged intakes, expressed as $[(IR_w \times EF \times ED) / BW]$. For radionuclide intake, the equation would be modified as indicated in Section 5.4.1.

$$\text{Intake (mg/kg} \cdot \text{d)} = \frac{C_{w,i} \times IR_{gw} \times EF \times ED}{BW \times AT}$$

where $C_{w,i}$ = concentration of contaminant i in exposure area water (mg/L)
 IR_{gw} = groundwater ingestion rate (L/day)
 EF = exposure frequency (day/yr)

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ED = exposure duration (year)

BW = body weight (kg)

AT = averaging time (day).

Ingestion Rate of Groundwater (IR_{gw}). Drinking water intake rates were obtained from a summary of recommended values (EPA/600/P-95/002Fc, Table 3-30). The RME value of 2.3 L/day for adults is the 90th percentile value drinking water intake rate of the general population. This value is not representative of more active subpopulations of adults, who may consume two to five times more water on a daily basis depending on climate and activity. However, such rates may be excessive when applied over a chronic residential exposure period. The CTE value of 1.4 L/day for adults is the mean value for the general population. The RME value of 0.9 L/day for children (ages 1 to 10 years) is the 90th percentile value for ingestion of community (i.e., tap water) drinking water (EPA-600-P00-002B, Table 4-12). The CTE child value of 0.30 L/day is the 50th percentile value for ingestion of community drinking water (EPA-600-P00-002B, Table 4-12). Adult water consumption of 3 L/day, plus an additional 1 L/day during use of the seat lodge, is applied for the CTUIR scenario (Harris and Harper 2004). A drinking water consumption rate of 1.5 L/day, corresponding to the 95th percentile of children ages 1 to 10 years, is used for children in the CTUIR scenario (EPA/600/P-95/002Fc, Table 3-30).

Values for exposure frequency, exposure duration, body weight, and averaging time are identical to those described for the soil ingestion exposure pathway in Section 5.4.2. For drinking water ingestion in the sweat lodge, an exposure frequency (EF_{sweat}) of 365 days/yr is used (Harris and Harper 2004).

5.4.9 Foodstuffs Ingestion

Contaminant intake from ingestion of home-raised foodstuffs is calculated using the following equation. This equation applies to both chemical carcinogens (adult and child exposure summed) and radionuclides and noncarcinogens (adult and child exposure evaluated separately) because the ingestion rate values for home-raised foodstuffs published by EPA (EPA/600/P-95/002Fb) are based on survey data across a general population that includes adults and children. For radionuclide intake, the equation would be modified as indicated in Section 5.4.1. To correct for body weight in the radionuclide calculation, a value of 60 kg is recommended by EPA (EPA/600/P-95/002Fb).

$$\text{Intake (mg/kg} \cdot \text{d)} = \frac{C_{\text{food},i} \times IR_{\text{food}} \times EF_{\text{food}} \times ED \times 10^{-3} \text{ kg / g}}{AT}$$

where $C_{\text{food},i}$ = concentration of contaminant i in foodstuff (mg/kg)
 IR_{food} = foodstuff ingestion rate (g/kg-d)
 EF_{food} = exposure frequency for ingesting food products (day/yr)
 ED = exposure duration (year)
 AT = averaging time (day).

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Intake rates for each of the foodstuffs evaluated in the risk assessment (fruits and vegetables, beef, milk, chicken, eggs, fish, and wild game) are described in the following paragraphs. EPA's *Exposure Factors Handbook* (EPA/600/P-95/002Fb) is the reference employed for all values except those associated with the CTUIR exposure scenario (Harris and Harper 2004). These values are specific to subpopulations who engage in the activities in question, including gardening, fishing, and the raising of livestock. In addition to *Exposure Factors Handbook, Volume II* (EPA/600/P-95/002Fb), the *Child-Specific Exposure Factors Handbook* (EPA-600-P00-002B) provides some information relating to a child's uptake of home-raised foods. For example, a summed value for total home-produced meat ingestion, including all domestic animals and game, is provided in EPA-600-P00-002B. Although EPA's *Child-Specific Exposure Factors Handbook* (EPA-600-P00-002B) provides recommendations for children alone, which are lacking in EPA/600/P-95/002Fb, ingestion rates of specific home-produced foods are not differentiated in EPA-600-P00-002B. The ability to estimate ingestion of home-produced foodstuffs is considered critical for many of the foodstuff exposure pathways.

Ingestion Rate of Vegetables (IR_{veg}). The ingestion rate information for home-grown produce for the Rural-Residential scenario was obtained from EPA/600/P-95/002Fb and has its basis in the Nationwide Food Consumption Survey conducted periodically by the U.S. Department of Agriculture. The RME value of 1.2 g/kg-day is based on the 75th percentile of seasonally adjusted consumer intake of home-grown vegetables for the western United States. Because the survey results reflect quantities of produce brought into the house rather than actual portions of food, the ingestion rate values incorporate a correction of 18% to reflect the average preparation loss for common garden vegetables including corn, pumpkin (squash), peppers, and tomatoes. The CTE value of 0.40 g/kg-day is based on the 50th percentile of seasonally adjusted consumer intake of home-grown vegetables for the western United States and also incorporates the 18% correction. The values of intake rates and food preparation losses were obtained from EPA/600/P-95/002Fb (Tables 13-33 and 13-7).

A 95th percentile value of exposure for the general population is commonly used in risk assessment to represent RME conditions. For the Rural-Residential scenario, a 75th percentile value was selected because ingestion of home-grown vegetables is evaluated in conjunction with ingestion of other home-grown foodstuffs (fruits, beef, and poultry and eggs). It is unlikely that a single resident would produce and consume 95th percentile quantities of home-grown foodstuffs across multiple food categories.

Ingestion rates for vegetables for the CTUIR scenario were obtained from Table 5 of Harris and Harper (2004). Ingestion rates were summed across the categories of roots, greens, and "other" for a value of 1,225 g/day. Assuming that the ingestion rates are applicable to an adult, a body weight of 70 kg was used to calculate a body weight-normalized value of 17.5 g/kg-day. No preparation loss was used to modify this value.

Ingestion Rate of Fruit (IR_{fruit}). The ingestion rate information for home-grown fruit for the Rural-Residential scenario is similar to that for home-grown vegetables. The RME value of 1.4 g/kg-day is based on the 75th percentile of seasonally adjusted consumer intake of home-grown fruit for the western United States. The ingestion rate values incorporate a correction of

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23% to reflect the average preparation loss for common orchard fruit including apples, pears, and peaches. The CTE value of 0.53 g/kg-day is based on the 50th percentile of seasonally adjusted consumer intake of home-grown fruit for the western United States and also incorporates a 23% correction. The values of intake rates and food preparation losses were obtained from EPA/600/P-95/002Fb (Tables 13-33 and 13-6). The basis for selecting the 75th percentile for the RME estimate is analogous to that described for home-grown vegetables.

The ingestion rate for fruits and berries for the CTUIR scenario (125 g/day) was obtained from Table 5 of Harris and Harper (2004). Assuming that the ingestion rate is applicable to an adult, a body weight of 70 kg was used to calculate a body weight-normalized value of 1.8 g/kg-day.

Ingestion Rate of Home-Raised Chicken and Eggs (IR_{chick} and IR_{egg}). The ingestion rate information for home-raised poultry used for the Rural-Residential scenario is similar to that for home-grown fruits and vegetables. For the RME calculation of chicken ingestion, the 75th percentile value (1.3 g/kg-d) of annual-average home-produced poultry intake for the western United States was used. Because the survey results reflect quantities of poultry brought into the house rather than actual portions of food, the ingestion rate values incorporate a correction of 31% to reflect the average preparation loss for chicken. For the RME calculation of egg ingestion, the 75th percentile value (1.05 g/kg-d) of annual-average home-produced egg intake for the western United States was used. For the CTE calculation, the 50th percentile value (0.70 g/kg-d) of annual-average home-produced poultry intake for the western United States was used. This value also incorporates a correction of 31% to reflect the average preparation loss for chicken. For the CTE calculation, the 50th percentile value (0.67 g/kg-d) of annual-average home-produced egg intake for the western United States was used. The values of chicken intake rates and food preparation loss were obtained EPA/600/P-95/002Fb (Tables 13-55 and 13-5). The egg ingestion rate was obtained from EPA/600/P-95/002Fb (Table 13-43).

Because ingestion of home-raised poultry and eggs is evaluated in conjunction with home-grown beef, fruits, and vegetables, a 75th percentile value is used in the RME calculations rather than a more-traditional 95th percentile. A lower percentile of the population was selected because it is unlikely that a single resident would produce and consume very high quantities of home-grown foodstuffs across multiple food categories.

The ingestion rate for fowl for the CTUIR scenario (62.5 g/day) is one-half of the combined rate for game and fowl (125 g/day) obtained from Table 5 of Harris and Harper (2004). Assuming that the ingestion rate is applicable to an adult, a body weight of 70 kg was used to calculate a body weight normalized value of 0.89 g/kg-day. This value is applied both to hunted game birds (waterfowl and upland birds) and domesticated poultry.

Ingestion Rate of Beef (IR_{beef}). The ingestion rate information for home-raised beef used in the Rural-Residential scenario is similar to that for other foodstuffs. For the RME calculation of beef ingestion, the 75th percentile value (2.2 g/kg-d) of annual-average home-produced beef intake for the western United States was used. For the CTE calculation, the 50th percentile value (1.2 g/kg-d) of annual-average home-produced beef intake for the western United States was used. Because the survey results reflect quantities of beef brought into the house rather than

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actual portions of food, the ingestion rate values incorporate a correction of 24% to reflect the average preparation loss for meat. The ingestion rates and food preparation loss were obtained from EPA/600/P-95/002Fb (Tables 13-36 and 13-5).

Ingestion of home-raised beef is evaluated in conjunction with ingestion of other home-grown foodstuffs in the Rural-Residential scenario. Therefore, a 75th percentile value is used in the RME calculations rather than a more traditional 95th percentile. A lower percentile of the population was selected because it is unlikely that a single resident would produce and consume very high quantities of home-grown foodstuffs across multiple food categories.

The ingestion rate for beef for the CTUIR scenario (62.5 g/day) is one-half of the combined rate for game and fowl (125 g/day) obtained from Table 5 of Harris and Harper (2004). Assuming that the ingestion rate is applicable to an adult, a body weight of 70 kg was used to calculate a body weight normalized value of 0.89 g/kg-day. This value is applied to both hunted game mammals (elk, deer, etc.) and domesticated cattle.

Ingestion Rate of Milk (IR_{milk}). The ingestion rate information for home-produced milk (dairy) used in the Rural-Residential scenario is similar to that for other foodstuffs. For the RME calculation of milk ingestion, the 75th percentile value (19.5 g/kg-d) of annual-average home-produced milk intake for all regions of the U.S. was used. A specific value for regional intake in the western United States was not provided in EPA/600/P-95/002Fb because of too few observations. For the CTE calculation, the 50th percentile value (10.2 g/kg-d) of annual-average home-produced milk intake for all regions of the United States was used. The ingestion rates were obtained from EPA/600/P-95/002Fb (Table 13-28). Ingestion of home-produced milk is evaluated in conjunction with ingestion of other home-grown foodstuffs in the Rural-Residential scenario. Therefore, a 75th percentile value is used in the RME calculations rather than a more-traditional 95th percentile. A lower percentile of the population was selected because it is unlikely that a single resident would produce and consume very high quantities of home-grown foodstuffs across multiple food categories.

Ingestion Rate of Fish (IR_{fish}). The ingestion rate information for home-caught fish used in the Rural-Residential scenario is similar to that for other foodstuffs. For the RME calculation of fish ingestion, the 75th percentile value (0.86 g/kg-d) of annual-average home-caught fish intake for the western United States was used. For the CTE calculation, the 50th percentile value (0.49 g/kg-d) of annual-average home-caught fish intake for the western United States was used. The 95th percentile value (3.3 g/kg-d) is used for Avid Angler recreational scenario. Because the survey results reflect quantities of fish brought into the house rather than actual portions of food, the ingestion rate values incorporate a correction of 11% to reflect the average preparation loss for fish. The ingestion rates and food preparation loss were obtained from EPA/600/P-95/002Fb (Tables 13-27 and 13-5).

Ingestion of home-caught fish is evaluated in conjunction with ingestion of other home-grown foodstuffs in the Rural-Residential scenario. Therefore, a 75th percentile value is used in the RME calculations rather than a more-traditional 95th percentile. A lower percentile of the

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population was selected because it is unlikely that a single resident would produce and consume very high quantities of home-grown foodstuffs across multiple food categories.

The ingestion rate for fish for the CTUIR scenario (620 g/day) was obtained from Table 5 of Harris and Harper (2004). Assuming that the ingestion rate is applicable to an adult, a body weight of 70 kg was used to calculate a body weight normalized value of 8.9 g/kg-day.

Ingestion Rate of Wild Game (IR_{game}). The ingestion rate information for home-produced game used in the Avid Hunting recreational scenario is similar to that for fish used for Avid Angler recreational scenario. For the RME calculation of Avid Hunting game ingestion, the 95th percentile value (2.2 g/kg-d) of annual-average home-produced game intake for the western United States was used. For the CTE calculation, the 50th percentile value (0.57 g/kg-d) of annual-average home-caught fish intake for the western United States was used. Because the survey results reflect quantities of beef brought into the house rather than actual portions of food, the ingestion rate values incorporate a correction of 24% to reflect the average preparation loss for meat. The ingestion rates and food preparation loss were obtained from EPA/600/P-95/002Fb (Tables 13-44 and 13-5).

The ingestion rate of game for the CTUIR scenario (125 g/day) is the combined rate for game and fowl obtained from Table 5 of Harris and Harper (2004). Assuming that the ingestion rate is applicable to an adult, a body weight of 70 kg was used to calculate a body weight normalized value of 1.8 g/kg-day.

Exposure Frequency for Home-Grown Food Ingestion (EF_{food}). The ingestion rate data for home-produced foods are provided as annual averages (EPA/600/P-95/002Fb). Therefore, an exposure frequency of 365 days/yr is used.

Values for exposure duration and averaging time are identical to those described for the soil ingestion exposure pathway in Section 5.4.2. Separate exposure duration values for children and adults are not used because the study cohort from which home-produced food ingestion rate values were derived consisted of children and adults. Also, because body weight is factored into the ingestion rate terms, a separate term for body weight is unnecessary. For the radionuclide risk and dose calculations, where body weight is not expressed at all in the intake equations, a body weight of 60 kg (EPA/600/P-95/002Fb, Section 13.3) is used to express food ingestion rates with units of g/kg-day.

5.5 TOXICITY ASSESSMENT

5.5.1 Overview

Potential health effects related to intake of chemical COPCs are assessed using dose-response information described by cancer slope factors (CSFs) for carcinogenic effects of chemicals, and reference doses (RfDs) for systemic (noncarcinogenic) effects of chemicals. These values describe a relationship between the intensity of exposure and the likelihood or severity of associated health effects. The EPA has evaluated available dose-response information for many

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chemicals and has published this information in the form of toxicity values and accompanying information.

The hierarchy of references for chemical toxicity criteria is described in a 2003 memorandum from EPA's Office of Solid Waste and Emergency Response (EPA 2003b). In accordance with this memorandum, the primary source of toxicity values used in the RCBRA is EPA's *Integrated Risk Information System* database (IRIS) (EPA 2007). Only toxicity criteria published in IRIS have gone through a peer review and EPA consensus review process. The second tier of toxicity criteria are the provisional peer-reviewed toxicity criteria (PPRTV) published by the National Center for Environmental Assessment (NCEA) in EPA's Office of Research and Development. These values are developed on a chemical specific basis when requested by EPA's Superfund program, but the documentation for them is generally not citable. The third tier of references include values published in EPA's *Health Effects Assessment Summary Tables* (HEAST; EPA-540-R-97-036), and other sources such as California EPA and the Agency for Toxic Substances and Disease Registry. Application of this hierarchy for identifying route-specific toxicity criteria is described in Section 5.5.7.

The potential health effects related to radionuclide exposure are also assessed in the RCBRA. Radiation dose is evaluated as the committed effective dose equivalent using radiation dose conversion factors (DCFs) published in *Federal Guidance Report No. 11* (EPA 520/1-88-020) and *Federal Guidance Report No. 12* (EPA-402-R-93-081). Radiation cancer risk is also assessed, using radionuclide cancer slope factors published in *Federal Guidance Report No. 13* (EPA 402/R-99/001) and available on-line from EPA's Office of Radiation and Indoor Air (EPA 2006).

5.5.2 Chemical Hazard

The toxicity value used to evaluate systemic health effects related to long-term exposures is the chronic RfD. The chronic RfD is an estimate of daily exposure likely to be without appreciable risk of adverse effects for exposure of several years or longer (EPA 1989).

The general model of toxicity for noncarcinogenic effects is that there exists a range of exposure from zero to some "threshold" in which exposure can be tolerated without a significant probability of an adverse effect. An RfD represents an estimate of this threshold and is expressed as a body weight-normalized rate of exposure with the same units as intake (e.g., mg/kg-day). This model of toxicity is reflected in the averaging time for noncarcinogenic effects, which is equivalent to the exposure duration. The effect is generally assumed to manifest only when exposure exceeds a threshold and not to occur when exposure is less than the threshold or at some time following the exposure.

An RfD is derived by EPA using human dose-response data from adequate studies, if available. If human data are unavailable, dose-response information from animal studies may be employed. EPA will preferentially base an RfD on the highest dose level not associated with adverse effects (the no-observable-adverse-effects-level, or NOAEL). If such a value was not identified in the literature, the lowest-observable-adverse-effects-level (LOAEL) is generally used as the basis of the RfD. In practice, EPA will generally first identify the critical study and adverse effect for a

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chemical from a review of the available toxicological data. Once these are specified the NOAEL or LOAEL is identified. The RfD is then calculated from the NOAEL or LOAEL using uncertainty factors (UFs) to account for uncertainty in extrapolating from the NOAEL or LOAEL to a chronic RfD. Uncertainty factors may relate to potential variability in sensitivity in the human population, to interspecies variability between humans and test animals, to inadequate dosing periods in a critical study, or to use of a LOAEL instead of a NOAEL. A modifying factor is sometimes also employed to account for additional uncertainties in the derivation of a chronic RfD.

5.5.3 Chemical Cancer Risk

The toxicity value used to evaluate chemical carcinogenic health effects is the CSF. A CSF is a quantitative relationship between dose and carcinogenic response and is usually representative of a plausible upper-bound estimate of the lifetime probability of developing cancer associated with exposure to a specific quantity of a potential carcinogen (EPA 1989). The units of a chemical CSF are expressed as cancer risk per intake, with units of $(\text{mg}/\text{kg}\text{-day})^{-1}$. EPA's CSFs are associated with a WOE classification that indicates the strength of the evidence by which the chemical is suspected to be a human carcinogen. These classifications include the following:

- A human carcinogen
- B1 probable human carcinogen (limited human data available indicating carcinogenicity)
- B2 probable human carcinogen (inadequate or no human data available)
- C possible human carcinogen
- D not classifiable as to human carcinogenicity
- E evidence for noncarcinogenicity in humans

Because class D and E carcinogens are not classified as human carcinogens, CSFs have not been developed for them; therefore, they are not included in a quantitative analysis of potential carcinogenicity.

The great majority of CSFs are based on carcinogenic effects observed at relatively high dose rates that have been extrapolated to lower doses. There are multiple mathematical models used for this extrapolation that relate both to the goodness-of-fit with the dose-response data, as well as theoretical models of carcinogenesis. The CSF is commonly calculated as the 95% UCL on the slope of the dose-response curve, although in some cases where the data are more robust, a "best estimate" is used instead.

One of the principal differences in assumptions regarding carcinogenic and noncarcinogenic effects pertains to the presumption of a threshold of exposure for noncarcinogenic effects. As described in Section 5.2, it is assumed for systemic effects that there exists a range of exposure from zero to some "threshold" in which exposure can be tolerated without a significant probability of an adverse effect. By contrast, EPA believes that the underlying mechanisms of carcinogenesis imply that there is no threshold of exposure (EPA 1989). That is, any exposure, no matter how small, provides some finite possibility of resulting in a carcinogenic effect. A CSF therefore represents the incremental risk of cancer incidence associated with some finite

exposure and is expressed as cancer risk per unit intake [risk/ (mg/ kg-day)], or (mg/ kg-day)⁻¹. Because there may be a decades-long latency period between exposure and effect (EPA/630/P-03/001F), effects are averaged over an entire lifetime.

In their 2005 *Guidelines for Carcinogen Risk Assessment* (EPA/630/P-03/001F), EPA's Risk Assessment Forum stressed the use of mode-of-action⁶ information for evaluating chemical carcinogenicity. A companion document, *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (EPA/630/R-03/003F), provided guidance for adjusting cancer potency estimates for childhood exposures to mutagenic carcinogens. Table 1b of EPA/630/R-03/003F lists 12 chemicals that have been identified as mutagens based on animal experiments where both early-life and adult exposures were conducted with separate groups of animals. Of particular interest relative to the COPCs identified in the RCBRA are the four PAHs listed in Table 1b (benzo(a)pyrene, dibenzanthracene, dimethylbenz(a)anthracene, and 3-methylcholanthrene).

The mutagenicity of these and certain other PAHs is related to the formation of electrophilic metabolites, leading to the creation of DNA adducts (Klaassen 2001). However, the carcinogenicity of these compounds may also be related to other modes of action, including immune system suppression and the tumor-promoting effect of increased mitogenicity (Klaassen 2001). The relative importance of these modes of action with chronic exposure is uncertain. With the exception of 3-methylcholanthrene, for which repeat doses were given, the experiments cited in EPA/630/R-03/003F (Table 1b) for the four PAHs involved acute exposures. Due to a number of concerns described in EPA/630/R-03/003F (Section 3.2.1), EPA does not recommend the use of acute dosing studies for quantitative adjustment of cancer potency values. In light of these issues, quantitative adjustment of the IRIS chronic CSF for benzo(a)pyrene for application to childhood exposures is not performed in the RCBRA. The possibility and magnitude of risk underestimation related to childhood exposures to mutagenic carcinogens will be discussed relative to other sources of uncertainty and bias in the uncertainty analysis.

5.5.4 Radionuclide Cancer Risk

The toxicity value used to evaluate radionuclide carcinogenic health effects is also referred to as a CSF. The radionuclide CSF is a quantitative relationship between radiation dose and carcinogenic response, but (unlike the chemical CSF) reflects an average estimate of the lifetime risk of cancer associated with exposure to a specific concentration of a carcinogen in an environmental medium (EPA 402/R-99/001). The units of a radionuclide CSF are expressed as cancer risk per annual intake of radionuclide activity, with units of risk per activity (pCi)⁻¹. For external irradiation, radionuclide CSFs define the relationship between annual cancer risk and the radionuclide activity in the source medium (risk/year per pCi/g).

Radionuclide slope factors published by EPA are preferred to the use of risk factors applied as multipliers to calculated radiation dose equivalents. Although such dose equivalents are

⁶ A sequence of key events and processes, starting with interaction of an agent with a cell, proceeding through operational and anatomical changes, and resulting in cancer formation (EPA/630/P-03/001F).

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applicable for comparison to dose-based radiation protection standards, they were derived for application to adults in a workplace setting. More recent radionuclide slope factors from *Federal Guidance Report No. 13* (EPA 402/R-99/001), by contrast, were derived to pertain to the general U.S. population and are therefore applicable for use in estimating cancer risks for a general population composed of adults and children. *Federal Guidance Report 13* slope factors are derived using age- and gender-specific values for intake and radionuclide dosimetry.

5.5.5 Radionuclide Dose

Radiation dose is not an “effect” per se, but rather a measure of the radiation dose absorbed by a tissue that accounts for the biological effectiveness of the radiation (e.g., alpha particles, photons) in causing cellular damage in different tissues. For external dose, the effective dose equivalent is calculated. For internal dose, the committed effective dose equivalent is used.

The effective dose equivalent is the weighted sum of the dose equivalents to different organs and tissues, where the weighting factors express an individual tissue dose as an equivalent dose to the whole body, with respect to the probability of developing a fatal cancer. The committed effective dose equivalent is a variation on the effective dose equivalent and is defined as the total effective dose equivalent deposited in the body in a 50-year period following the intake of a radionuclide (ICRP 1977).

The DCFs provided in *Federal Guidance Report No. 11* (EPA 520/1-88-020) and *Federal Guidance Report No. 12* (EPA-402-R-93-081) do not discriminate among various age groups of receptors in the manner of the radionuclide CSFs from *Federal Guidance Report No. 13* (EPA 402/R-99/001). However, *Federal Guidance Report 13* has a statement to the effect that it does not replace *Federal Guidance Report No. 11* and *Federal Guidance Report No. 12*. In the preface to *Federal Guidance Report No. 13*, it says that the DCFs in *Federal Guidance Report No. 11* and *Federal Guidance Report No. 12* “...continue to be recommended for determining conformance with the radiation protection guidance to Federal agencies issued by the President”. This suggests that these DCFs should continue to be used to assess compliance with dose-based standards.

5.5.6 Development of Dermal Toxicity Values

As discussed in EPA/540/R/99/005, dermal toxicity values would ideally incorporate assessment of direct toxicity in the skin and be based on dose-response data for systemic effects via percutaneous absorption. In the absence of such information, EPA/540/R/99/005 recommends the use of dose-response relationships obtained from oral administration studies with adjustment for gastrointestinal absorption efficiency so that the dermal toxicity values reflects absorbed rather than administered dose. Values for oral absorption (ABS_{oral}) were obtained from Exhibit 4-1 of EPA's *Dermal Risk Assessment Guidance for Superfund* (EPA/540/R/99/005). In addition, an ABS_{oral} of 60% is applied to lead, as discussed in Section 5.4.2. ABS_{oral} values are provided in Table 5-10.

As discussed in Section 5.4.2, for all organic chemicals listed in Exhibit 4-1, EPA recommends that oral toxicity value not be adjusted for application to dermal exposure due to limited

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bioavailability. For unlisted organic chemicals, EPA/540/R/99/005 (Section 4.2) also recommends that bioavailability in the gastrointestinal tract be assumed to be 100%. Therefore, the oral toxicity values are used without modification for assessing toxicity related to dermal absorption for all organic chemicals. The bioavailability values for specific metals are summarized in Table 5-10. For all other metals and radionuclides, an ABS_{oral} value of 100% is used.

For noncarcinogenic effects, the oral RfD is adjusted according to

$$RfD_{dermal} = RfD_{oral} \times ABS_{oral}$$

For carcinogenic effects, the oral CSF is adjusted according to

$$SF_{dermal} = SF_{oral} / ABS_{oral}$$

A potential problem in the use of oral toxicity values for dermal absorption involves chemicals whose critical toxicity may occur in the skin following absorption. These issues are addressed on a chemical-specific basis in the risk assessment.

5.5.7 Protocol for Identification of Route-Specific Toxicity Criteria

The toxicity criteria that EPA recommends for use in human health risk assessments correspond to one of three data quality categories. The categories reflect the confidence in the toxicity studies used to develop the toxicity criteria, as well as their preference for use in human health risk assessments. The categories, listed from most to least preferred, are as follows: Tier 1 (IRIS), Tier 2 (NCEA PPRTV), and Tier 3 (NCEA / HEAST). Tier 1 toxicity data are associated with the highest degree of confidence because of rigorous peer review. In general, a Tier 1 criterion for a particular chemical is not mixed with lower tier criteria for the same chemical. For example, a Tier 2 or Tier 3 criterion will not be used if a Tier 1 criterion is available for a chemical for the route/effect of concern, instead route-route (oral to inhalation or vice versa) extrapolation will be used.

Route-to-route extrapolation is the process of employing a toxicity value based on either oral or inhalation administration to characterize the potential toxicity for exposure via the other exposure route when toxicity data for that route are unavailable. Typically, this involves using toxicity values for oral administration to represent toxicity via inhalation.

A route-to-route extrapolation presumes that the critical toxicity of a chemical is systemic. In other words, that the toxic effect is remote from the contact sites of the gastrointestinal or inhalation systems. If this condition is violated, there is a potential for significant under- or overrepresentation of an agent's potential toxicity when extrapolating between exposure routes. Ideally, adequate information regarding a chemical's mode of action exists to understand potential differences in the relative absorption, metabolism, binding, and excretion rates between the oral and inhalation exposure routes. Therefore, route-to-route extrapolation for organic

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chemicals is suggested only when the toxic effects of interest occur at a site remote from the portal of entry and when absorption characteristics between the two routes is similar (EPA/540/R-95/128, EPA/630/P-03/001F).

Because absorption efficiencies of metals is likely to be quite different in the lungs and gastrointestinal tract, route-to-route extrapolation will not be conducted for inorganic chemicals. Practically, toxicological data for establishing absorption efficiency and portal of entry effects are not always available, and in many cases a protective route extrapolation will have minimal impacts on calculated risks when summed across all relevant exposure pathways. Therefore, extrapolation of oral RfDs and CSFs to the inhalation route will be performed in this assessment for organic chemicals when an inhalation toxicity criterion is lacking. If inhalation risk is determined to be significant for a chemical based on such an extrapolation, the chemical-specific validity of the extrapolation will be evaluated in the uncertainty analysis.

On a chemical-specific basis, if no toxicity criteria are available, a surrogate chemical may be identified based on similarity in the mechanism of toxicity or chemical structure.

The protocol used to select toxicity criteria is described below and presented diagrammatically in Figures 5-4 and 5-5.

If available, use a Tier 1 value. If not, use a Tier 1 route-route extrapolation for the same health effect endpoint (i.e., carcinogenic or noncarcinogenic effect). If a Tier 1 route-route extrapolation for the same effect is not possible, then determine if at least the route is covered by a Tier 1 value for a different effect (i.e., a Tier 1 oral CSF will cover the oral route in the absence of Tier 1 data for the oral RfD and vice versa). If a Tier 1 value for the route is available, then do not apply a lower tier value. If a Tier 1 value for the route is unavailable, then use a Tier 2 value. If a Tier 2 value is not available, use a Tier 3 value or a valid surrogate value. Preferably, the surrogate value selected is a Tier 1 value.

5.5.8 Toxicity Assessment for Lead

Lead exposure can result in significant health effects, particularly among children, whose physiology and behavior are generally believed to cause them to be more susceptible to the effects of lead in environmental media such as soil and dust. Health effects associated with exposure to inorganic forms of lead include neurotoxicity, developmental delays, hypertension, impaired hearing acuity, impaired hemoglobin synthesis, and male reproductive impairment (EPA 2007).

High-level exposure to lead produces encephalopathy in children, with signs of encephalopathy associated with blood lead concentrations of approximately 90 to 800 $\mu\text{g}/\text{dL}$ (mean, 330 $\mu\text{g}/\text{dL}$) (ATSDR 2005). The distribution of blood lead concentrations associated with death was approximately the same as that related to encephalopathy. At low levels of lead exposure in children, effects including impaired neurobehavioral development and decreased intelligence have been observed (ATSDR 2005).

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Children are more susceptible to the health effects of lead in soil than are adults for several reasons (ATSDR 2005). First, because children have more frequent hand-to-mouth behavior than do adults, young children ingest more soil and dust than adults. Second, because the gastrointestinal efficiency of lead absorption is greater in children, a bigger proportion of the amount of lead swallowed will enter the blood in children. Finally, children are physiologically more sensitive to the neurological effects of lead because of their developing nervous system.

The EPA has not established any toxicity criteria for lead (EPA 2007). Instead, potential health risks related to lead exposure are evaluated by modeling blood lead concentrations and comparing these concentrations to published criteria. A 5% probability of a child having a blood lead level exceeding 10 µg/dL is generally used as a criterion to determine whether potential blood lead levels are of concern (EPA/540/F-98-030). The blood level of 10 µg/dL derives from a 1991 recommendation by the Centers for Disease Control (CDC) (EPA 2007). However, because it is recognized that health effects are still observable below this level, the CDC convened an Advisory Committee on Childhood Lead Poisoning Prevention to consider whether the level of concern should be changed (EPA 2007). To date, the CDC has not changed the blood lead level of concern for three reasons (<http://www.cdc.gov/nceh/lead/faq/changeBLL.htm>):

1. No effective clinical interventions are known to lower blood lead levels for children with levels less than 10 µg/dL or to reduce the risks for adverse developmental effects.
2. Children cannot be accurately classified as having blood lead levels above or below 10 µg/dL because of the inaccuracy inherent in laboratory testing.
3. There is no evidence of a threshold below which adverse effects are not experienced. Thus, any decision to establish a new level of concern would be arbitrary and provide uncertain benefits.

EPA has assigned lead a WOE classification for human carcinogenicity of “B2,” a “probable human carcinogen.” This designation is based on rodent bioassays that have shown statistically significant increases in renal tumors with dietary and subcutaneous exposure to several soluble lead salts (EPA 2007). The available human evidence is considered by EPA to be inadequate to refute or demonstrate any potential carcinogenicity for humans from lead exposure.

EPA has recommended a residential screening level for lead in soil of 400 mg/kg, derived using the Integrated Exposure Uptake Biokinetic model (EPA 1994b). More recently, this screening level has been associated with bare soil in a play area and an additional screening level of 1,200 mg/kg defined for other bare-earth portions of a residential yard (EPA 2001a). The 400 mg/kg screening level was developed such that a typical child would have no more than a 5% chance of having a blood lead level exceeding 10 µg/dL, a level associated with health effects in children (EPA 1994b). Site-related residential exposures contributing to the 400 mg/kg screening level include soil ingestion from the yard and indoor ingestion of house dust contaminated with soil. In addition to these site-related exposures, the 400 mg/kg screening level incorporates background levels of lead exposure from nonsite-related sources including

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ambient air, drinking water, and diet. These background exposures were defined using “national averages, where suitable, or typical values” (EPA 1994b). The 400 mg/kg residential screening level, rather than the 1,200 mg/kg value, will be employed as a protective screening for use with the soil exposure point concentrations described in Section 5.3. The 400 mg/kg screening value will be employed across all exposure scenarios.

5.5.9 Toxicity Assessment for Detected Analytes Not Included in the Quantitative Risk Assessment

As discussed in Section 4.5, among the detected analytes in soil and groundwater several constituents were excluded from the human health and ecological risk assessments because they have negligible toxicological relevance. These included radionuclides with a half-life less than 3 years, essential nutrients present at relatively low concentrations, and silicon. The list of the chemical analytes includes the following:

- Calcium
- Iron
- Magnesium
- Silica and silicon
- Sodium
- Potassium.

Sulfate, a commonly detected analyte in groundwater, is generally soluble in water, with the exception of lead, barium, and strontium sulfate compounds. The major health effect associated with sulfate is a laxative action. The acceptably safe levels from various health organizations for infants for the ingestion of water range from 250 to 500 mg/L. However, the aesthetic quality of water is considered poor at a level of 400 mg/L (http://rais.ornl.gov/tox/profiles/sulfate_c_V1.shtml). The poor taste and color at higher concentrations most likely serves as a deterrent to consumption. However, concentrations in excess of 1,000 mg/L have been measured in groundwater samples used in this assessment. Therefore, although no toxicity criteria are available for quantification of potential health risks, the presence of higher concentrations of sulfate in groundwater is a potentially relevant metric of groundwater quality for both health and aesthetic reasons.

A number of other detected chemicals do not have toxicity criteria published by EPA. In some cases, these analytes are structurally and/or toxicologically similar to chemicals having published criteria. In such cases, these chemicals are used as toxicological surrogates; these instances are described in Tables 5-13 through 5-20. However, a number of these chemicals are unlikely to present significant human health risks and, therefore, are not included in the quantitative human health risk assessment. The basis for concluding that these chemicals present negligible potential human health risk is presented in the following paragraphs.

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Bismuth. Bismuth has a long history of use as a pharmaceutical in Europe and North America. A well known clinically used form of bismuth is bismuth subsalicylate, or Pepto-Bismol®. Most bismuth compounds are insoluble and poorly absorbed from the gastrointestinal tract with less than 1% of an oral dose being absorbed. Bismuth compounds are also poorly absorbed when applied to the skin even when the skin is abraded or burned. Acute renal toxicity is possible with oral administration of bismuth, particularly in children. Chronic toxicity with a broad spectrum of symptoms and manifestations is also possible at clinical doses (Klaassen 2001). Even though bismuth may be toxic at doses related to clinical treatment, effects from exposure to environmental concentrations are unlikely to be seen due to the very low absorption potential of bismuth. The amount of bismuth subsalicylate in a single dose of Pepto-Bismol is 524 mg, or 7.8 mg/kg bw/d for a 70-kg adult.

Bromide. Bromide has been used in over-the-counter and prescription formulations as a sedative-hypnotic drug. Currently, the bromide salt is only available in prescription drugs and as part of the antihistamine molecule brompheniramine. Oral bioavailability has been observed to be relatively high, but acute poisoning is rare. In adults, the therapeutic dose is 3,000 to 5,000 mg, while a fatal dose is estimated at 10,000 to 20,000 mg (Schonwald 2001). The dose to a 70-kg adult corresponds to 43 to 71 mg/kg and 143 to 286 mg/kg, respectively. Although bromide, like bismuth, may be toxic in clinical treatment, effects from exposure at environmentally relevant concentrations are unlikely to be seen due to the much lower dosages in such a context.

Chloride. Chloride is an essential mineral for humans. Excessive intake of chloride salts is associated with fluid retention and high blood pressure, but excess chloride is normally excreted in the urine, sweat, and bowels, so overt toxicity is generally not observed. Healthy individuals can tolerate high levels of chloride with adequate intake of fresh water. The average intake of salt from a regular salt-free diet is approximately 100 mg/day, or 1.4 mg/kg for a 70-kg adult. Adverse effects from exposure to environmental concentrations of chloride are unlikely to be seen due to the relatively low concentrations encountered in the environment and the ability of the body to process excess intake of chloride.

Heptacosane and Octacosane. Heptacosane is long chain alkane (27 carbons), which is an acyclic chemical compound consisting of carbon and hydrogen linked together by single bonds. Octacosane is a 28 carbon alkane. Alkanes longer than approximately 20 carbons are also known as paraffin and the solid forms of paraffin are called paraffin wax. Alkanes are not very reactive and have very little biological activity. Food grade paraffin wax has many uses and it is edible, but nondigestible. It passes through the body without being broken down. Because of the biological inactivity of these compounds, adverse effects from low concentrations in the environment are unlikely.

Hexadecanoic acid. Hexadecanoic acid, or palmitic acid, is one of the most common saturated fatty acids found in animal and plant products. It is a major component of the oil from palm trees. Butter, cheese, milk and meat also contain this fatty acid. Being an acid, it may cause

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irritation to the eye and skin, and upon inhalation or ingestion it may cause irritation of the mucus membranes. However, these effects are related to exposure to the pure substance and are not relevant to the low environmental concentrations measured in the environment.

Lithium. Lithium has a variety of therapeutic uses including the treatment of manic-depressive affective disorders, as well as industrial uses such as in nuclear reactor coolant. Occupational toxicity is rare; however, it does have a narrow toxic therapeutic index with chronic use and has a bioavailability over 95% via ingestion (Schonwald 2001). Even though lithium can be toxic with therapeutic use and is readily absorbed via the oral route, effects from exposure to the low concentrations encountered in the environment are unlikely to be seen.

Octadecanoic acid. Octadecanoic acid, also known as stearic acid, is a saturated fatty acid found in many animal and vegetable fats and oils. Stearic acid is an ingredient in candles, soaps, plastics, oil pastels, and cosmetics, and is used as a rubber softener. Like hexadecanoic acid, it may in a pure form cause irritation to the eye and skin, and upon inhalation or ingestion may cause irritation of the mucus membranes. It is unlikely to result in any adverse effects at environmentally relevant concentrations.

Orthophosphate. Orthophosphate, also known as phosphoric acid, is a corrosion inhibitor added to finish drinking water and is added to foods and beverages to control pH. Phosphoric acid is a generally recognized as safe (GRAS) substance by the Food and Drug Administration.

Phosphorus. Phosphorus is an essential nutrient required for the development of strong bones and teeth, for metabolism of fats and carbohydrates, and for protein synthesis. The recommended dietary intake is 100 mg/day for infants less than 1 year old to 1,250 mg/day for children and pregnant/ lactating women younger than 18. It is unlikely to result in any adverse effects at environmentally relevant concentrations.

Sodium dithionite. Sodium dithionite is a water-soluble salt used as a reducing agent in industry. It has slight oral toxicity, causing gastrointestinal disturbances. The probable oral lethal dose (human) is 500 to 5000 mg/kg, or between 1 ounce or 1 pint for a 70-kg person (<http://toxnet.nlm.nih.gov/cgi-bin/sis/search/f?/temp/~yrzUu6:1>). Based on this information, the low environmental concentrations of this chemical are not expected to be associated with adverse health effects in humans.

5.6 RISK CHARACTERIZATION

The risk characterization for each investigation area will differentiate between total risk and background risk, where total and background risks are calculated using exposure point concentrations described in Section 5.3. Differences between total risk and background risk will be identified. Where total risk exceeds background risk for any exposure scenario, the increment is attributable to one or more COPCs and exposure pathways. The risk results and associated interpretation are summarized in Section 5.7.1 and presented in detail in Sections 5.7.2 through 5.7.6 of the RCBRA. Results of the groundwater risk assessment are presented in Sections 5.8.1 through 5.8.3.

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A discussion of the general protocol for characterizing risk for each of the health effects endpoints addressed in the RCBRA is provided in Sections 5.6.1 through 5.6.4. The uncertainty analysis for the waste site risk assessments and groundwater assessment is presented in Sections 5.7.9 and 5.8.4, respectively.

5.6.1 Carcinogenic Effects of Chemicals

Cancer risk is evaluated as the incremental probability that an individual will develop cancer during their lifetime. An incremental lifetime cancer risk (ILCR) is the product of the average daily dose (i.e., chemical intake) and a cancer slope factor. Chemical intake and CSFs are described in Sections 4 and 5.3 of this methodology, respectively. The ILCR is calculated as:

$$\text{ILCR} = \text{Intake} \left(\frac{\text{mg}}{\text{kg} \cdot \text{d}} \right) \times \text{CSF} \left(\frac{\text{mg}}{\text{kg} \cdot \text{d}} \right)^{-1}$$

where ILCR = incremental lifetime cancer risk
 Intake = daily intake across all exposure pathways
 CSF = cancer slope factor.

Incremental cancer risks for each exposure route and chemical are then summed to calculate the total ILCR to an individual. The acceptability of any calculated excess cancer risk is generally evaluated relative to the target risk range of 10^{-6} to 10^{-4} described in the NCP. As a point of comparison, national cancer statistics have been developed that provide the background incidence of cancer for the U.S. population. For example, national cancer statistics indicate that each male has approximately a 1-in-2 chance of developing cancer during his lifetime and that each female has approximately a 1-in-3 chance of developing cancer in her lifetime (American Cancer Society 2005). An individual with a theoretical ILCR of 1-in-100,000⁷ due to site-related exposure therefore has an approximate total cancer risk of 50,001-in-100,000 (male) or 33,334-in-100,000 (female), where the “background” levels are 50,000-in-100,000 (male) and 33,333-in-100,000 (female), respectively.

In most risk assessments, the carcinogenic risks from all carcinogenic chemicals are treated as additive and summed to produce an overall estimate of carcinogenic risk from the site (EPA 1989). Interactions that alter the toxicity may also occur among chemicals in a mixture. That is, the potential exists for synergistic effects or antagonistic effects. Synergistic effects occur when the combined effects are greater than the toxicity of each component of a mixture individually, while antagonistic effects occur when the combined effects are less than the toxicity of each component of a mixture individually. Failure to consider potential synergistic or antagonistic effects on toxicity may result in either an underestimation or an overestimation (similar to the assumption of additivity) of the risk, respectively.

⁷ Risk assessors denote a one-in-one hundred thousand risk as “1E-05” or “ 1×10^{-5} ”.

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In practice, synergistic or antagonistic effects between the carcinogens are difficult to quantify due to the lack of information on the toxicity of specific chemical mixtures. Because slope factors are upper 95th percentile estimates and are not strictly additive, the total cancer risk estimate becomes increasingly biased in a conservative manner as the number of summed carcinogens increases. Summing the risks from all carcinogens equally also gives as much weight to Class B or C carcinogens as to Class A carcinogens, and gives CSFs derived from animal data the same weight as CSFs derived from human data. Uncertainties and protective biases introduced in the risk characterization for carcinogenic effects will be addressed in the uncertainty analysis.

5.6.2 Noncarcinogenic Effects of Chemicals

Noncarcinogenic effects for individual chemicals are expressed as hazard quotients (HQs). An HQ is the ratio of the average daily dose (i.e., chemical intake) of a chemical to the corresponding RfD for that chemical. Chemical intake and RfDs were discussed in Sections 5 and 5.3 of this methodology, respectively. The HQ is calculated as:

$$HQ = \frac{\text{Intake} \left(\frac{\text{mg}}{\text{kg} \cdot \text{d}} \right)}{\text{RfD} \left(\frac{\text{mg}}{\text{kg} \cdot \text{d}} \right)}$$

where Intake = chronic daily intake

HQ = hazard quotient

RfD = reference dose.

Hazard quotients for each chemical may be summed to calculate a hazard index (HI) across chemicals for each exposure pathway if target organs and mechanisms of toxicity are similar. In some cases, additivity may be protectively assumed even in situations where target organs and mechanisms of toxicity are dissimilar. Hazard indices across exposure pathways may also be summed to calculate an overall HI. It should be noted that there is also a possibility of synergistic effects among chemicals, where simple additivity may in fact underestimate potential toxicity. An HQ or HI value of greater than 1.0 is indicative of the potential for adverse effects.

The level of concern does not increase linearly as an HI of unity is approached or exceeded because the individual HQ values do not have equal accuracy or precision and are not based on the same severity of effect. RfDs are also associated with varying levels of confidence due to differences in the uncertainty and modifying factors across chemicals. Uncertainties and protective biases introduced in the risk characterization for noncarcinogenic effects will be addressed in the uncertainty analysis.

Because the uncertainties related to exposure to chemical mixtures affect whether the risk is over- or underestimated, it is important to determine the conditions under which additivity versus synergism may occur. For example, *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (EPA 630-R-00-002) suggests that additivity be assumed as a

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“default” approach when mixture components are at low doses and when toxicity occurs via the same mechanism. In this assessment, values of HI will initially be calculated across all chemicals and exposure routes. If an HI is potentially significant, the issue of similarity of the toxicological mechanisms of action across the major contributors to the HI will be explored in the uncertainty analysis.

5.6.3 Radiation Cancer Risk

Radiation cancer risk, like chemical cancer risk, is evaluated as the incremental probability that an individual will develop cancer during their lifetime. An ILCR is the product of the average daily radionuclide intake or external dose and a cancer slope factor. Radionuclide intake and CSFs are described in Sections 5.4 and 5.5, respectively. The ILCR is calculated as:

$$\text{ILCR} = \text{Intake (pCi)} \times \text{CSF (pCi)}^{-1}$$

where ILCR = incremental lifetime cancer risk
 Intake = daily intake across all exposure pathways
 CSF = cancer slope factor.

The units in the equation for external irradiation differ as described in Sections 5.4.6 and 5.5.4:

$$\text{ILCR} = \text{Intake (pCi - hr/g soil)} \times \text{CSF} \left(\frac{\text{risk/yr}}{\text{pCi/g}} \right)^{-1} \times 1.14 \times 10^{-4} \text{ yr/hr}$$

Incremental cancer risks for each exposure route and radionuclide are summed to calculate the total ILCR to an individual. The acceptability of any calculated excess cancer risk is evaluated relative to the target risk range of 10^{-6} to 10^{-4} described in the NCP. A context for this risk, with respect to national background rates of cancer, is provided in Section 5.6.1.

5.6.4 Radiation Dose

Radiation dose, which is a measure of the amount of energy deposited in body tissues, is calculated as the product of the intake or exposure rate for a single radionuclide and the DCF for that radionuclide and exposure route. The specific radiation dose associated with the EPA dose conversion factors from *Federal Guidance Report No. 11* and *Federal Guidance Report No. 12* is the annual committed effective dose equivalent (internal) or annual effective dose equivalent (external). Radionuclide intake and DCFs are described in Sections 5.4 and 5.5, respectively.

Internal and external radiation dose are calculated as:

$$\text{Dose} = \text{Intake} \times \text{DCF}$$

where Dose = annual radiation dose (mrem/yr)
 Intake = annual intake across all exposure pathways (pCi/yr)
 DCF = dose conversion factor (mrem/pCi).

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The units in the equation for external irradiation differ as described in Sections 5.4.6 and 5.5.4:

$$\text{Dose} = \text{Intake} \times \text{DCF} \times 1.14 \times 10^{-4} \text{ yr/hr}$$

where Dose = annual radiation dose (mrem/yr)

Intake = annual intake across all exposure pathways (pCi-hr/g soil-yr)

DCF = dose conversion factor (mrem/yr/pCi/g).

Radiation doses for each exposure route and radionuclide are summed to calculate the total annual dose to an individual. The acceptability of a calculated annual dose is evaluated for human receptors in the RCBRA relative to a threshold dose limit of 15 mrem/yr. The DOE has published health and safety orders of which DOE Order 5400.5, *Radiation Protection of the Public and the Environment*, is most pertinent to the identification of a radiation dose threshold. DOE Order 5400.5 requires the reduction of all DOE-source radiation doses to a level as low as reasonably achievable (ALARA) below a primary dose limit of 100 mrem/yr above background. CERCLA authorizes the EPA to regulate hazardous substances, including radionuclides, released into the environment. EPA has published guidelines for establishing cleanup levels for radionuclides under CERCLA that state that 15 mrem/yr above background levels should “generally be the maximum dose limit for humans” (EPA 1997).

By comparison, the average annual radiation dose in the United States from natural and man-made sources of radiation is about 360 mrem/yr (NRC 2004). Natural sources of radiation, predominantly radon gas, make up about 80% of this total. The remainder is related to such sources as medical x-rays, nuclear medicine, and consumer products. In some areas of the country, however, background radiation dose may be considerably higher. For example, the average annual dose in Denver, Colorado, is over 1,000 mrem/yr (NRC 2004).

5.7 HUMAN HEALTH RISK ASSESSMENT RESULTS

The scope of the human health risk assessment, as it relates to exposure scenarios, pathways, and receptors, is described in Section 5.2 and outlined in Table 5-1. The specific health effects for which risks are quantified are described in Section 5.6. Human health risk assessment calculations have been conducted primarily on the scale of an individual waste site (i.e., “local area”) and individual monitoring well. Additionally, in the Recreational scenarios and the occupational portion of the Resident Monument Worker scenario, as well as for certain exposure pathways within the CTUIR and Rural-Residential scenario, risk calculations are conducted over much larger regions (i.e., “broad area”). A crosswalk of spatial scale with exposure scenarios and pathways is shown in Table 5-2. As discussed in Section 5.2, risks related to groundwater exposures are calculated independently of risks associated with other exposure media.

The specific waste sites that are included in the human health risk assessment are those for which remedial activities (if necessary) have been completed and soil confirmation data exist. These waste sites, organized according to operational area and with reference to the associated CVP or waste site reclassification form document, are listed in Table 5-23. There are a total of 163 waste sites: 45 waste sites in the 100-B/C Area, 28 in the 100-D Area, 38 in the 100-F Area, 8 in

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the 100-H Area, 14 in the 100-K Area, 2 in the 100-N Area, 6 in the 100-IU-2 OU, 5 in the 100-IU-6 OU, and 17 in the 300 Area.

The draft 100-B/C Pilot Project risk assessment (DOE/RL-2005-40) employed historical CVP and RSVP waste site soil confirmation data from approximately 35 waste sites in the 100-B/C Area. This risk assessment also includes the CVP and RSVP soil data for these sites, as well as for approximately 10 additional sites in the 100-B/C Area. Riparian soil data, sediment data, and aquatic organisms data collected under the 100-B/C Pilot Project (DOE/RL-2005-40) and the 100-NR-2 investigation (DOE/RL-2005-22) are also included in this assessment, as described in Section 5.2.

Risk assessment calculations for the Rural-Residential, CTUIR, Industrial/Commercial Worker, and Resident Monument Worker scenarios are presented for each individual remediated waste site. As discussed in Sections 5.2 and 5.3, soil exposures for these scenarios occur primarily in the vicinity of a residence and incorporate contributions from residual contamination related to the individual waste site. Risk assessment calculations for the Recreational scenarios (Avid Angler, Avid Hunter, and Casual Use) occur over broader spatial scales, for which exposure concentrations are integrated across all of the 100 Area and 300 Area (see Section 4.0). Therefore, only a single set of risk assessment results are presented for each of the Recreational scenarios.

Background risk calculations for all scenarios, as discussed in Section 5.3.2, employ the reference area soil data collected under the SAP (DOE/RL-2005-42). For the exposure scenarios that are implemented on the scale of an individual waste site (Rural-Residential, CTUIR, Industrial/Commercial Worker, and Resident Monument Worker scenarios), risk calculations are also conducted using the operational area upland soil data collected under the SAP (DOE/RL-2005-42). These additional "background" risk calculations pertain to a "no excavation" model, where a residence or commercial structure is located at some distance from a waste site or otherwise does not intrude into subsurface soils represented by the CVP/RSVP soil data. Some background data were also collected during the 100-B/C Pilot Project and the 100-NR-2 ecological investigation. However, these are an insufficient number of these background samples to quantify specific background risks for these individual operational areas.

The risk assessment results for exposures related to fish ingestion (represented by sculpin tissue data collected under the SAP [DOE/RL-2005-42]) are calculated and presented independently of the risk related to other exposure pathways. This impacts the risk results for the CTUIR, Rural-Residential, and Avid Angler scenarios. The main reason for this action is that calculated exposure point concentrations for certain organic chemicals in fish tissue (particularly carcinogenic polyaromatic hydrocarbons and, to a lesser extent, PCBs) result in fish ingestion cancer risk, dose, and hazard values far greater than the values related to other exposure pathways. Summing the fish ingestion risks with those of other exposure pathways would render the risk assessment results for these scenarios insensitive to any other factors. However, the analytical results for these constituents in fish tissue indicate that detected concentrations are virtually all within the range of nondetect (U-qualified) values. A plot of detected and nondetect polyaromatic hydrocarbon and PCB concentrations in fish tissue is shown in Figure 5-6a.

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Similar plots for pesticides, and for other SVOCs, are shown in Figures 5-6b and 5-6c, respectively. Because the range of nondetect results commonly exceeds the detected concentrations, confidence in the exposure point concentrations based on these analytical results is low.

Risk assessment results for a subset of naturally occurring radionuclides are also calculated and presented independently of risks for other analytes. These radionuclides include potassium-40, radium-226, radium-228, thorium-228, thorium-230, and thorium-232. As discussed in Section 4.3, the concentrations of these analytes are practically identical in operational area and reference area samples. These radionuclides are also not associated with historical Hanford Site processes and operations. The contribution of background levels of these radionuclides to calculated cancer risk and radiation dose results is very high, such that the potential impacts of residual levels of Hanford Site-related contamination is indiscernible when these radionuclides are included in the risk calculation sums.

As discussed in Section 5.2.2, pathway-specific health risks related to groundwater contamination for the Rural-Residential, CTUIR, and Resident Monument Worker exposure scenarios are calculated for each monitoring well in each investigation area. However, these risks are not directly summed with those related to other exposure pathways because there is no tractable way at this time to correlate existing groundwater data with potential future groundwater exposure concentrations for the individual waste sites. Instead, the range of potential groundwater-related risks across the monitoring wells (see Section 5.8) are discussed in relation to risks from other pathways to support risk management decisions.

To support the uncertainty analysis (Section 5.7.9), risk results are calculated using both best-estimate (CTE) and upper bound (RME) values for exposure point concentrations and exposure parameter values. Exposure parameter values are summarized for RME and CTE assumptions in Tables 5-8 and 5-9, respectively.

Electronic files of the representative concentrations used to calculate exposure point concentrations in the various exposure media, the exposure point concentrations, and the human health risk assessment results are provided in electronic format in Appendices F-5 and G-1.

5.7.1 Summary of the Risk Assessment Results

A summary of the results of the RME human health risk calculations across all exposure scenarios is presented in Table 5-24. The first four scenarios listed in Table 5-24 are scenarios for which risks are calculated on a relatively small spatial scale, such that an individual risk calculation is conducted for each of the 163 remediated waste sites. As indicated in this table, the range of soil-related⁸ RME risk results across the 163 remediated waste sites was often as great as a factor of 100 and occasionally even larger. To a great extent, the range of these risk results shown in Table 5-24 are skewed by a relatively few remediated waste sites where RME risk calculations are inordinately affected by very high UCL values for certain analytes. This can

⁸ Including exposure via foodstuffs where contaminant concentrations in foods are modeled from soil data; see Table 5-1.

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be observed in Figures 5-7 through 5-9, and Figures 5-16 through 5-21, where the range of CTE results is more often about a factor of 10. These figures are particularly suitable for this inspection because the variability shown in the RME results (i.e., the spread of the calculated risks above a theoretical line identical to the CTE results but shifted “higher” on the y-axis due to different behavioral assumptions) is wholly a function of the calculation of the UCL values for exposure point concentrations in soil. The location of the RME risk results in these figures if they were calculated using the mean rather than the UCL value for COPC concentrations would be at or below a line drawn to intersect the lowest of the plotted RME risk results.

The specific remediated waste sites associated with some of the highest calculated RME cancer risks and/or radiation dose are relatively consistent across all four of the scenarios for which risks are calculated on a relatively small spatial scale (Rural-Residential, CTUIR [local exposures], Resident Monument Worker, and Industrial/Commercial Worker). A selection of these waste sites, with the analytes that are predominantly associated with the calculated risks across one or more of the four scenarios and either cancer risk and/or radiation dose, include the following:

- 316-5 (arsenic, isotopic uranium, cesium-137, cobalt-60)
- 316-2 (isotopic uranium, arsenic, cobalt-60)
- 300-10 (arsenic)
- 116-F-14 (isotopic europium, cesium-137, cobalt-60)
- 316-1 (arsenic, isotopic uranium)
- 100-F-35 (strontium-90, arsenic, cesium-137)
- 100-F-37 (arsenic)
- 118-B-3 (isotopic europium, arsenic, cesium-137)
- 116-B-11 (isotopic europium, cesium-137)
- 118-F8-1 (isotopic europium, cesium-137).

A different set of waste sites is predominantly associated with the highest calculated HI values, but again the sites are relatively consistent across all four of scenarios for which risks are calculated on a relatively small spatial scale. A selection of these waste sites for the Rural-Residential and CTUIR scenarios (RME HI values did not exceed 1.0 for any remediated waste sites under the Resident Monument Worker, and Industrial/Commercial Worker exposure scenarios) include the following:

- 100-K-33 (mercury)
- 100-K-30 (mercury)
- 128-C-1 (copper)
- 110-K-32 (mercury)
- 300-10 (arsenic).

For the Rural-Residential, CTUIR, Resident Monument Worker, and Industrial/Commercial Worker exposure scenarios, “background” risks are calculated and presented in two ways. The fourth column of Table 5-24 shows “background” risks for soil-related exposures that are

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calculated using soil data from samples collected in the reference area. The reference area risk calculations are intended to represent background levels of risk for the soil COPCs evaluated in the risk assessment independent of the impacts of the Hanford Site. These are the background risk values that are used, by subtraction from the total risk values for these scenarios shown in the first column of Table 5-24, to calculate incremental cancer risk and dose. The results shown in the “operational area (No Excavation)” for the Rural-Residential, CTUIR, Resident Monument Worker, and Industrial/Commercial exposure scenarios present the results for these scenarios calculated using present-day surface soil COPC concentrations across the upland portions of the 100 Area and 300 Area. The term “no excavation” is applied because these results are intended to represent potential risks under the condition that these scenarios occur without excavation into residual subsurface contamination.⁹ Conceptually, these results portray a situation where a hypothetical residence or commercial structure is located within the operational area but the construction of which does not intrude into the subsurface soils represented by the CVP/RSVP soil verification data.

For the Rural-Residential and CTUIR (with only local exposures) exposure scenarios, arsenic via produce ingestion was the primary contributor to background cancer risks. In the variation of the CTUIR scenario where native plants and wild game were evaluated, exposure to arsenic, PCBs, and pesticides via wild plants were of most significance to background cancer risks. Produce and native plant exposures were also the primary exposure pathways for cancer risks related to the remediated waste sites for these scenarios. Exposure to isotopic europium and cobalt-60 via external irradiation was the main contributor to radiation dose for the Rural-Residential, Resident Monument Worker, and Industrial/Commercial Worker scenarios. For the CTUIR scenario, exposure to radionuclides including americium-241, strontium-90, and carbon-14 via plant ingestion was of primary importance for background radiation dose.

The operational area risk results for the Recreational scenarios (Casual User, Avid Hunter, and the sediment-based exposure pathways of the Avid Angler scenario) shown in Table 5-24 reflect exposures that occur over a larger area than that associated with any particular waste site. The relatively higher RME cancer risk and HI values for the Avid Hunter reflect modeled exposures to PCBs and certain metals in soil via ingestion of wild game. The range of results shown for the Avid Angler exposure scenario pertains to the four exposure areas where COPC sediment concentrations were differentiated: the 100-B/C Area, the 100-N Area, the 300 Area, and the entire 100 Area assessed in aggregate.

As noted in the introductory paragraphs of Section 5.7, the risk assessment results for exposures related to fish ingestion are calculated and presented independently of the risk related to other exposure pathways. Risks related to a subset of naturally occurring radionuclides (potassium-40, radium-226, radium-228, thorium-228, thorium-230, and thorium-232) that are not associated with Hanford Site operations are also calculated and presented independently. The reason that these risk calculations are not integrated with the other risk assessment results is that the relative risks shown in these calculations are very high, such that the potential impacts of residual levels of Hanford Site-related contamination in the remediated waste sites and in operational area

⁹ A discussion of the basement excavation and water well model used to calculate a hypothetical surface soil exposure area related to subsurface residual contamination is presented in Section 5.3.2.

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surface soils is indiscernible when these results are included in the risk calculation sums. For the fish ingestion pathway, the high risk results are an artifact of the calculated exposure point concentrations for certain organic chemicals in fish tissue (particularly carcinogenic polycyclic aromatic hydrocarbons and, to a lesser extent, PCBs) being inordinately affected by elevated detection limits and also to widespread levels of these and other organic compounds being present in fish of the Columbia River Basin (EPA 910-R-02-006).

Potential risks related to groundwater were calculated for 64 monitoring wells sampled under the SAP (DOE/RL-2005-42). The purpose of evaluating possible groundwater-related risks is primarily to provide an approximate measure of the relative significance of soil and groundwater as exposure media in the 100 Area and 300 Area. Cancer risk and radiation dose values across the Rural-Residential, CTUIR, and Resident Monument Worker scenarios were approximately equivalent for the soil-related exposure pathways (not including the thorium, radium, and potassium isotopes) and the groundwater exposure pathways. With the inclusion of these radionuclides, soil-related risks at the various remediated waste sites generally exceeded those calculated for groundwater exposures. The calculated HI values were generally somewhat higher for the soil-related exposure pathways. There are a small subset of monitoring wells with high concentrations of Aroclor-1254 (well A4614 in the 100-H Area) and strontium-90 (wells A9910 and A4679) in groundwater where calculated cancer risk and dose were higher than at most of the remediated waste sites. Also, very high cancer risk estimates for certain wells with elevated concentrations of hexavalent chromium (B8778, B8753, A4570, B8750, 199-N-80, A4647, C4670, 199-K-22, and A4600) were calculated for the CTUIR exposure scenario via the sweat lodge inhalation exposure pathway. There are, however, significant protective biases inherent in the exposure estimates for this pathway. An important distinction between the risk calculations for groundwater wells and remediated waste sites is that the contribution of background to the total calculated risks was generally quite small for the monitoring wells in comparison to the remediated waste sites.

Risks Related to Lead in Soil. The highest calculated RME representative concentration for lead in soil at any waste site is 210 mg/kg at 100-F-37. The next highest RME representative concentrations for lead are approximately 100 mg/kg at 1607-8 and 50 mg/kg at 628-4. No representative concentrations for lead at any waste site are approaching EPA's recommended residential screening level for lead in soil of 400 mg/kg, derived using the Integrated Exposure Uptake Biokinetic model, that has been associated with bare soil in a play area (EPA 2001a). Because soil concentrations for lead are well below the most restrictive of EPA's soil screening criteria, no additional evaluation of lead is included in this risk assessment. More detailed information on lead toxicity and the basis of EPA's soil screening criteria is provided in Section 5.5.8.

Broad area soil RME representative concentrations for lead are 32 mg/kg and 45 mg/kg for upland and riparian soils in operational areas, respectively. As noted in Section 5.7.9.1, variability in lead results among the five MIS at waste site 600-131 was considerable. The highest individual lead sample result was 327 mg/kg, although the average over all five samples at 600-131 was only 116 mg/kg.

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5.7.2 Rural-Residential Scenario Results

The Rural-Residential exposure scenario encompasses potential exposures related to surface and subsurface soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in potentially affected areas. This scenario also includes exposures related to ingestion of Columbia River fish and exposure to groundwater via a domestic water well. Risks related to fish ingestion and groundwater use are addressed in Sections 5.7.7 and 5.8, respectively. The scenario is more fully described in the CSM (Section 2.0) and in Section 5.2, and is summarized in Table 5-1.

Cancer Risk

The range of RME total cancer risk results for the Rural-Residential scenario (radionuclides + chemicals) is from $2\text{E-}04$ to $7\text{E-}03$, with a reference area RME background cancer risk value of $2\text{E-}04$. The operational area (no excavation) RME cancer risk is slightly higher than background, being $3\text{E-}04$. The range of the CTE total cancer risk results is smaller, being from $1\text{E-}05$ to $1\text{E-}04$, with a reference area CTE background cancer risk value of $3\text{E-}05$. The operational area (no excavation) CTE cancer risk is $4\text{E-}05$. Rural-Residential RME and CTE total cancer risk results are tabulated by waste site, according to magnitude, in Tables 5-25a and 5-25b, respectively. Per EPA guidance (EPA 1989), cancer risk results are displayed with only one significant figure.

The range of Rural-Residential RME and CTE total cancer risk values (radionuclides + chemicals) for all 163 waste sites, relative to operational area and reference area background risks, is shown in Figure 5-7. As indicated in the figure legend, RME and CTE total cancer risk values for reference area background and operational area (no excavation) cancer risk are shown as points of reference. The waste sites plotted along the x-axis are ranked according to the magnitude of the CTE total cancer risk. The variability shown in the RME results (i.e., the spread of the calculated risks above a theoretical line along the lowest calculated RME values) is a function of the protocols used in calculating the UCL exposure point concentrations in soil.

The Rural-Residential scenario includes three exposure pathways (beef ingestion, milk ingestion, and dust inhalation) that may be assessed on either a local or broad spatial scale (see Section 5.3). The summary of risk results in the previous paragraph pertains to the "local area" risk calculation. The results of the "broad area" calculations are generally higher than the local area results, differing by up to about 25% from the "local area" results. Because the higher "broad area" risk values are related to beef and milk ingestion, this finding is likely due to the modeling of direct soil ingestion by grazing cattle in the "broad area" risk calculations but not for penned cattle in the "local area" calculations.

The particular exposure pathways and analytes contributing to total cancer risks at the remediated waste sites vary most for sites with calculated risks highest above the operational area baseline value. As site risks approach the operational area baseline, the majority of the calculated risk is a function of the same baseline conditions in surface soil. The following is a breakdown of the RME total cancer risks for remediated waste sites with the highest calculated RME risk values.

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316-5	(65% Cs-137 via external; 30% U-238, mostly via poultry)
316-2	(70% U-234/235/238, mostly via poultry and external)
300-10	(95% arsenic, via produce and other foods)
116-F-14	(60% Eu-152, via external; 20% Cs-137 and Co-60 combined, via external)
316-1	(50% arsenic, mostly via produce; 34% U-238 and U-234, mostly via poultry and external)
100-F-35	(55% Sr-90 via produce and milk; 25% arsenic, mostly via produce)
100-F-37	(83% arsenic via produce and other foods)
118-B-3	(49% Eu-152 via external; 25% arsenic, via produce and other foods)

The following is a breakdown of the CTE total cancer risks for remediated waste sites with the highest calculated CTE risk values.

316-5	(40% U-238; 35% Cs-137 and U-234 combined – all mostly via external)
316-2	(60% U-234/235/238, mostly via poultry and external; 30% arsenic via produce and other foods)
100-F-37	(87% arsenic via produce and other foods)
116-F-14	(50% Eu-152 via external; 24% arsenic, via produce and other foods)
316-1	(76% arsenic, via produce and other foods)
100-D-49:4	(44% arsenic, via produce and other foods; 35% Eu-152 via external)
100-F-35	(44% arsenic via produce and other foods; 22% Sr-90, mostly via produce and milk; 18% Cs-137 via external)
116-C-5	(45% arsenic, via produce and other foods; 24% Eu-152 via external)

Both reference area and operational area background cancer risk for the Rural-Residential scenario is primarily arsenic (55% to 75%), primarily via produce and other biota. Isotopes of europium contribute 5% to 10% to reference area and operational area CTE risk values. For the RME operational area risk results, alpha-BHC and PCBs contribute about 10% of the calculated risk results.

The ILCR for radionuclides and chemicals, calculated as total cancer risk for each remediated waste site minus the reference area background cancer risk of risks of 2E-04 (RME) and 3E-05 (CTE), is shown in Tables 5-26a and 5-26b. The range of Rural-Residential scenario ILCR values (radionuclides + chemicals) can be seen graphically as the difference between waste site risks and the reference area background risk in Figure 5-7. The RME ILCR for the majority of remediated waste sites is in the range of 1 to 3E-04, with four sites having RME ILCR values above 1E-03. CTE ILCR values are all within or below EPA's target risk range of 10⁻⁶ to 10⁻⁴ described in the NCP.

It is important to note that, as discussed in the introductory paragraphs of Section 5.7, these cancer risk results do not include contributions from fish ingestion, certain naturally occurring radionuclides in soil, or domestic uses of groundwater. RME risks for these calculations are

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shown relative to the range of risks calculated for the remediated waste sites in Table 5-24 and are presented in detail in subsequent sections of this risk assessment.

Radiation Dose

The range of RME total radiation dose results for the Rural-Residential scenario is from 1.0 to 370 mrem/yr, with a reference area RME background dose value of 1.8 mrem/yr. The operational area (no excavation) RME dose is slightly higher than background, being 2.7 mrem/yr. The range of the CTE total radiation dose results is smaller, being from 0.39 to 26 mrem/yr, with a reference area CTE background dose value of 1.3 mrem/yr. The operational area (no excavation) CTE radiation dose is 1.6 mrem/yr. Rural-Residential RME and CTE total radiation dose results are tabulated by waste site, according to magnitude in Tables 5-27a and 5-27b.

The range of Rural-Residential RME and CTE total radiation dose values for all 163 waste sites, relative to operational area and reference area background dose, is shown in Figure 5-8. As indicated in the figure legend, RME and CTE total radiation dose values for reference area background and operational area (no excavation) radiation dose are shown as points of reference. The waste sites plotted along the x-axis are ranked according to the magnitude of the CTE total radiation dose. The variability shown in the RME results (i.e., the spread of the calculated doses above a theoretical line along the lowest calculated RME values) is a function of the protocols used in calculating the UCL exposure point concentrations in soil.

As discussed for the Rural-Residential cancer risk results, this scenario includes three exposure pathways (beef ingestion, milk ingestion, and dust inhalation) that may be assessed on either a local or broad spatial scale (see Section 5.3). The waste site dose results shown in Figure 5-8 pertain to the "local area" risk calculation. However, the results of the "broad area" calculations for Rural-Residential radiation dose are essentially identical to those shown.

The particular exposure pathways and analytes contributing to total radiation dose at the remediated waste sites vary most for sites with calculated doses highest above the operational area baseline value. As site doses approach the operational area baseline, the majority of the calculated dose is a function of the same baseline conditions in surface soil. The following is a breakdown of the RME total radiation dose for remediated waste sites with the highest calculated RME dose values. At waste site 118-F-8:1, the RME total radiation dose is about 17 mrem/yr.

316-5	(57% Cs-137 via external; 36% U-238, mostly via poultry)
316-2	(30% U-235, 65% via external; 28% each U-234 and U-238, mostly via poultry)
116-F-14	(70% Eu-152 via external; 14% Cs-137 and 11% Co-60, via external)
100-F-35	(80% Sr-90, via produce and other foods; 12% Cs-137 via external)
316-1	(53% U-238, mostly via poultry and external; 36% U-234, via poultry)
118-B-3	(76% Eu-152 via external; 14% Cs-137 via external)
116-B-11	(49% Eu-154 via external; 19% Eu-152, 12% Cs-137 and 10% Co-60 via external)
118-F-8:1	(60% Eu-152 via external; 24% Cs-137 via external)

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The following is a breakdown of the CTE total radiation dose for remediated waste sites with the highest calculated CTE dose values. The next ranked site for CTE total radiation dose, 116-F-14, has a calculated CTE dose of just 9.3 mrem/yr.

- | | |
|-------|--|
| 316-5 | (45% U-238, mostly via poultry and external; 28% U-234 via poultry; 14% Cs-137 via external;) |
| 316-2 | (42% U-238; 32% U-234; 17% U-235) |

Both reference area and operational area background radiation dose for the Rural-Residential scenario is associated primarily with external irradiation via short-lived radioisotopes including isotopes of europium, cobalt-60, and cesium-137. Because the half-lives of these isotopes are on the order of 30 years or less and they are not naturally occurring, background dose levels related to these isotopes in soil will decrease relatively quickly over time.

The incremental radiation dose for radionuclides, calculated as total radiation dose for each waste site minus the reference area background radiation dose of 48 mrem/yr (RME) and 27 mrem/yr (CTE) is shown in Tables 5-28a and 5-28b, respectively. The range of incremental dose values can be seen graphically as the difference between waste site doses and the reference area background dose in Figure 5-8. The RME incremental dose for the majority of remediated waste sites is in the range of 1 to 10 mrem/yr, with eight sites having RME incremental dose values above 15 mrem/yr. With two exceptions (316-5 and 316-2), CTE incremental dose values are all below 15 mrem/yr.

As was stated for the cancer risk results, the radiation dose results do not include contributions from fish ingestion, certain naturally occurring radionuclides in soil, or domestic uses of groundwater. RME doses for these calculations are shown relative to the range of doses calculated for the remediated waste sites in Table 5-24 and are presented in detail in subsequent sections of this risk assessment.

Chemical Hazard

As described in Section 5.6.2, chemical hazard is calculated separately for adult and child receptors. For the Rural-Residential scenario, RME and CTE child HIs generally exceed those for adults by a factor of approximately 5% to 15%. Because they are consistently higher, only child HI values are presented here.

The range of RME total child HI results for the Rural-Residential scenario is from 5 to 200, with a reference area RME background HI value of 20^{10} . The operational area (no excavation) RME HI is lower than reference area background, being just 8. The range of the CTE total HI results is smaller, being from 2 to 8, with a reference area CTE background HI value of 2. The operational area (no excavation) CTE HI is 2. Rural-Residential RME and CTE child HI results are tabulated by waste site, according to magnitude, in Tables 5-29a and 5-29b. Because the majority of EPA reference dose toxicity criteria are presented with only one significant figure, HI results are also displayed with only one significant figure.

¹⁰ The relatively high reference area HI is related to an elevated UCL for thallium in reference area soil.

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The range of Rural-Residential child HI values for all 163 waste sites, relative to operational area and reference area background HI values, is shown in Figure 5-9. As indicated in the figure legend, RME and CTE total HI values for reference area background and operational area (no excavation) HI are shown as points of reference. The waste sites plotted along the x-axis are ranked according to the magnitude of the CTE total HI. The variability shown in the RME results (i.e., the spread of the calculated HI values above a theoretical line along the lowest calculated RME values) is a function of the protocols used in calculating the UCL exposure point

As discussed for the Rural-Residential cancer risk results, this scenario includes three exposure pathways (beef ingestion, milk ingestion, and dust inhalation) that may be assessed on either a local or broad spatial scale (see Section 5.3). For these few remediated waste sites where HI values are highest, "local area" HI values are well above the equivalent "broad area" values. For the remaining sites, the broad area values are generally about 50% higher than those for the local area. Because the higher "broad area" HI values are related to beef and milk ingestion, this finding is likely due to the modeling of direct soil ingestion by grazing cattle in the "broad area" risk calculations but not for penned cattle in the "local area" calculations. The waste site child HI results shown in Figure 5-9 pertain to the "local area" risk calculation.

The particular exposure pathways and analytes contributing to total child HI values at the remediated waste sites vary most for sites with calculated HIs highest above the operational area baseline value. As site HIs approach the operational area baseline, the majority of the calculated HI is a function of the same baseline conditions in surface soil. The following is a breakdown of the RME total HI values for remediated waste sites with the highest calculated RME values. Note that at remediated waste site 100-K-33, the calculated soil UCL for mercury is approximately 100 times larger than the mean.

100-K-33	(96% mercury, via beef and produce)
100-K-30	(80% mercury, via beef and produce)
128-C-1	(93% copper, mostly via produce and milk)
100-K-32	(78% mercury, via beef and produce)
300-10	(73% arsenic via produce and poultry)
100-K-31	(46% mercury, via beef and produce; 37% organics, mainly PCBs via beef and milk; 12% cadmium, mostly via produce)
618-4	(64% organics, mainly PCBs via beef and milk; 11% thallium, mostly via produce)
600-23	(60% organics, mainly PCBs via beef and milk; 10% thallium, mostly via produce)

The following is a breakdown of the CTE total HI values for remediated waste sites with the highest calculated CTE values. The total CTE HI at remediated waste site 100-F-37 is about 4. The relative contribution of analytes and pathways for most other sites does not differ significantly from that for 100-F-37.

100-K-30	(39% mercury, via beef and produce; 11% thallium, mostly via produce; 7% arsenic)
128-C-1	(59% copper, mostly via produce and milk; 7% arsenic via produce and poultry)
100-K-31	(55% mercury, via beef and produce; 7% thallium)

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- 100-K-32 (56% mercury, via beef and produce; 7% thallium)
 100-F-37 (47% arsenic via produce and poultry; 9% thallium, mostly via produce; 9% organics, mostly PCBs via foods)

As footnoted above, the reference area RME HI value is related primarily to thallium (77%), mostly via produce ingestion. By contrast, the CTE reference area HI is a function of a more diverse group of analytes (21% thallium, mostly via produce; 17% arsenic via produce and poultry; 13% manganese via produce and milk; and 11% zinc). The operational area RME child HI for the Rural-Residential scenario is associated with the following analytes and pathways: (30% PCBs, via milk and beef; 16% thallium, mostly via produce; 15% arsenic via produce and poultry; 13% manganese via produce and milk; and 11% zinc). The operational area CTE child HI for the Rural-Residential scenario is a function of arsenic via produce and poultry (17%), thallium via produce (14%), 13% PCBs, and 10% zinc.

As discussed in Section 5.5.2, an HQ does not reflect a probability of an adverse effect, but rather a level of concern above a threshold. Therefore, rather than showing an incremental value as was done for cancer risk and radiation dose, the contribution of background concentrations is shown as the ratio of background hazard to total hazard. The ratios of reference area RME background HI (6.0) to total RME HI, and of reference area CTE background HI (2.1), are shown in Tables 5-30a and 5-30b, respectively. A small ratio is indicative of an HI that is primarily related to specific contaminants at a remediated waste site rather than baseline operational area conditions. There are only a few remediated waste sites where these ratios are below 0.5.

It is important to note that, as discussed in the introductory paragraphs of Section 5.7, these HI results do not include contributions from fish ingestion or domestic uses of groundwater. RME risks for these calculations are shown relative to the range of risks calculated for the remediated waste sites in Table 5-24 and are presented in detail in subsequent sections of this risk assessment.

The HI values are calculated as the sum of chemical-specific HQs, as described in Section 5.5.2. However, the critical effects related to the chronic reference doses for important chemicals are not necessarily additive. For example, while the critical effects for both PCBs and mercury include effects on the immune system, the critical effects for arsenic relate to skin disorders and possible vascular complications. For oral exposure to thallium, the critical effect underlying the oral RfD is an increased blood level of the enzymes lactate dehydrogenase and serum glutamic oxaloacetic transaminase, the latter of which is used as a marker of liver function.

5.7.3 CTUIR Scenario Results

The CTUIR exposure scenario encompasses potential exposures related to surface and subsurface soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in potentially affected areas. This scenario emphasizes exposures related to ingestion of Columbia River fish, as well as activities including the use of native plants and hunting of game animals. Risks related to fish ingestion and

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groundwater use are addressed in Sections 5.7.7 and 5.8, respectively. The scenario is more fully described in the CSM (Section 2.0) and in Section 5.2, and is summarized in Table 5-1.

The CTUIR scenario defined in Harris and Harper (2004) generally presents only a single set of exposure parameter values. Therefore, differentiation of RME and CTE results are not presented for this scenario. Risks related to two variations of the CTUIR scenario are presented. In the “local area only” variation, exposure via ingestion of garden produce and meat from penned cattle is assessed. In the “local and broad areas” variation, ingestion of gathered wild plants is assessed using the RCBRA upland plant data collected under the SAP (DOE/RL-2005-42). Ingestion of game meat, with contaminant concentrations modeled from the RCBRA upland area soil data, is substituted for penned cattle.

Cancer Risk

The range of total cancer risk results for the “local area only” CTUIR scenario (radionuclides + chemicals) is from $1E-03$ to $>1E-02$, with a reference area background cancer risk value of $8E-03$. Calculated risk values above $1E-02$ are defined simply as $>1E-02$ because this represents the upper boundary of the applicability of EPA’s chemical slope factor models. The operational area (no excavation) cancer risk is $>1E-02$. “Local area only” CTUIR total cancer risk results are tabulated by waste site, according to magnitude, in Table 5-31a. Per EPA guidance (EPA 1989), cancer risk results are displayed with only one significant figure.

For the “local and broad areas” variation of the CTUIR scenario, the range of total risk results were essentially identical to the operational area baseline. All results were $>1E-02$ and are presented in this manner in Table 5-31b.

The range of “local area” CTUIR scenario total cancer risk values (radionuclides + chemicals) for all 163 waste sites, relative to operational area and reference area background risks, is shown in Figure 5-10. The analogous total cancer risk values for the “local and broad areas” CTUIR scenario are shown in Figure 5-11. Although the risk results have not been censored in these figures, allowing for inspection of the range of calculated values, differences in the values above $1E-02$ are not intrinsically meaningful. As indicated in the figure legend, total cancer risk values for reference area background and operational area (no excavation) cancer risk are shown as points of reference.

The particular exposure pathways and analytes contributing to total cancer risks at the remediated waste sites vary most for sites with calculated risks highest above the operational area baseline value. The following is a breakdown of the “local area only” total cancer risks for remediated waste sites with the highest calculated risk values. Because the risk results for the “local and broad areas” are equivalent to the operational area baseline, they are not differentiated by waste site.

300-10	(99% arsenic via produce ingestion)
100-F-37	(99% arsenic via produce ingestion)
100-F-35	(70% Sr-90; 12% arsenic - both via produce ingestion)

316-5	(53% Cs-137, mostly via external; 27% U-238 via produce and other pathways; 14% arsenic via produce)
316-1	(93% arsenic via produce ingestion)
316-2	(67% arsenic via produce ingestion; 27% iso-uranium)
116-DR-1&2	(53% Sr-90; 44% arsenic - both via produce ingestion)
100-H-21	(98% arsenic via produce ingestion)

For the “local area only” calculation, both reference area and operational area background cancer risk for the CTUIR scenario is associated primarily with arsenic (about 95%) via produce ingestion. For the “local and broad areas” variation, exposure to arsenic and organic chemicals via ingestion of native plants was the main contributor to cancer risk. The reference area background cancer risk is associated with the following analytes: 32% arsenic, 31% PCBs, 10% delta-BHC, 9% heptachlor, and 6% strontium-90. The operational area cancer risk is associated with the following analytes: 41% dieldrin, 20% PCBs, 12% arsenic, and 8% aldrin.

The ILCR for radionuclides and chemicals, calculated as total cancer risk for each waste site minus the reference area background cancer risk of risk of $8E-03$, is shown in Tables 5-32a and 5-32b for the “local area only” and “local and broad areas” applications of the CTUIR scenario. The range of CTUIR scenario ILCR values (radionuclides + chemicals) can be seen graphically as the difference between waste site risks and the reference area background risk in Figures 5-10 and 5-11. The “local area only” ILCR for the majority of remediated waste sites is in the range of $1E-03$ to $5E-03$, with seven sites having ILCR values above $1E-02$. “Local and broad areas” ILCR values were all above $1E-02$, primarily as a function of exposure to dieldrin via ingestion of native plants.

It is important to note that, as discussed in the introductory paragraphs of Section 5.7, these cancer risk results do not include contributions from fish ingestion, certain naturally occurring radionuclides in soil, or domestic uses of groundwater. Cancer risks for these calculations are shown relative to the range of risks calculated for the remediated waste sites in Table 5-24 and are presented in detail in subsequent sections of this risk assessment.

Radiation Dose

The range of total radiation dose results for the “local area only” CTUIR scenario (radionuclides + chemicals) is from 2.4 to 620 mrem/yr, with a reference area background dose value of 4.8 mrem/yr. The operational area (no excavation) radiation dose is 5.4 mrem/yr. “Local area only” CTUIR total radiation dose results are tabulated by waste site, according to magnitude, in Table 5-33a. For the “local and broad areas” variation of the CTUIR scenario, the range of total radiation dose results is from 41 to 360 mrem/yr, with a reference area background dose value of 180 mrem/yr. “Local area only” CTUIR total radiation dose results are tabulated by waste site, according to magnitude, in Table 5-33b. The operational area (no excavation) radiation dose for the “local and broad areas” calculation is 43 mrem/yr. The higher dose rate calculated for the Recreational Area is due to a higher calculated UCL for americium-241 in soil in this area.

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The range of “local area” CTUIR scenario total radiation dose values for all 163 waste sites, relative to operational area and reference area background doses, is shown in Figure 5-12. The analogous total radiation dose values for the “local and broad areas” CTUIR scenario are shown in Figure 5-13. As indicated in the figure legend, total radiation dose values for reference area background and operational area (no excavation) radiation dose are shown as points of reference.

The particular exposure pathways and analytes contributing to total radiation dose at the remediated waste sites vary most for sites with calculated doses highest above the operational area baseline value. The following is a breakdown of the “local area only” total radiation dose for remediated waste sites with the highest calculated dose values.

100-F-35	(98% Sr-90 via produce ingestion)
316-5	(52% Cs-137, mostly via external and produce ingestion; 40% U-238 via produce and other pathways)
116-DR-1&2	(98% Sr-90 via produce)
316-2	(88% isotopic uranium via produce and external)
116-F-6	(97% Sr-90 via produce)
116-F-14	(57% Eu-152 via external; 16% Cs-137, mostly via external and produce)
118-DR-2:2	(88% Sr-90 via produce)
316-1	(93% isotopic uranium via produce and external)

The following is a breakdown of the “local and broad areas” total radiation dose for remediated waste sites with the highest calculated dose values.

316-5	(58% Cs-137, via external; 26% U-238 mostly via external, chicken and soil ingestion)
316-2	(26% U-235, mostly via external; 21% Am-241 via native plant ingestion; 19% U-238 and 15% U-234, mostly via external, chicken and soil ingestion)
116-F-14	(52% Eu-152 via external; 22% Am-241, via native plant; 10% Cs-137 via external)
118-B-3	(50% Am-241, via native plant; 29% Eu-152 via external)
316-1	(54% Am-241, via native plant; 17% U-238 mostly via external, chicken and soil ingestion)
116-B-11	(56% Am-241, via native plant; 16% Eu-152 via external)
118-F-8:1	(56% Am-241, via native plant; 19% Eu-152 via external)
100-D-48:2	(59% Am-241, via native plant; 17% Cs-137 via external)

For the “local area only” calculation, both reference area and operational area background radiation dose for the CTUIR scenario is associated primarily with strontium-90 and carbon-14 via produce ingestion, although exposure to europium isotopes via external irradiation is also significant (26%) for the operational area. For the “local and broad areas” variation, exposure to americium-241 via ingestion of native plants was the main contributor (75% to 80%) to radiation dose.

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The incremental radiation dose for radionuclides, calculated as total radiation dose for each waste site minus the reference area background radiation dose, is shown for the “local area only” and “local and broad areas” applications of the CTUIR scenario in Tables 5-34a and 5-34b, respectively. The “local area only” incremental radiation dose for the majority of remediated waste sites is in the range of 0.5 to 20 mrem/yr, with six sites having incremental dose values above 100 mrem/yr. “Local and broad areas” incremental radiation dose values were all zero with the exception of 316-5 (200 mrem/yr), due to the high reference area dose calculated from americium-241.

It is important to note that, as discussed in the introductory paragraphs of Section 5.7, these radiation dose results do not include contributions from fish ingestion, certain naturally occurring radionuclides in soil, or domestic uses of groundwater. Radiation doses for these calculations are shown relative to the range of doses calculated for the remediated waste sites in Table 5-24 and are presented in detail in subsequent sections of this risk assessment.

Chemical Hazard

As described in Section 5.6.2, chemical hazard is calculated separately for adult and child receptors. For the CTUIR scenario, child HIs generally exceed those for adults by a factor of approximately 5% or less. Because they are consistently higher, only child HI values are presented here.

The range of total HI results for the “local area only” CTUIR scenario (radionuclides + chemicals) is from 30 to 700, with a single outlier (100-K-33) of about 3,600. The reference area background HI value is 500. The operational area (no excavation) HI is 90. CTUIR child HI results for the “local area only” application of the CTUIR scenario are tabulated by waste site, according to magnitude, in Table 5-35a. As described for the Rural-Residential HI, the high reference area HI of 500 is related to an elevated UCL for thallium in reference area soil. Because the majority of EPA reference dose toxicity criteria are presented with only one significant figure, HI results are also displayed with only one significant figure.

For the “local and broad areas” variation of the CTUIR scenario, the range of total HI results were essentially identical to the operational area baseline, which is 270. The reference area background HI for the “local and broad areas” variation of the CTUIR scenario is 220. CTUIR child HI results for the “local and broad areas” application of the CTUIR scenario are tabulated by waste site, according to magnitude, in Table 5-35b.

The range of “local area” CTUIR scenario child HI values for all 163 waste sites, relative to operational area and reference area background HI, is shown in Figure 5-14. The analogous child HI values for the “local and broad areas” CTUIR scenario are shown in Figure 5-15. As indicated in the figure legend, total radiation dose values for reference area background and operational area (no excavation) radiation dose are shown as points of reference.

The particular exposure pathways and analytes contributing to total HI values at the remediated waste sites vary most for sites with calculated risks highest above the operational area baseline value. The following is a breakdown of the “local area only” total HIs for remediated waste sites

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with the highest calculated HI values. Because the HI results for the “local and broad areas” are practically equivalent to the operational area baseline, they are not differentiated by waste site. At 100-K-33, where the calculated HI was 3,600, it should be noted that the UCL value for mercury in waste site soil (330 mg/kg) is about a factor of ten greater than the mean.

100-K-33	(97% mercury via produce ingestion)
128-C-1	(87% copper; 6% thallium, via produce ingestion)
100-K-30	(83% mercury; 6% thallium, via produce ingestion)
100-K-32	(84% mercury; 8% thallium, via produce ingestion)
300-10	(90% arsenic via produce ingestion)
100-K-31	(46% mercury; 25% cadmium; 16% thallium, via produce ingestion)
316-2	(70% thallium; 18% arsenic, via produce ingestion)
316-1	(57% thallium; 30% arsenic, via produce ingestion)

As mentioned above, the reference area HI value for the “local area only” calculation is related primarily to thallium (91%), via produce ingestion. By contrast, the operational area HI for the “local area only” variation has a 50% contribution from thallium via produce ingestion, the remainder relating to arsenic (25%) and manganese (15%), also via produce. In the “local and broad areas” calculation, PCBs contribute about 75% to 80% of the operational and reference area HIs via ingestion of native plants.

As discussed in Section 5.5.2, an HQ does not reflect a probability of an adverse effect, but rather a level of concern above a threshold. Therefore, rather than showing an incremental value as was done for cancer risk and radiation dose, the contribution of background concentrations is shown as the ratio of background hazard to total hazard. The ratios of reference area background HI to total HI are shown in Tables 5-36a and 5-36b for the “local area only” and “local and broad areas” applications of the CTUIR scenario, respectively. A small ratio is indicative of an HI that is primarily related to specific contaminants at a remediated waste site rather than baseline operational area conditions. In only one case (the local area only HI for 100-K-33) is any ratio are below 0.7.

It is important to note that, as discussed in the introductory paragraphs of Section 5.7, these HI results do not include contributions from fish ingestion or domestic uses of groundwater. RME risks for these calculations are shown relative to the range of risks calculated for the remediated waste sites in Table 5-24 and are presented in detail in subsequent sections of this risk assessment.

As noted in the discussion of Rural-Residential HI results, the critical effects related to the chronic reference doses for important chemicals are not necessarily additive. For example, while the critical effects for both PCBs and mercury include effects on the immune system, the critical effects for arsenic relate to skin disorders and possible vascular complications. For oral exposure to thallium, the critical effect underlying the oral RfD is an increased blood level of the enzymes lactate dehydrogenase and serum glutamic oxaloacetic transaminase, the latter of which is used as a marker of liver function.

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5.7.4 Resident Monument Worker Scenario Results

The Resident Monument Worker exposure scenario encompasses potential residential exposures related to surface and subsurface soils around remediated waste sites, as well as potential occupational exposures across broad regions of the 100 Area and 300 Area. It focuses on direct contact exposure pathways and does not encompass the raising of produce or animals nor the ingestion of Columbia River fish. With respect to exposure pathways, this scenario is similar to the traditional urban/suburban residential scenario used by various EPA regions to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. It may be viewed as akin to the Rural-Residential scenario, minus the ingestion of foodstuffs. The scenario is more fully described in the CSM (Section 2.0) and in Section 5.2, and is summarized in Table 5-1.

Cancer Risk

The range of RME total cancer risk results for the Resident Monument Worker scenario (radionuclides + chemicals) is from $3\text{E-}05$ to $3\text{E-}03$, with a reference area RME background cancer risk value of $3\text{E-}05$. The operational area (no excavation) RME cancer risk is slightly higher than background, being $4\text{E-}05$. The range of the CTE total cancer risk results is smaller, being from $5\text{E-}06$ to $3\text{E-}05$, with a reference area CTE background cancer risk value of $7\text{E-}06$. The operational area (no excavation) CTE cancer risk is $8\text{E-}06$. Resident Monument Worker RME and CTE risk results are tabulated by waste site, according to magnitude, in Tables 5-37a and 5-37b, respectively. Per EPA guidance (EPA 1989), cancer risk results are displayed with only one significant figure.

The range of Resident Monument Worker RME and CTE total cancer risk values (radionuclides + chemicals) for all 163 waste sites, relative to operational area and reference area background risks, is shown in Figure 5-16. As indicated in the figure legend, RME and CTE total cancer risk values for reference area background and operational area (no excavation) cancer risk are shown as points of reference. The waste sites plotted along the x-axis are ranked according to the magnitude of the CTE total cancer risk. The variability shown in the RME results (i.e., the spread of the calculated risks above a theoretical line along the lowest calculated RME values) is a function of the protocols used in calculating the UCL exposure point concentrations in soil.

The Resident Monument Worker scenario includes a residential component that is assessed on a "local area" scale with a separate calculation for each remediated waste site. Additionally, the occupational component of the scenario includes exposure over a "broad area," represented by upland surface soils. Risks related to these two components of the Resident Monument Worker are calculated separately, with appropriate time allocation of for each part, and then summed.

The particular exposure pathways and analytes contributing to total cancer risks at the remediated waste sites vary most for sites with calculated risks highest above the operational area baseline value. As site risks approach the operational area baseline, the majority of the calculated risk is a function of the same baseline conditions in surface soil. The following is a breakdown of the RME total cancer risks for remediated waste sites with the highest calculated RME risk values.

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316-5	(82% Cs-137 via external; 15% U-238, mostly via external)
116-F-14	(72% Eu-152; 13% Cs-137; 12% Co-60, all via external)
316-2	(46% U-235; 25% Co-60; 17% U-238, mostly via external)
118-B-3	(75% Eu-152; 12% Cs-137, all via external)
116-B-11	(50% Eu-154; 16% Eu-152; 8% Cs-137; 8% Co-60, all via external)
118-F-8:1	(65% Eu-152; 22% Cs-137; 5% Eu-154, all via external)
116-B-1	(76% Eu-152; 9% Cs-137; 6% Eu-154, all via external)
100-D-48:2	(56% Cs-137; 30% Eu-152, all via external)

The following is a breakdown of the CTE total cancer risks for remediated waste sites with the highest calculated CTE risk values. Only three waste sites are shown, as total cancer risks for the remaining sites are generally at or below 1E-05.

316-5	(34% U-238; 31% Cs-137; 18% U-235, mostly via external)
116-F-14	(67% Eu-152; 16% Eu-154; 10% Co-60, all via external)
316-2	(31% U-238; 28% U-235; 18% Co-60; 13% Eu-154 and Eu-152 combined, mostly via external)

Both reference area and operational area background cancer risks for the Resident Monument Worker scenario are associated primarily with external irradiation via short-lived radioisotopes including isotopes of europium, cobalt-60, and cesium-137. Because the half-lives of these isotopes are on the order of 30 years or less and they are not naturally occurring, background risk levels related to these isotopes in soil will decrease relatively quickly over time. Arsenic, via soil ingestion, also contributes approximately 15% to RME risks in the reference and operational areas.

The ILCR for radionuclides and chemicals, calculated as total cancer risk for each waste site minus the reference area background cancer risk of risks of 3E-05 (RME) and 6E-06 (CTE), is shown in Tables 5-38a and 5-38b. The range of Resident Monument Worker scenario ILCR values (radionuclides + chemicals) can be seen graphically as the difference between waste site risks and the reference area background risk in Figure 5-16.

The RME ILCR for the majority of remediated waste sites is in the range of 1E-05 to 1E-04, with two sites (316-5 and 116-F-14) having RME ILCR values at or above 1E-03. CTE ILCR values are all within or below EPA's target risk range of 10^{-6} to 10^{-4} described in the NCP.

It is important to note that, as discussed in the introductory paragraphs of Section 5.7, these cancer risk results do not include contributions from certain naturally occurring radionuclides in soil or domestic uses of groundwater. RME risks for these calculations are shown relative to the range of risks calculated for the remediated waste sites in Table 5-24 and are presented in detail in subsequent sections of this risk assessment.

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Radiation Dose

The range of RME total radiation dose results for the Resident Monument Worker scenario is from 1.3 to 150 mrem/yr, with a reference area RME background dose value of 1.5 mrem/yr. The operational area (no excavation) RME dose is slightly higher than background, being 2.3 mrem/yr. The range of the CTE total radiation dose results is smaller, being from 0.85 to 7.1 mrem/yr, with a reference area CTE background dose value of 1.2 mrem/yr. The operational area (no excavation) CTE radiation dose is 1.5 mrem/yr. Resident Monument Worker RME and CTE radiation dose results are tabulated by waste site, according to magnitude, in Tables 5-39a and 5-39b.

The range of Resident Monument Worker RME and CTE total radiation dose values for all 163 waste sites, relative to operational area and reference area background dose, is shown in Figure 5-17. As indicated in the figure legend, RME and CTE total radiation dose values for reference area background and operational area (no excavation) radiation dose are shown as points of reference. The waste sites plotted along the x-axis are ranked according to the magnitude of the CTE total radiation dose. The variability shown in the RME results (i.e., the spread of the calculated doses above a theoretical line along the lowest calculated RME values) is a function of the protocols used in calculating the UCL exposure point concentrations in soil.

As discussed for the Resident Monument Worker cancer risk results, this scenario includes a residential component that is assessed on a "local area" scale and an occupational component of the scenario assessed over a "broad area." Risks related to these two components of the Resident Monument Worker are calculated separately, with appropriate time allocation of for each part, and then summed.

The particular exposure pathways and analytes contributing to total radiation dose at the remediated waste sites vary most for sites with calculated doses highest above the operational area baseline value. As site doses approach the operational area baseline, the majority of the calculated dose is a function of the same baseline conditions in surface soil. The following is a breakdown of the RME total radiation dose for remediated waste sites with the highest calculated RME dose values. At waste site 118-B-3, the RME total radiation dose is about 15 mrem/yr.

316-5	(80% Cs-137 via external; 19% U-238 and U-235, mostly via external)
116-F-14	(71% Eu-152 via external; 13% Cs-137 and 12% Co-60, both via external;)
316-2	(45% U-235 via external; 23% Co-60; 20% U-238 mostly via external)
118-B-3	(76% Eu-152; 13% Cs-137, both via external)

The following is a breakdown of the CTE total radiation dose for remediated waste sites with the highest calculated CTE dose values. The calculated CTE dose for site 316-2 is below 5 mrem/yr.

316-5	(39% U-238, mostly via external; 28% Cs-137 via external; 17% U-235 via external)
116-F-14	(67% Eu-152; 16% Eu-154, both via external)
316-2	(64% U-238 and U-235, mostly via external)

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Both reference area and operational area background radiation dose for the Resident Monument Worker scenario are associated primarily with external irradiation via short-lived radioisotopes including isotopes of europium, cobalt-60, and cesium-137. Because the half-lives of these isotopes are on the order of 30 years or less and they are not naturally occurring, background dose levels related to these isotopes in soil will decrease relatively quickly over time.

The incremental radiation dose for radionuclides, calculated as total radiation dose for each waste site minus the reference area background radiation dose of 1.5 mrem/yr (RME) and 1.2 mrem/yr (CTE) is shown in Tables 5-40a and 5-40b, respectively. The range of incremental dose values can be seen graphically as the difference between waste site doses and the reference area background dose in Figure 5-17. With the exception of the RME incremental dose at three waste sites (316-5, 116-F-14, and 316-2), RME and CTE incremental dose values for the Resident Monument Worker scenario are below 15 mrem/yr.

As was stated for the cancer risk results, the radiation dose results do not include contributions from certain naturally occurring radionuclides in soil or domestic uses of groundwater. RME doses for these calculations are shown relative to the range of doses calculated for the remediated waste sites in Table 5-24 and are presented in detail in subsequent sections of this risk assessment.

Chemical Hazard

As described in Section 5.6.2, chemical hazard is calculated separately for adult and child receptors. Because only adult receptors are evaluated in the Resident Monument Worker scenario, the HI results described in this section pertain to adults.

The range of RME total child HI results for the Resident Monument Worker scenario is from 0.09 to 0.7, with a reference area RME background HI value of 0.2. The operational area (no excavation) RME HI is also 0.2. The range of the CTE total HI results is from 0.03 to 0.1, with a reference area CTE background HI value of 0.04. The operational area (no excavation) CTE HI is 0.05. Resident Monument Worker RME and CTE HI results are tabulated by waste site, according to magnitude, in Tables 5-41a and 5-41b. Because the majority of EPA reference dose toxicity criteria are presented with only one significant figure, HI results are also displayed with only one significant figure.

The range of Resident Monument Worker RME and CTE hazard index for all 163 waste sites, relative to operational area and reference area background HI, is shown in Figure 5-18. As indicated in the figure legend, RME and CTE total HI values for reference area background and operational area (no excavation) HI are shown as points of reference. The waste sites plotted along the x-axis are ranked according to the magnitude of the CTE total HI. The variability shown in the RME results (i.e., the spread of the calculated HI values above a theoretical line along the lowest calculated RME values) is a function of the protocols used in calculating the UCL exposure point.

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The particular exposure pathways and analytes contributing to total HI values at the remediated waste sites vary most for sites with calculated HIs highest above the operational area baseline value. As site HIs approach the operational area baseline, the majority of the calculated HI is a function of the same baseline conditions in surface soil. The following is a breakdown of the RME total HI values for remediated waste sites with the highest calculated RME values. At remediated waste site 128-C-1, the calculated total RME HI is just 0.4. Because the HI values are low relative to a threshold of 1.0, a breakdown for the CTE results is not presented.

100-K-33	(80% mercury, via soil ingestion; 10% PCBs via soil ingestion and dermal)
600-23	(90% PCBs via soil ingestion and dermal)
618-4	(85% PCBs via soil ingestion and dermal)
128-C-1	63% copper via soil ingestion; 18% PCBs via soil ingestion and dermal; 6% arsenic via soil ingestion

The reference area RME HI value is related primarily to thallium (75%), mostly via soil ingestion, with arsenic contributing another 8%. The CTE reference area HI is a function of three analytes (25% aluminum; 24% arsenic; 21% thallium - all mostly via soil ingestion). The operational area RME HI for the Resident Monument Worker scenario is associated with the following analytes and pathways: 55% PCBs, via soil ingestion and dermal; and 16% arsenic, mostly via soil ingestion. The operational area CTE HI is a function of PCBs (34%), arsenic (24%), and aluminum and thallium (26% combined).

As discussed in Section 5.5.2, an HQ does not reflect a probability of an adverse effect, but rather a level of concern above a threshold. Therefore, rather than showing an incremental value as was done for cancer risk and radiation dose, the contribution of background concentrations is shown as the ratio of background hazard to total hazard. The ratios of reference area RME background HI (0.25) to total RME HI, and of reference area CTE background HI (0.037), are shown in Tables 5-42a and 5-42b, respectively. A small ratio is indicative of an HI that is primarily related to specific contaminants at a remediated waste site rather than baseline operational area conditions. There are fewer than 10 remediated waste sites where these ratios are below 0.5.

It is important to note that, as discussed in the introductory paragraphs of Section 5.7, these HI results do not include contributions from domestic uses of groundwater. RME risks for these calculations are shown relative to the range of risks calculated for the remediated waste sites in Table 5-24 and are presented in detail in subsequent sections of this risk assessment.

The HI values are calculated as the sum of chemical-specific HQs, as described in Section 5.5.2. However, the critical effects related to the chronic reference doses for important chemicals are not necessarily additive. For example, while the critical effects for both PCBs and mercury include effects on the immune system, the critical effects for arsenic relate to skin disorders and possible vascular complications. For oral exposure to thallium, the critical effect underlying the oral RfD is an increased blood level of the enzymes lactate dehydrogenase and serum glutamic oxaloacetic transaminase, the latter of which is used as a marker of liver function. The oral RfD for copper is a provisional value related to minimum dietary requirements, copper absorption is

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generally well regulated by homeostatic processes except in individuals with certain medical conditions.

5.7.5 Industrial/Commercial Worker Scenario Results

The Industrial/Commercial Worker exposure scenario encompasses potential occupational exposures related to surface and subsurface soils around remediated waste sites. Like the Resident Monument Worker scenario, it focuses on direct contact exposure pathways and does not encompass the raising of produce or animals, nor the ingestion of Columbia River fish. It is similar to the Industrial/Commercial Worker scenario used by various EPA regions to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. The scenario is more fully described in the CSM (Section 2.0) and in Section 5.2, and is summarized in Table 5-1.

Cancer Risk

The range of RME total cancer risk results for the Industrial/Commercial Worker scenario (radionuclides + chemicals) is from $3E-06$ to $2E-03$, with a reference area RME background cancer risk value of $1E-05$. The operational area (no excavation) RME cancer risk is slightly higher than background, being $2E-05$. The range of the CTE total cancer risk results is from $5E-07$ to $2E-05$, with a reference area CTE background cancer risk value of $3E-06$. The operational area (no excavation) CTE cancer risk is $4E-06$. Industrial/Commercial RME and CTE risk results are tabulated by waste site, according to magnitude, in Tables 5-43a and 5-43b, respectively. Per EPA guidance (EPA 1989), cancer risk results are displayed with only one significant figure.

The range of Industrial/Commercial RME and CTE total cancer risk values (radionuclides + chemicals) for all 163 waste sites, relative to operational area and reference area background risks, is shown in Figure 5-19. As indicated in the figure legend, RME and CTE total cancer risk values for reference area background and operational area (no excavation) cancer risk are shown as points of reference. The waste sites plotted along the x-axis are ranked according to the magnitude of the CTE total cancer risk. The variability shown in the RME results (i.e., the spread of the calculated risks above a theoretical line along the lowest calculated RME values) is a function of the protocols used in calculating the UCL exposure point concentrations in soil.

The particular exposure pathways and analytes contributing to total cancer risks at the remediated waste sites vary most for sites with calculated risks highest above the operational area baseline value. As site risks approach the operational area baseline, the majority of the calculated risk is a function of the same baseline conditions in surface soil. The waste sites, analytes, and pathways are very similar to those described for the Resident Monument Worker scenario, which employs a similar exposure model. The following is a breakdown of the RME total cancer risks for remediated waste sites with the highest calculated RME risk values.

316-5	(84% Cs-137 via external; 14% U-238, mostly via external)
116-F-14	(72% Eu-152; 13% Cs-137; 11% Co-60, all via external)

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316-2	(49% U-235; 26% Co-60; 17% U-238, mostly via external)
118-B-3	(79% Eu-152; 13% Cs-137, both via external)
116-B-11	(54% Eu-154; 22% Eu-152; 12% Cs-137; 11% Co-60, all via external)
118-F-8:1	(69% Eu-152; 24% Cs-137, both via external)
116-B-1	(80% Eu-152; 9% Cs-137; 3% Eu-154, all via external)
100-D-48:2	(63% Cs-137; 29% Eu-152, both via external)

The following is a breakdown of the CTE total cancer risks for remediated waste sites with the highest calculated CTE risk values. Only three waste sites are shown, as total cancer risks for the remaining sites are generally at or below $1E-05$.

316-5	(38% U-238; 36% Cs-137; 21% U-235, all via external)
116-F-14	(72% Eu-152; 14% Eu-154; 8% Co-60, all via external)
316-2	(38% U-238; 36% U-235; 18% Co-60, mostly via external)

Both reference area and operational area background cancer risks for the Industrial/Commercial Worker scenario are associated primarily with external irradiation via short-lived radioisotopes including isotopes of europium, cobalt-60, and cesium-137. Because the half-lives of these isotopes are on the order of 30 years or less and they are not naturally occurring, background risk levels related to these isotopes in soil will decrease relatively quickly over time. Arsenic, via soil ingestion, also contributes approximately 15% to RME risks in the reference and operational areas.

The ILCR for radionuclides and chemicals, calculated as total cancer risk for each waste site minus the reference area background cancer risk of risks of $1E-05$ (RME) and $3E-06$ (CTE), are shown in Tables 5-44a and 5-44b. The range of Industrial/Commercial scenario ILCR values (radionuclides + chemicals) can be seen graphically as the difference between waste site risks and the reference area background risk in Figure 5-19.

The RME ILCR for the majority of remediated waste sites is in the range of $1E-06$ to $1E-04$, with two sites (316-5 and 116-F-14) having RME ILCR values at or above $1E-03$. CTE ILCR values are all within or below EPA's target risk range of 10^{-6} to 10^{-4} described in the NCP.

It is important to note that, as discussed in the introductory paragraphs of Section 5.7, these cancer risk results do not include contributions from certain naturally occurring radionuclides in soil. RME risks for these calculations are shown relative to the range of risks calculated for the remediated waste sites in Table 5-24 and are presented in detail in subsequent sections of this risk assessment.

Radiation Dose

The range of RME total radiation dose results for the Industrial/Commercial scenario is from 0.19 to 120 mrem/yr, with a reference area RME background dose value of 0.66 mrem/yr. The operational area (no excavation) RME dose is slightly higher than background, being

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1.0 mrem/yr. The range of the CTE total radiation dose results is from 0.093 to 5.7 mrem/yr, with a reference area CTE background dose value of 0.59 mrem/yr. The operational area (no excavation) CTE radiation dose is 0.75 mrem/yr. Industrial/Commercial RME and CTE radiation dose results are tabulated by waste site, according to magnitude, in Tables 5-45a and 5-45b.

The range of Industrial/Commercial RME and CTE total radiation dose values for all 163 waste sites, relative to operational area and reference area background dose, is shown in Figure 5-20. As indicated in the figure legend, RME and CTE total radiation dose values for reference area background and operational area (no excavation) radiation dose are shown as points of reference. The waste sites plotted along the x-axis are ranked according to the magnitude of the CTE total radiation dose. The variability shown in the RME results (i.e., the spread of the calculated doses above a theoretical line along the lowest calculated RME values) is a function of the protocols used in calculating the UCL exposure point concentrations in soil.

The particular exposure pathways and analytes contributing to total radiation dose at the remediated waste sites vary most for sites with calculated doses highest above the operational area baseline value. As site doses approach the operational area baseline, the majority of the calculated dose is a function of the same baseline conditions in surface soil. The following is a breakdown of the RME total radiation dose for remediated waste sites with the highest calculated RME dose values. At waste site 118-B-3, the RME total radiation dose is about 12 mrem/yr.

316-5	(81% Cs-137 via external; 17% U-238, mostly via external)
116-F-14	(71% Eu-152; 13% Cs-137; 12% Co-60, all via external)
316-2	(48% U-235 via external; 24% Co-60 via external; 20% U-238 mostly via external)
118-B-3	(79% Eu-152; 13% Cs-137, both via external)

The following is a breakdown of the CTE total radiation dose for remediated waste sites with the highest calculated CTE dose values. The calculated CTE doses for waste site ranked lower than 316-2 is below 2 mrem/yr.

316-5	(44% U-238, mostly via external; 32% Cs-137; 20% U-235, both via external)
116-F-14	(72% Eu-152; 13% Eu-154, both via external)
316-2	(77% U-238 and U-235, mostly via external)

Both reference area and operational area background radiation dose for the Industrial/Commercial Worker scenario are associated primarily with external irradiation via short-lived radioisotopes including isotopes of europium, cobalt-60, and cesium-137. Because the half-lives of these isotopes are on the order of 30 years or less and they are not naturally occurring, background dose levels related to these isotopes in soil will decrease relatively quickly over time.

The incremental radiation dose for radionuclides, calculated as total radiation dose for each waste site minus the reference area background radiation dose of 1.5 mrem/yr (RME) and 1.2 mrem/yr (CTE), is shown in Tables 5-46a and 5-46b, respectively. The range of incremental

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dose values can be seen graphically as the difference between waste site doses and the reference area background dose in Figure 5-20. With the exception of the RME incremental dose at three waste sites (316-5, 116-F-14, and 316-2), RME and CTE incremental dose values for the Industrial/Commercial Worker scenario are below 15 mrem/yr.

As was stated for the cancer risk results, the radiation dose results do not include contributions from certain naturally occurring radionuclides in soil. RME doses for these calculations are shown relative to the range of doses calculated for the remediated waste sites in Table 5-24 and are presented in detail in subsequent sections of this risk assessment.

Chemical Hazard

As described in Section 5.6.2, chemical hazard is calculated separately for adult and child receptors. Because only adult receptors are evaluated in the Industrial/Commercial Worker scenario, the HI results described in this section pertain to adults.

The range of RME total child HI results for the Industrial/Commercial Worker scenario is from 0.01 to 0.2, with a reference area RME background HI value of 0.04. The operational area (no excavation) RME HI is 0.07. The range of the CTE total HI results is from 0.005 to 0.04, with a reference area CTE background HI value of 0.01. The operational area (no excavation) CTE HI is also 0.01. Industrial/Commercial RME and CTE HI results are tabulated by waste site, according to magnitude, in Tables 5-47a and 5-47b. Because the majority of EPA reference dose toxicity criteria are presented with only one significant figure, HI results are also displayed with only one significant figure.

The range of Industrial/Commercial Worker RME and CTE hazard index for all 163 waste sites, relative to operational area and reference area background HI, is shown in Figure 5-21. As indicated in the figure legend, RME and CTE total HI values for reference area background and operational area (no excavation) HI are shown as points of reference. The waste sites plotted along the x-axis are ranked according to the magnitude of the CTE total HI. The variability shown in the RME results (i.e., the spread of the calculated HI values above a theoretical line along the lowest calculated RME values) is a function of the protocols used in calculating the UCL exposure point.

The particular exposure pathways and analytes contributing to total HI values at the remediated waste sites vary most for sites with calculated HIs highest above the operational area baseline value. As site HIs approach the operational area baseline, the majority of the calculated HI is a function of the same baseline conditions in surface soil. The following is a breakdown of the RME total HI values for remediated waste sites with the highest calculated RME values. At remediated waste site 128-C-1, the calculated total RME HI is below 0.2. Because the HI values are very low relative to a threshold of 1.0, a breakdown for the CTE results is not presented.

100-K-33	(87% mercury, via soil ingestion; 5% PCBs, via soil ingestion and dermal)
600-23	(90% PCBs via soil ingestion and dermal)
618-4	(89% PCBs via soil ingestion and dermal)

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128-C-1 (75% copper via soil ingestion; 11% PCBs via soil ingestion and dermal; 5% arsenic via soil ingestion)

The reference area RME HI value is related primarily to thallium (76%), mostly via soil ingestion, with aluminum (10%) and arsenic (7%) contributing most of the remainder. The CTE reference area HI is also a function of three analytes (27% aluminum; 25% arsenic; 24% thallium - all mostly via soil ingestion). The operational area RME HI for the Industrial/Commercial Worker scenario is associated with the following analytes and pathways: 50% PCBs, via soil ingestion and dermal; and 17% arsenic, mostly via ingestion. The operational area CTE HI is a function of PCBs (26%), arsenic (26%), and aluminum and thallium (31% combined).

As discussed in Section 5.5.2, an HQ does not reflect a probability of an adverse effect, but rather a level of concern above a threshold. Therefore, rather than showing an incremental value as was done for cancer risk and radiation dose, the contribution of background concentrations is shown as the ratio of background hazard to total hazard. The ratios of reference area RME background HI (0.075) to total RME HI, and of reference area CTE background HI (0.01), are shown in Tables 5-48a and 5-48b, respectively. A small ratio is indicative of an HI that is primarily related to specific contaminants at a remediated waste site rather than baseline operational area conditions. There are 10 or fewer remediated waste sites where these ratios are below 0.5.

The HI values are calculated as the sum of chemical-specific HQs, as described in Section 5.5.2. However, the critical effects related to the chronic reference doses for important chemicals are not necessarily additive. For example, while the critical effects for both PCBs and mercury include effects on the immune system, the critical effects for arsenic relate to skin disorders and possible vascular complications. For oral exposure to thallium, the critical effect underlying the oral RfD is an increased blood level of the enzymes lactate dehydrogenase and serum glutamic oxaloacetic transaminase, the latter of which is used as a marker of liver function. The oral RfD for copper is a provisional value related to minimum dietary requirements; copper absorption is generally well regulated by homeostatic processes except in individuals with certain medical conditions.

5.7.6 Recreational Scenario Results

The Recreational exposure scenario includes three separate recreational receptors: Casual User, Avid Angler, and Avid Hunter. Each scenario includes both an adult and child receptor, although in the hunting and fishing scenarios children are represented as 7 to 12 year olds rather than traditional 1 to 6 year olds because of the nature of these activities. The Recreational scenarios are more fully described in the CSM (Section 2.0) and in Section 5.2, and are summarized in Table 5-1.

As described in the introductory paragraphs of Section 5.7, risks related to fish ingestion and risks related to certain naturally occurring radionuclides are not incorporated in the Recreational scenario risks. Risk results for these calculations are presented in Sections 5.7.7 and 5.7.8,

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respectively. Therefore, the Avid Angler scenario risks described here include only exposures related to sediments and (for the inhalation pathway) and riparian soil. The Avid Angler risk results are presented for four exposure areas, differentiated by residual contaminant concentrations in sediments: the 300 Area, the 100/B-C Area, the 100-N Area, and the entire 100 Area in aggregate.

The Avid Hunter scenario results are presented both with and without inclusion of the game ingestion exposure pathway. In the absence of meat ingestion, this upland area exposure scenario approximates the exposure model for the Casual User scenario, which is applied to the riparian environment. In this way, risks related to casual use of both upland and riparian areas is evaluated.

Cancer Risk

The RME and CTE total risk results for the Avid Hunter and Casual User scenarios (radionuclides + chemicals) are shown in Table 5-49a. Cancer risks related to reference area background for these scenarios are provided in Table 5-49b. Cancer risks for sediment and riparian soil exposures for the Avid Angler scenario are shown for the four operational exposure areas and the reference area in Table 5-50. Per EPA guidance (EPA 1989), cancer risk results are displayed with only one significant figure.

Total cancer risk (radionuclides and chemicals) for the Casual User and Avid Hunter Recreational exposure scenarios is shown in Figure 5-22a. The background cancer risks (radionuclides and chemicals) for these scenarios, calculated using reference area soil data, are shown in Figure 5-22b. Cancer risks (radionuclides and chemicals) related to sediment and riparian soil exposures for the Avid Angler exposure scenario in all four near-shore operational exposure areas and the reference area are shown in Figure 5-23.

In the Avid Angler scenario, RME cancer risks across the four operational exposure areas ranged from $2E-06$ to $3E-05$, with a background RME risk of $4E-06$. CTE cancer risks ranged from $1E-07$ to $1E-06$, with a background CTE risk of $4E-06$. Cancer risks in each of the four operational exposure areas are related primarily to external irradiation from europium-152, cesium-137, and cobalt-60. Because these are short-lived anthropogenic isotopes, cancer risk related to sediment exposures for these radionuclides will decrease relatively quickly over time. To a lesser extent, exposure to arsenic (100 Area and 300 Area) and PAHs (100 Area) via soil ingestion contributes to cancer risks in these areas. In the reference area, cancer risk is related to the following analytes: 27% arsenic, 19% europium-154, 13% cobalt-60, 13% europium-152, and 10% uranium-235.

In the Avid Hunter scenario, RME and CTE cancer risks were $1E-04$ and $4E-06$, respectively. RME and CTE background risks were $3E-05$ and $2E-06$, respectively. operational area RME and CTE risks in the absence of the game ingestion pathway were just $3E-06$ and $2E-07$, respectively. RME cancer risks for the Avid Hunter scenario are attributable to PCBs (28%), arsenic (26%), benzo(a)pyrene (9%), and aldrin (8%) - all via game ingestion. However, CTE risks are nearly 50% attributable to arsenic, reflecting the fact that the UCL values for the organic chemicals are skewed to large values relative to the means. About 80% of the cancer

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risk for the Avid Hunter scenario absent the game ingestion pathway is attributable to isotopic europium, cesium-137, and cobalt-60 via external irradiation.

Casual User scenario RME and CTE cancer risk values in the operational area were $3E-06$ and $1E-07$, respectively. RME and CTE background risks were also $3E-06$ and $2E-07$, respectively. These results are approximately equivalent to those calculated for the Avid Hunter (without game ingestion) in the upland environment. operational area RME cancer risks are attributable to the following analytes and exposure pathways: 21% arsenic, 33% europium-152 and cesium-137 combined; and additional contributions from europium-154, cobalt-60, and PAHs. reference area RME cancer risks for the Casual User scenario are attributable as follows: 28% arsenic, 58% europium-154, cobalt-60, and europium-152 combined.

Radiation Dose

The RME and CTE total radiation dose results for the Avid Hunter and Casual User scenarios are shown in Table 5-51a. Radiation doses related to background for these scenarios are provided in Table 5-51b. Radiation doses for sediment and riparian soil exposures for the Avid Angler scenario are shown for the four operational exposure areas and the reference area in Table 5-52.

Total radiation dose for the Casual User and Avid Hunter Recreational exposure scenarios is shown in Figure 5-24a. The background radiation doses for these scenarios, calculated using reference area soil data, are shown in Figure 5-24b. Radiation doses related to sediment and riparian soil exposures for the Avid Angler Recreational exposure scenario in all four near-shore operational exposure areas and the reference area are shown in Figure 5-25.

RME radiation doses in the Avid Angler scenario for the four operational exposure areas ranged from 0.04 to 1.1 mrem/yr, with a background RME dose of 0.15 mrem/yr. CTE radiation doses ranged from 0.005 to 0.18 mrem/yr, with a background CTE dose of 0.023 mrem/yr. Total radiation dose results for the Avid Angler scenario are well below a threshold criterion of 15 mrem/yr above background levels.

In the Avid Hunter scenario, RME and CTE radiation dose were 0.27 and 0.054 mrem/yr, respectively. RME and CTE background doses were 0.08 and 0.02 mrem/yr, respectively. operational area RME and CTE risks in the absence of the game ingestion pathway were just 0.1 and 0.03 mrem/yr, respectively. Total radiation dose results for the Avid Hunter scenario are well below a threshold criterion of 15 mrem/yr above background levels.

Casual User scenario RME and CTE radiation dose values in the operational area were 0.1 and 0.01 mrem/yr, respectively. RME and CTE background risks were similar, being 0.09 and 0.01 mrem/yr, respectively. These results are approximately equivalent to those calculated for the Avid Hunter (without game ingestion) in the upland environment. Total radiation dose results for the Casual User scenario are well below a threshold criterion of 15 mrem/yr above background levels.

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As described in Section 5.6.2, chemical hazard is calculated separately for adult and child receptors. For the recreational scenarios, child HIs generally exceed those for adults. An exception is the Avid Hunter scenario, where child and adult values are approximately equal because exposure via the game ingestion pathway is shared between receptors. Because child HIs are either larger than or equal to the HI values for adults, only child HI values are presented here. Because the majority of EPA reference dose toxicity criteria are presented with only one significant figure, HI results are also displayed with only one significant figure.

The RME and CTE total child HI results for the Avid Hunter and Casual User scenarios are shown in Table 5-53a. Child HI values related to background for these scenarios are provided in Table 5-53b. Child HI values for sediment and riparian soil exposures for the Avid Angler scenario are shown for the four operational exposure areas and the reference area in Table 5-54.

Total child HI values for the Casual User and Avid Hunter Recreational exposure scenarios are shown in Figure 5-26a. The background HIs for these scenarios, calculated using reference area soil data, are shown in Figure 5-26b. Child HIs related to sediment and riparian soil exposures for the Avid Angler Recreational exposure scenario in all four near-shore operational exposure areas and the reference area are shown in Figure 5-27.

RME child HI values in the Avid Angler scenario for the four operational exposure areas ranged from 0.02 to 0.08, with a background RME dose of 0.04. CTE child HI values ranged from 0.002 to 0.1, with a background CTE HI of 0.004. Total HI results for the Avid Angler scenario are well below a threshold criterion of 1.0.

In the Avid Hunter scenario, RME and CTE child HI values were 3 and 0.5, respectively. RME and CTE background HIs were 4 and 0.3 mrem/yr, respectively. Operational area RME and CTE risks in the absence of the game ingestion pathway were just 0.03 and 0.003, respectively. The operational area RME HI is related primarily to PCBs (55%), and to a combination of thallium, zinc and mercury (25% combined) – all via ingestion of game meat. In the CTE calculation, only PCBs contribute only 35% of the total, an indication that the soil UCL values for these PCBs is skewed to a high value relative to the mean. In the reference area, Avid Hunter RME HI values relate to thallium (74%) and to zinc and mercury (11% combined). However, only 21% of the Avid Hunter CTE value for the reference area is attributable to thallium.

Casual User scenario RME and CTE HI values in the operational area were 0.03 and 0.002, respectively. RME and CTE background HIs were identical. These results are approximately equivalent to those calculated for the Avid Hunter (without game ingestion) in the upland environment. The child HI results for the Casual User scenario are well below a threshold criterion of 1.0.

For the Avid Hunter, the potential additivity of hazards for the key COPCs mentioned in the previous paragraphs is important for interpreting potential protective bias in the calculated HI values. Critical effects underlying the oral RfDs for both PCBs and inorganic forms of mercury include effects on the immune system, suggesting that additivity of hazard quotients for these chemicals is reasonable. Toxic effects related to zinc are rare and may be associated more with inhibition of copper absorption from the gastrointestinal tract (Klaassen 2001). The critical

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effect of thallium in the study underlying the oral RfD relates to possible impairment of liver function as measured by blood enzyme markers.

5.7.7 Fish Ingestion Risk Assessment Results

The fish ingestion exposure pathway is included in the CTUIR, Rural-Residential, and Avid Angler exposure scenarios. Risk calculations are performed for each of four exposure areas, as described in Section 5.3.15. Separate risk calculations are conducted using fish (sculpin) tissue samples collected from the 100-N Area and 300 Area because data analysis (see Section 4.4) indicates that concentrations of strontium-90 (100-N Area) and uranium (300 Area) are, respectively, higher in these than in other sampling locations. Fish tissue data collected under the 100-B/C Pilot investigation are also assessed separately. All 100 Area fish tissue data are also combined and evaluated collectively to assess potential risks related to fish ingestion across the entire operational area. The four exposure areas for evaluating fish ingestion are as follows:

1. 100 Area (inclusive of all 100 Area sculpin fish tissue data)
2. 300 Area
3. 100-N Area (data collected under the 100-NR-2 investigation)
4. 100-B/C Area (data collected under the 100-B/C Pilot investigation).

Risks are also computed using the reference area fish tissue samples. The ILCR and incremental dose, and the ratio of operational and reference area HIs, are also computed for the 100 Area and 300 Area fish ingestion risk calculations as described in a manner analogous to that described in Sections 5.7.2 through 5.7.6. The quantity of background fish tissue samples in the 100-B/C and 100-NR-2 data sets does not support these computations. Therefore, only "total" fish ingestion risk values are calculated for these exposure areas.

The risk assessment results for the fish ingestion exposure pathway are affected by a systematic problem related to elevated detection limits for organic chemicals in fish tissue, particularly for PAHs. Figures 5-6a through 5-6c show the range of detected values and sample-specific nondetect reporting limits for different types organic chemicals in fish tissue. As discussed in Section 5.7.9.1, the elevated PAH detection limits create an appearance of very high potential risks. It also appears likely that some organic chemical results listed as J-qualified might in fact be UJ (not detected). Additionally, an assessment protocol that requires the use of one-half detection limits to calculate exposure point concentrations for PCBs that have no positive detections (see Section 4.0 and Section 5.7.9.1) contributes to high risk values for these analytes. For these reasons, a summary of fish ingestion cancer risks and chemical hazard is also presented for just metals and radionuclides so that the impact of the elevated organic chemical detection limits can be evaluated.

Cancer Risk

The RME and CTE risk results for the fish ingestion exposure pathways in the CTUIR, Rural-Residential, and Avid Angler exposure scenarios (radionuclides + chemicals) are shown in Tables 5-55, 5-56, and 5-57, respectively. Cancer risks (radionuclides and chemicals) related to fish ingestion exposure for the Avid Angler, Rural-Residential, and CTUIR exposure scenarios

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in all four near-shore operational exposure areas and the reference area are shown in Figures 5-28, 5-29, and 5-30, respectively. Per EPA guidance (EPA 1989), cancer risk results are displayed with only one significant figure.

In the 100 Area and 300 Area, about 70% to 80% of the RME and CTE cancer risk is related to carcinogenic PAHs, primarily benzo(a)pyrene and dibenz(a,h)anthracene. For all three exposure scenarios, the calculated RME risks in the 100 Area and 300 Area are above the upper limit of $1\text{E-}02$ beyond which the chemical CSF models are inapplicable. In the 100-B/C Area, fish tissue data are limited to strontium-90, technetium-99, Aroclor-1254, and Aroclor-1260. The two Aroclors are responsible for effectively 100% of the RME and CTE cancer risks in the 100-B/C Area. RME risks are at (Rural-Residential) or above (Avid Angler) $1\text{E-}02$ for these scenarios, and above $1\text{E-}02$ for the CTUIR scenario, in the 100-B/C Area. However, it must be noted that Aroclor-1260 was not detected in any of the RCBRA fish tissue samples. In the 100-N Area, where fish tissue data are limited to only strontium-90 and technetium-99, cancer risks for fish ingestion ranged from $7\text{E-}05$ (CTUIR) to $3\text{E-}06$ (RME calculation for the Rural-Residential). CTE fish ingestion risks for the Rural-Residential and Avid Angler scenarios were below $1\text{E-}06$ at the 100-N Area.

The background RME and CTE cancer risk values for fish ingestion, calculated using reference area fish tissue data, were about a factor of 100 lower than those in the 100 Area and 300 Area. This is because the problem of elevated PAH detection limits was not present in the reference area fish tissue results. reference area fish ingestion CTE cancer risk for both the Rural-Residential and Avid Angler scenarios was $1\text{E-}04$, with 60% attributable to PCBs and the remainder mostly to aldrin and delta-BHC. The RME reference Area fish ingestion cancer risks for the Rural-Residential and Avid Angler scenarios were $9\text{E-}04$ and $4\text{E-}03$, respectively. CTUIR fish ingestion cancer risks for the reference area were greater than $1\text{E-}02$. Like the CTE results, these calculated risks were attributable primarily to PCBs, with smaller contributions from aldrin and delta-BHC.

Potential fish ingestion cancer risks excluding the organic chemical fish tissue data would be substantially lower than those shown in Tables 5-55, 5-56, and 5-57. For example, the Avid Angler CTE cancer risk of $7\text{E-}03$ (100 Area) and $1\text{E-}02$ (300 Area) would be approximately $4\text{E-}05$ for just radionuclides and metals. The combinations of analytes and exposure pathways contributing to fish ingestion cancer risks for radionuclides and metals are similar in both operational and reference area tissue data. reference area RME background cancer risk for fish ingestion in the Avid Angler scenario is associated primarily with arsenic (80%) and (as discussed in Section 5.7.8) about 15% via potassium-40. However, arsenic in fish tissue is generally not present in toxic elemental form, which is the basis for the oral cancer slope factor used in the calculation of cancer risk via fish ingestion. Instead, arsenic in fish tissue is more commonly present as organic species such as monomethylarsenic acid or dimethylarsenic acid, which are considered to be far less toxic than inorganic arsenic. The percentage of inorganic arsenic in fish tissue has been estimated at approximately 10% (FDA 1993). Therefore, it is likely that arsenic fish ingestion cancer risk is overestimated by approximately a factor of 10 for this reason.

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Radiation Dose

The RME and CTE radiation dose results for the fish ingestion exposure pathways in the CTUIR, Rural-Residential, and Avid Angler exposure scenarios are shown in Tables 5-58, 5-59, and 5-60, respectively. Radiation doses related to fish ingestion exposure for the Avid Angler, Rural-Residential, and CTUIR exposure scenarios in all four near-shore operational exposure areas and the reference area are shown in Figures 5-31, 5-32, and 5-33, respectively.

Total radiation dose results for fish ingestion in the 100 Area and 300 Area are approximately equivalent. CTE results for the Rural-Residential and Avid Angler scenarios are about 5 to 6 mrem/yr. RME results for these scenarios are about 10 mrem/yr and 50 mrem/yr, respectively. CTUIR fish ingestion dose is approximately 130 mrem/yr in both the 100 Area and 300 Area. In all cases, americium-241 is the primary contributor to fish ingestion radiation dose. In the 100-B/C and 100-N Areas, where radionuclide fish tissue data are limited to only strontium-90 and technetium-99, calculated radiation doses were approximately 1 mrem/yr or less for all scenarios.

The background RME radiation dose values for fish ingestion, calculated using reference area fish tissue data, were notably higher than the values for the 100 Area and 300 Area. RME background doses for the Rural-Residential and Avid Angler scenarios were about 20 and 90 mrem/yr, respectively. The background fish ingestion dose for the CTUIR scenario was about 200 mrem/yr. Americium-241 was again the primary contributor to fish ingestion dose, although the contribution of uranium-233/234 (about 25%) was higher in the reference area than in the 100 or 300 Areas. The CTE background fish ingestion risks were about the same as in the 100 Area and 300 Area, indicating that the higher reference area RME doses are related to higher UCL values from a larger degree of variance in the concentrations of key radionuclides.

Chemical Hazard

The RME and CTE HI results for the fish ingestion exposure pathways in the CTUIR, Rural-Residential, and Avid Angler exposure scenarios are shown in Tables 5-61, 5-62, and 5-63, respectively. Hazard indices related to fish ingestion exposure for the Avid Angler, Rural-Residential, and CTUIR exposure scenarios in all four near-shore operational exposure areas and the reference area are shown in Figures 5-34, 5-35, and 5-36, respectively.

Hazard indices in the 100 Area, 300 Area, and 100-B/C Area are almost wholly related to PCBs. The highest HI values were calculated for the 300 Area, although all PCB measurements were "nondetect" in this area. An HI of about 11,000 was calculated for the CTUIR scenario in the 300 Area, and an RME value of about 1,000 for the Rural-Residential. By contrast, the CTE HI value for the Rural-Residential scenario in the 300 Area was about 60, an indication of the instability in the UCL calculation for PCB nondetects. In fact, the UCL values (about 4 mg/kg) exceed the mean value by a factor of 10. In the 100 Area, where Aroclor-1254 was detected in some 100-B/C Pilot samples, RME HI values were about threefold lower than in the 300 Area, but the CTE HI for the Rural-Residential scenario was about 90. No chemical data for fish tissue samples were obtained in the 100-NR-2 investigation.

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The background RME HI values for fish ingestion, calculated using reference area fish tissue data, were approximately 60 and 200 for the Rural-Residential and Avid Angler scenarios, respectively. The CTE HI for these scenarios was approximately 20. The background HI for the CTUIR scenario was approximately 600. More than 90% of the CTE HIs is related to PCBs. But only about 50% of the RME HIs, and 50% of the CTUIR HI, is related to PCBs. The remaining contribution is from 3,4-methylphenol. The importance of 3,4-methylphenol only in the UCL calculation is again an indication of instability in this calculation. There were four measurements of 3,4-methylphenol in reference area fish tissue samples; the mean value was about 90 mg/kg, but the UCL was approximately 1,400 mg/kg.

Because PCBs dominate the calculated HI values for fish ingestion, the issue of additivity in the critical effects across chemicals is not a major source of uncertainty in the HI results in the operational areas. Neurotoxicity, in addition to decreased body weight, is the critical effect related to the oral RfD for 3,4-methylphenol. Although the critical effects related to the IRIS oral RfD for Aroclor-1254 do not specifically include neurological effects, neurological effects have been among the impacts attributed to PCBs in some human and animal studies.

Potential fish ingestion HI values excluding the organic chemical fish tissue data would be substantially lower than those shown in Tables 5-61, 5-62, and 5-63. The Avid Angler RME HIs of 1,000 (100 Area) and 4,000 (300 Area) would be approximately 15 for just metals. The CTE HI value for the Avid Angler would decrease from approximately 60 (300 Area) and 90 (100 Area) to just 1. A similar impact would be observed for the Rural-Residential and CTUIR scenarios. As was the case for cancer risk, arsenic is also a major contributor to fish ingestion HI values, particularly in the 300 Area. Because arsenic was not detected in the RCBRA reference area fish tissue samples, it is not a contributor to background fish ingestion HI values.

5.7.8 Background Risks for Selected Naturally Occurring Radionuclides

Risk assessment results for a subset of naturally occurring radionuclides are presented in this section. These radionuclides include potassium-40, radium-226, radium-228, thorium-228, thorium-230, and thorium-232. As discussed in Section 4.3, the concentrations of these analytes are practically identical in operational area and reference area soil samples. However, the contribution of background levels of these radionuclides to calculated cancer risk and radiation dose results is still very high. Because of this, when risk and dose related to these radionuclides are included in the risk calculation sums, the potential impacts of residual levels of Hanford Site-related contamination in soil are not observable for all but a few of the remediated waste sites. These radionuclides are not associated with historical Hanford Site processes and operations.

Cancer risk results (radionuclides + chemicals) related to reference area concentrations of potassium-40, isotopic thorium, and isotopic radium for all exposure scenarios, as well as the fish ingestion pathway for the Avid Angler, Rural-Residential, and CTUIR exposure scenarios, are shown in Table 5-64. Radiation dose results related to reference area concentrations of potassium-40, isotopic thorium, and isotopic radium for all exposure scenarios, as well as the fish ingestion pathway for the Avid Angler, Rural-Residential, and CTUIR exposure scenarios, are shown in Table 5-65.

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Cancer risks (radionuclides + chemicals) for those exposure scenarios related to the individual remediated waste sites (Industrial/Commercial Worker, Resident Monument Worker, Rural-Residential, and CTUIR) are shown in Figure 5-37. In the Rural-Residential and CTUIR scenarios, approximately 70% to 90% of the cancer risk is related to potassium-40. In the CTUIR scenario, ingestion of either produce (local exposures only) or native plants is the primary exposure route contributing to cancer risk. In the Rural-Residential scenario, milk ingestion is the dominant exposure route related to cancer risk for potassium-40. In the Resident Monument Worker and Industrial/Commercial Worker scenarios, which do not incorporate exposure via foodstuffs, external irradiation is the most important exposure route. Potassium-40, and a combination of radium-228, radium-228, and thorium-228, contribute in approximately equal proportions to cancer risk for these scenarios.

Radiation doses for the Industrial/Commercial Worker, Resident Monument Worker, Rural-Residential, and CTUIR scenarios are shown in Figure 5-38. The main difference in the contribution of particular radionuclides and pathways from those discussed for cancer risks is the slightly increased importance of other radionuclides (especially radium-228) in addition to potassium-40 in the Rural-Residential and CTUIR scenarios.

Cancer risks for the Casual User and Avid Hunter recreational scenarios, and for sediment exposures related to the Avid Angler scenario, are shown in Figure 5-39. Radiation doses for these same scenarios are shown in Figure 5-40. The contribution of radionuclides and exposure routes for the Recreational scenarios is similar to that described for the Industrial/Commercial, Resident Monument Worker, Rural-Residential, and CTUIR scenarios. Potassium-40 contributes virtually all of the cancer risk and radiation dose via meat ingestion for the Avid Hunter exposure scenario. For the sediment-related exposure pathways of the Avid Angler scenario, and for exposure routes other than game ingestion in the Avid Hunter scenario, potassium-40, and a combination of radium-228, radium-228, and thorium-228, contribute in approximately equal proportions to cancer risk and radiation dose.

Cancer risks and radiation doses related to the fish ingestion exposure pathway for the Avid Angler, Rural-Residential, and CTUIR exposure scenarios are shown in Figures 5-41 and 5-42, respectively. In the fish ingestion exposure pathway, potassium-40 is the dominant contributor to cancer risk and radiation dose across all exposure scenarios that include this pathway (Rural-Residential, Avid Angler, and CTUIR scenarios).

5.7.9 Uncertainty Analysis for the Human Health Risk Assessment

The principal tools applied in the RCBRA for quantifying uncertainties in the risk estimates are (1) the use of RME and CTE parameter values in the risk calculations, and (2) the use of multiple exposure scenarios to address a range of low- and high-intensity land uses. The range of exposure parameter values related to behavioral and/or physiological characteristics (i.e., ingestion and inhalation rates, exposure frequency) provide a measure of uncertainty related to the attributes of individual receptors within a receptor population. The use of the mean and UCL exposure concentrations in the CTE and RME calculations, respectively, provides a measure of the importance of uncertainty in the COPC concentrations in exposure media to the risk

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estimates. Finally, the range of the RME and CTE results across the various exposure scenarios provides information on the importance of the different exposure pathways and receptor characteristics on the risk estimates.

The quantitative measures of uncertainty evaluated via the CTE and RME calculations, and the use of multiple exposure scenarios, can only address those aspects of uncertainty that relate to the choice of specific input parameter values and exposure pathways. A semiquantitative or qualitative assessment of uncertainty will be provided for other aspects of the risk assessment that affect the final estimates. These include the following:

1. Uncertainty in data collection and evaluation, including analytical data quality and data representativeness
2. Uncertainty in the exposure assessment, including the basement excavation model and various intermedia transport models for developing the exposure point concentrations
3. Uncertainty in the toxicity assessment, including models of chemical toxicity and radiation dosimetry upon which assessment of potential health effects are based.

Both the quantitative and qualitative assessments of uncertainty are directed towards identifying key assumptions and parameters that contribute the most towards potentially significant human health exposures and effects. A summary of key uncertainties in the human health risk assessment is provided below. More detailed discussions of each key uncertainty are provided in the following subsections.

Risk Assessment Section	Description of Uncertainty	Potential Bias
Data Collection and Evaluation	Analytical data quality for historical CVP and RSVP soil data assumed to be unbiased	Neutral
	Evaluation of only radiation dose and cancer risk, not systemic effects, with isotopic uranium data	Underestimate
	Infrequent detections and elevated detection limits for organic chemicals in fish tissue	Overestimate
	Use of 1/2 the detection limit to calculate exposure concentrations for PCBs and selected radionuclides with all nondetect values	Overestimate
	Representativeness of waste site soil verification data for average soil constituent concentrations	Overestimate
	Estimation of UCL values when biased verification sampling results in one or more outlier values	Overestimate
	Use of sculpin to represent food fish in the fish ingestion exposure pathway	Neutral

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Exposure Assessment	Applicability of any specific exposure scenario to future conditions	Unknown; residential less likely under current land use plan
	Average concentration in sampled media	Quantified via RME and CTE
	Activity of short-lived radionuclides not decayed	Variable
	Values for behavioral variables (except the CTUIR scenario)	Quantified via RME and CTE
	Values for behavioral variables (CTUIR scenario)	Possible overestimate
	Modeled exposure concentrations in hypothetical surface soil following basement excavation	Overestimate
	Modeled exposure concentrations in produce and livestock tissues	Variable
	Exposure concentrations of potassium-40 in beef and milk	Overestimate
	Toxicity Assessment	Intentional bias in chemical cancer slope factors and hazard quotients
Application of dose conversion factors to estimate dose in children and young adults		Underestimate
Summation of hazard quotients across chemicals to estimate a hazard index; Rural-Residential and CTUIR scenarios		Overestimate
No adjustment of CSFs for childhood exposure to PAHs		Underestimate
Use of no-threshold dose-response model for CSFs of nonmutagenic carcinogens		Overestimate
Use of inorganic arsenic oral CSF for exposure to arsenic measured in aquatic organisms		Overestimate

5.7.9.1 Uncertainties Related to Data Collection and Evaluation. Uncertainty pertaining to data collection and evaluation encompasses sample collection activities, laboratory sample preparation and analysis, and data preparation and analysis. Uncertainty related to chemical concentrations in soil and biota samples, including sample collection and laboratory sample preparation and analysis, is generally not a significant contributor to overall uncertainty in risk assessment results. A major reason for this is that QC samples are used to ensure that analytical results are within acceptable levels of precision and accuracy. However, the use of environmental data from a variety of sampling programs over time in this assessment introduces a potentially higher degree of uncertainty in the consistency of analytical results than is usual due to differences in sample acquisition methods, sample preparation techniques, and analytical methods over time.

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A condition of the CVP/RSVP soil verification data is that the analytical results are targeted to those specific contaminants identified as the likely risk drivers at a particular waste site. Therefore, unlike the RCBRA soil and sediment data, the particular analytes for which results are available are quite limited and varies among the waste sites. In the calculation of Local Area soil exposure point concentrations, if an analyte was not part of the analytical suite for a waste site, a value calculated from the RCBRA operational area soil data set was substituted for the missing analyte. Both the 100-B/C Pilot Project analytical data (DOE/RL-2005-40) and the data collected for the 100-NR-2 investigation (DOE/RL-2005-22), like the CVP/RSVP soil verification data, are targeted to those specific contaminants identified as the likely risk drivers in these areas. For example, fish tissue data for the 100-NR-2 investigation are limited to strontium-90 and technetium-99.

In many environmental samples, data obtained for isotopic uranium (in units of activity per mass or activity per volume) could be converted to total uranium data (in units of mass uranium per mass of sample, or mass uranium per volume). In this way, the effects of uranium metal as a kidney toxicant could be assessed in addition to evaluation of radiation dose and cancer risk when only isotopic uranium data are available. This conversion is most important when evaluating depleted uranium, because uranium activity relative to mass is reduced relative to natural uranium. However, the isotopic uranium data evaluated for this report do not indicate the presence of depleted uranium.

Analytical Data Quality

As discussed in Section 4.0, laboratory, review, and data validation qualifiers are reported in the database supporting this risk assessment. Detection status is included as a derived field in the database, where a detect status of "TRUE" is assigned when a "U" qualifier (indicating the result was reported below the analytical detection limit) does not occur in any one of the three qualifier fields mentioned above. Analytical results for soil, sediment, water, and biota collected under the SAP (DOE/RL-2005-42) were evaluated against the quality criteria specified in that document. Specifically, the QAPP within the SAP (DOE/RL-2005-42) specified the analytical performance requirements for soil, sediment, water, and biota data. Data from other investigations used to quantify health risks, including the waste site soil verification data, have not been reviewed for analytical performance requirements and have not been not evaluated against any specific quality criteria. Waste site soil verification data were collected under specific revisions of the remedial action/remedial design work plans and SAPs for this work, which were generally updated annually. Historical environmental surveillance data are used in a strictly qualitative manner in the human health risk assessment. Although these data were collected under unknown specifications, this is not a significant impediment to their use in such manner.

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All analytical data used in the human health and ecological risk assessments were subjected to a process for ascertaining their usability to support such assessments. All data were required to have, at a minimum, the following attributes in order to be considered “usable.”

1. An analyte name or CAS identification number
2. A numerical result without a rejected (“R”) qualifier in any field
3. Associated units for the results
4. A media type
5. Definitive locational information.

Even in cases where all five attributes were present, analytical data were at times labeled “not usable” for 1 or more of 15 reasons for which usability codes have been assigned in the database (see Section 4.0). Some of these reasons include inappropriate analytical method, nonstandard units that cannot be converted, physically infeasible results, and mixed media type such as paint chips or concrete. A complete discussion of data usability is provided in Section 4.0.

The application of the data usability protocol described above improves confidence in the analytical data used in the risk assessment by ensuring that all data used to quantify potential health risks have in common a shared set of attributes. The CVP/RSVP data have been used to support remedial decision making, and it has been assumed that the analytical data meet the performance criteria described in the remedial action/remedial design work plans and SAPs governing their collection. However, the CVP/RSVP data were not evaluated relative to the target PQLs provided in Section 2 of the SAP (DOE/RL-2005-42).

Results identified as nondetects in the RCBRA data set (i.e., results for soil, sediment, water, and biota qualified by the laboratory, reviewer, or validator as “U”) were compared to the laboratory required PQL prescribed in the QAPP, as discussed in Section 4.3.4. Of approximately 100,000 reported values in the RCBRA dataset, about 7,000 values (7%) were reported as U-qualified at values higher than the PQL. The data assessment protocol then calls for a comparison of these roughly 7,000 values with human health and/or ecological benchmark criteria to identify nondetect results exceeding media-based lookup values in order to discuss same in the uncertainty analysis. This data assessment protocol has now been superseded by an agreement adopted during discussions with regulatory stakeholders to calculate representative concentrations for a number of radionuclides, and for PCBs, even when all sample results in an exposure area are nondetect. This latter protocol provides a protective evaluation of potential risks related to these analytes, which were selected by Ecology as key analytes within the aforementioned group of 7,000 results. If all results are nondetects, then a mean and UCL value is calculated based on replacing the values of each sample-specific detection limit by one-half of that limit and assuming a lognormal distribution across the data. This protocol was applied to nine PCBs and the following radionuclides: americium-241, carbon-14, cesium-137, cobalt-60, europium-152, europium-154, europium-155, plutonium-238, plutonium-239/240, strontium-90, tritium, uranium-233/234, uranium-235, and uranium-238. Although in principle this should have a minimal impact on the risk results, there can be a significant consequence if analytical detection limits are elevated. Such a situation is described below.

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Analytical results for organic chemicals in fish tissue suffer from a systematic problem related to elevated detection limits. Figure 5-6a shows the range of detected values and sample-specific nondetect reporting limits for PCBs and PAHs in fish tissue. Similar plots for pesticides and for other SVOCs are shown in Figures 5-6b and 5-6c, respectively. Nondetect reporting limits generally meet or exceed detected values for all organic chemicals. In the case of PAHs and PCBs, this analytical data quality problem creates the appearance of significant risk via the fish ingestion pathway. For PCBs, the use of one-half the detection limit to calculate exposure point concentrations for PCBs that are not detected in any fish samples¹¹ contributes to the high risk values. Previous fish sampling in the Columbia River has shown that organic chemicals, including PAHs and PCBs, are detected at locations upstream and downstream of the Hanford Site (EPA 910-R-02-006). Average concentrations of various PAHs in fish tissues were generally between 5 and 10 µg/kg, with maximum values up to about 500 µg/kg (EPA 910-R-02-006, Table 2-1a). Average whole-body summed concentrations of Aroclor-1242, Aroclor-1254, and Aroclor-1260 in various fish species were generally between 30 and 200 µg/kg (EPA 910-R-02-006, Table 2-6). By contrast, fish tissue UCL values for Aroclor-1254 and Aroclor-1260 used in this risk assessment were often about 3,000 µg/kg and reached a high of over 100,000 µg/kg in the 100-B/C Pilot data sets. It seems likely that the very high 100-B/C Pilot results are erroneously reported as detected values.

Data Collection and Evaluation: Spatial and Temporal Distributions of Contaminants

Remediated Waste Site Soil Data. The waste site soil verification data were collected for the purpose of determining compliance with shallow-zone and deep-zone remediation criteria. Therefore, they have reportedly been collected at many sites specifically from locations where residual soil concentrations would likely be highest. In such situations, although the number of confirmation samples may be limited, there would be an expected high bias on the estimates of the mean concentrations of soil constituents. A sampling protocol focused on locations where higher residual contamination might be located also may produce a data set with one or two outlying results. In these cases, UCL calculations may be unstable and result in unrealistically high UCL values that do not reflect average analyte concentrations. The waste soil confirmation data are not ideally suited to evaluating the spatial distribution of contamination within a waste site because the only locational identifiers associated with them is whether they were collected in the shallow (0 to 4.6 m) or deep (>4.6 m) zone of the waste site. More importantly, these data are employed in the framework of a basement excavation model to estimate hypothetical chronic exposure point concentrations in surface soil resulting from such an excavation. This is a highly protective assessment framework and is discussed further in Section 5.7.9.2.

RCBRA Upland and Riparian Soil Data. For the majority of the metals, radionuclides, and organic chemical data, there is relatively little differentiation in results among the various MIS soil sampling locations. However, for lead, and to a lesser extent certain PCBs, significant differences in concentrations were observed at a few MIS locations. For lead at site 600-131, the individual lead results for the five MIS composite samples were 327 ppm, 198 ppm, 29.4 ppm, 12.4 ppm, and 12.3 ppm (mean = 116 ppm). Lead was also relatively high at sites 600-139 and

¹¹ Including Aroclor-1016, Aroclor-1221, Aroclor-1232, Aroclor-1242, and Aroclor-1248.

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600-132, although the variance at these sites was small compared to the mean concentrations. At waste site 600-132, Aroclor-1254 has a result of 9.4 ppm in 1 of the 5 composite samples, which is more than 10 times higher than the next highest result. All of these 600 Area sites are debris sites, and these results are likely related to the heterogeneity associated with occasionally obtaining a small particle of debris in a soil sample. It appears likely that particulate debris at this and perhaps other debris sites may harbor residual levels of lead (possibly related to lead-based paint) that approach or exceed the 400 mg/kg criterion, although it is unlikely that spatially averaged concentrations over an exposure area would do so. Additional discussion of the spatial variability in constituent concentrations in upland and riparian soils is presented in Section 4.0.

100-B/C Pilot Project and 100-NR-2 Sediment Data. Historical sediment samples collected during these sampling campaigns often reveal much higher constituent concentrations than in later sediment data obtained under the SAP (DOE/RL-2005-42). This may be a function of changes in the sampling protocol for DOE/RL-2005-42 relative to the previous sampling. A much larger volume of sediment was reportedly obtained for each sample during the RCBRA sampling. It is possible that earlier sampling, using smaller volumes of sediments, was more specifically focused on characterizing highly localized contamination (such as might be associated with the historical N Area seep) rather than potential chronic exposure concentrations for humans and ecological receptors. Additionally, the sediment samples collected for the 100-B/C Pilot Project may have been prepared using hydrofluoric acid digestion, which completely extracts the mineral matrix of the sample. Soil and sediment samples collected under the SAP (DOE/RL-2005-42) were prepared by nitric acid (i.e., "weak acid") digestion, which more closely represents conditions in the gastrointestinal tract.

RCBRA and 100-B/C Pilot Fish (Sculpin) Tissue Data. The sculpin fish tissue data are representative of a fish species with a restricted home range of approximately one-tenth of a kilometer in diameter. Data for Hanford-related contaminants in this species are used to protectively represent tissue concentrations in other species that may be fished for subsistence or recreational purposes and which have a much broader home range or, in the case of salmon, are anadromous. This could potentially be a source of significant uncertainty. However, calculated fish tissue cancer risks and hazards are primarily associated with analytes (PAHs and PCBs) that are not key Hanford Site contaminants and are known to be widely distributed in the Columbia River. A discussion of uncertainty related to differences in measured tissue constituent concentrations across different aquatic species is provided in Section 5.7.9.2. More importantly, the analytical data quality issues related to organic chemicals in fish tissue overwhelm the issue of the representativeness of sculpin as a surrogate for food fish. Where significant differences in fish tissue concentrations were observed for Hanford Site-related contaminants (uranium in the 300 Area, strontium-90 in the 100-N Area) separate risk calculations were conducted for fish in each area.

5.7.9.2 Uncertainties Related to the Exposure Assessment. There are inherent uncertainties in the application of hypothetical exposure scenarios to unknown future conditions in the Hanford Site area. To an extent, these uncertainties are addressed by the use of multiple exposure scenarios to cover a range of potential exposure intensities. In the context of the

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uncertainty analysis, the exposure scenarios are viewed in a relative sense and organized from low-intensity to high-intensity alternatives as follows:

<u>Exposure Scenario</u>	<u>Intensity of Exposure</u>
Casual User, Industrial/Commercial	Low
Resident Monument Worker, Avid Hunting and Fishing	Medium
Rural-Residential	High
CTUIR	Very High

The likelihood of any particular exposure scenario being completely realized, in the sense that all exposure pathways are complete over a chronic exposure period, is related in an approximate manner to the number of exposure pathways. For example, the Rural-Residential scenario envisions receptors with a domestic water well, an active home garden and orchard, a poultry enclosure, cattle that are used for both meat and milk, and additional dietary contribution from fishing. By contrast, in the Industrial/Commercial scenario, the exposure pathways are all necessarily complete simply as a function of such a building being present and occupied. The differences in the RME risk results across the various exposure scenarios is captured in summary form in Table 5-24.

In 2004, EPA's Richland Project Office solicited an evaluation by experts in EPA Region 10 and Ecology of these scenarios as they were presented for the 100-B/C Pilot Project risk assessment (DOE/RL-2005-40) and Harris and Harper (2004). A variety of specific comments received on the CTUIR exposure scenario presented in Harris and Harper (2004) were presented in a draft memorandum (EPA Richland Project Office 2005). In particular, the reviewers questioned the viability of the chronic fish ingestion rate and chronic inhalation rate, as well as the likelihood that the scenario represents current or plausible future exposure conditions. These concerns were then rebutted in a response from the CTUIR Department of Science & Engineering (CTUIR, 2005). The divergence of opinions expressed in these memoranda suggests that uncertainty in exposure levels in the CTUIR scenario may be greater than that in the other scenarios. An example of particular relevance to the CTUIR risk assessment results is the use of a relatively high plant ingestion rate in the "local area only" variation of the CTUIR scenario. This variation, which envisions a domestic garden akin to that in the Rural-Residential scenario, may not support such a high rate that is justified in Harris and Harper (2004) as relating to undomesticated species with high fiber content.

One aspect of the uncertainty in whether any particular exposure scenario may be realized in the future is a related question of *when* any scenario may be realized. This has special relevance in the assessment of cancer risk and dose for short-lived radionuclides. The radionuclide data used in this risk assessment were employed as reported by the analytical laboratories without decay to any particular point in time. Some of short-lived radionuclides that were significant in the risk assessment, and associated half-life, include cobalt-60 (5.3 years), cesium-137 (30.1 years), europium-152 (13.5 years), europium-154 (8.6 years), and strontium-90 (28.8 years). For these radionuclides, calculated total cancer risk and dose will decrease proportionally over time as a

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function of their half-life. However, these radionuclides were also important in reference area background, so incremental cancer risk and dose may in fact increase in some cases. Even assuming that a scenario is implemented in the present, there will be some overestimation of cancer risk and dose for these radionuclides because their activity has not been decayed over any assumed exposure period.

Within each of the exposure scenarios excepting the CTUIR, uncertainty in the representative concentrations of contaminants in environmental media and in the exposure parameter values is assessed by the calculation of CTE and RME risks. For any given exposure scenario, the CTE risks represent a hypothetical individual with approximately "best estimate" levels of contact with contaminants in the exposure media across the various exposure pathways. The RME risk calculations represent a hypothetical individual with a "reasonable maximum" exposure condition of contact with contaminants in the exposure media.

Differences between CTE and RME risks vary as a function of exposure scenario and health effects endpoint, as well as among the different remediated waste sites. For those few waste sites where risk estimates were significantly higher than most others, differences between RME and CTE risks were often amplified because of a large discrepancy between the mean and the UCL for a key contaminant. For example, here is a comparison of RME and CTE total cancer risk results across scenarios for remediated waste site 116-F-14, one of the sites where both cancer risk and radiation dose were notably elevated compared to most sites.

Exposure Scenario	RME Cancer Risk (116-F-14)	CTE Cancer Risk (116-F-14)
Industrial/Commercial	1E-03	2E-05
Resident Monument Worker	1E-03	3E-05
Rural-Residential	2E-03	8E-05

The differences between CTE and RME risk results ranges between a factor of 25 and 50 for the different scenarios evaluated at site 116-F-14. By contrast, here is a comparison of RME and CTE total cancer risks for site 100-D-12, a site that is nearer the middle of the risk range across all remediated waste sites.

Exposure Scenario	RME Cancer Risk (100-D-12)	CTE Cancer Risk (100-D-12)
Industrial/Commercial	2E-05	4E-06
Resident Monument Worker	5E-05	8E-06
Rural-Residential	3E-04	4E-05

The differences between CTE and RME risk results is only between a factor of 5 and 7.5 for the different scenarios evaluated at site 100-D-12. The differences between CTE and RME exposure parameter values is constant across the different waste sites, only the differences in the exposure point concentrations contribute to variability between CTE and RME results across waste sites. This variability can be easily observed by examination of Figures 5-7 through 5-9 and Figures 5-16 through 5-21. The variability shown in the RME results (i.e., the spread of the calculated risks above a theoretical line identical to the CTE results but shifted "higher" on the

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y-axis due to different behavioral assumptions) is wholly a function of the calculation of the UCL values for exposure point concentrations in soil. The location of the RME risk results in these figures, if they were calculated using the mean rather than the UCL value for COPC concentrations, would be at or below a line drawn to intersect the lowest of the plotted RME risk results.

Differences between the RME and CTE calculations for total cancer risk for the recreational scenarios range between about 20 and 30, as shown below. Differences between mean and UCL exposure point concentrations, particularly for PCBs, is important in differentiating these results. However, differences between RME and CTE estimates for the recreational scenarios also reflect greater differentiation in the behavioral variables rather than in scenarios applied on the scale of a remediated waste site.

Exposure Scenario	RME Cancer Risk	CTE Cancer Risk
Casual User	3E-06	1E-07
Avid Angler (sediment exposure, 100 Area)	7E-06	3E-07
Avid Hunter	1E-04	4E-06

In addition to differentiation between RME and CTE calculations, a second aspect of uncertainty related to the exposure assessment involves the models used to estimate exposure point concentrations in unsampled exposure media. Across the various exposure scenarios, these media include the following:

- Hypothetical surface soil following basement excavation
- Foodstuffs (garden produce, poultry and eggs, beef and milk, and wild game)
- Ambient air.

A basement excavation model was used to estimate potential contaminant concentrations in surface soil following construction of a home or commercial building. As described in Section 5.3.2, the basement excavation model was developed to maximize potential exposures to contaminants in the shallow zone via excavation and to incorporate potential exposure to deep-zone contamination via deposition of drill cuttings on the ground surface. While physically plausible, this model is likely to significantly overestimate actual exposure to subsurface contamination. For sites where the depth of clean fill is more than 3 m, this overestimation is related to the assumed parallel orientation of the basement to the long axis of the waste site and the assumption of effectively infinite thickness of contamination in soils perpendicular to the sidewall. For sites with a lesser excavation depth, assignment of soil confirmation results to deeper soils more distant from the contaminant sources is likely to overestimate actual residual concentrations in the excavated material. Another potentially protective aspect of the model is the use of RCBRA operational area surface soil data to represent the material used as backfill.

As discussed in Section 5.3, both plant-soil concentration ratios (K_{p-s}) and feed-to-tissue transfer factors for meat, milk, and eggs (B_a) reflect an assumption that there is a linear and unchanging relationship between soil (used here as a surrogate for feed) and tissue concentrations. Because

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this assumption ignores physiological mechanisms controlling the accumulation of toxic substances, tissue concentrations of contaminants are susceptible to overestimation when soil concentrations are elevated. Additionally, the octanol-water partition coefficients used to model K_{p-s} and B_a values are known to be subject to error, and the regression models in which they are employed may also over- or underestimate these values for particular organic chemicals.

Among the transfer factors used in this assessment, those for chicken and eggs are particularly susceptible to a high degree of uncertainty. As noted in Section 5.3.12, many of the poultry transfer factors published in NUREG/CR-5512-V1 that were used in this assessment derive from data published by Ng et al. (1982). However, transfer factors for various elements were derived from data for chemically similar elements (NUREG/CR-5512-V1). Even when there were element-specific data available, there were generally very few observations for chicken meat or eggs for each element (Ng et al. 1982). A comparison of values from NUREG/CR-5512-V1 with the transfer factors for cadmium and zinc described EPA/530-R-05-006 is telling. The EPA/530-R-05-006 values of the chicken transfer factors for cadmium and zinc are smaller than those published in (NUREG/CR-5512-V1) by factors of approximately 8 and 700, respectively. Furthermore, the relative bioavailability of specific metals may be expected to vary between soil and feed, so that the application of the feed-to-tissue factors to soil in this assessment is a source of still more uncertainty in the modeled tissue concentrations of contaminants.

More commonly in this risk assessment, soil exposure concentrations calculated for the remediated waste site source terms were often close to those in the reference area. In these cases, it is likely that estimated tissue concentrations of some constituents would not be grossly overestimated and may even be underestimated. This might occur for essential trace nutrients, which could be concentrated in biotic tissues relative to low soil levels. Whether such tissue concentrations might be underestimated then depends on the range of soil concentrations used in the studies underlying the transfer factor values, among other factors.

Potassium-40 in foodstuffs was determined to be an important contributor to cancer risk and radiation dose in the CTUIR, Rural-Residential, and Avid Hunter exposure scenarios. Although potassium-40 concentrations are associated with natural background, it is also possible that modeled concentrations in beef, milk, chicken, and eggs have been biased. Representative concentrations in mule deer and cottontail rabbit muscle tissue for potassium-40 are approximately 3 to 3.5 pCi/g. By contrast, modeled beef tissue concentrations are approximately 8 pCi/g. Although there are no available survey data to assess measured concentrations in milk, a key exposure medium in the Rural-Residential scenario, it is likely that modeled concentrations of potassium-40 may also be biased high since the transfer factor models for these media are similar.

Ignoring the organic chemical data in the fish ingestion exposure pathway assessment, key COPCs in sculpin fish tissue for the CTUIR, Rural-Residential, and Avid Angler exposure scenarios would include arsenic and (among the naturally occurring radionuclides) potassium-40. Potassium-40 representative concentrations in bass and whitefish muscle tissue (3 to 4 pCi/g) are approximately equivalent to reference area sculpin values and about twice as high as operational area values for sculpin. Arsenic "whole organism" data are available for juvenile Chinook

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salmon, with representative concentrations of approximately 0.5 to 1.0 mg/kg. These values are commensurate with the sculpin data used in the risk assessment and also similar to arsenic concentrations measured in clams and aquatic macroinvertebrates (such as crayfish) from samples collected under the SAP (DOE/RL-2005-42). The available data indicate that health effects related to fish ingestion for arsenic and potassium-40 would not be affected by utilizing these different species to represent food fish.

The Particulate Emission Factor model (EPA/540/R-95/128, EPA 2002b) was used to estimate dust concentrations in ambient air related to surface soils. This exposure pathway was rarely identified as a significant contributor to potential health effects. Being a protective screening-level model, it is unlikely that estimated dust concentrations calculated with this model would be biased low. One uncertainty with this model identified in Section 5.4.3 is the appropriate value for the vegetation fraction parameter. Because the model is applied to varying conditions (cattle pen, garden, residential landscaping, etc.), it is possible that the assumed value of 0.3 may underestimate dust emissions for specific areas. Decreasing the value of this parameter from 0.3 to 0.1 results in an increase in ambient dust concentrations of about 20%, which would not have an appreciable effect on the results for this exposure pathway.

Modeling of VOC transport from soils to indoor air was not performed in the risk assessment. VOCs were suspected of being COPCs, and therefore sampled, at only a relatively few waste sites. These sites include 100-C-3, 100-F-14, 100-F-18, 116-F-4, 118-C-4, 128-B-2, 600-23, and 618-4. VOC detections at these sites were associated with concentrations of approximately 0.1 mg/kg or less. Detected VOCs in waste site soils include 1,1,2,2-tetrachloroethane (1 detect), 2-butanone (1 detect), methylene chloride (several detects), tetrachloroethene (2 detects), toluene (several detects), and trichloroethene (2 detects). Naphthalene and 2-methylnaphthalene, being included in the analytical suite for PAHs, were sampled more frequently at individual waste sites. But maximum detected concentrations for these analytes were also quite low, being approximately 1 mg/kg or less. The waste soil data do not indicate the presence of VOCs in soils at levels that could credibly represent a potential source term for chronic exposure via migration to indoor air.

5.7.9.3 Uncertainties Related to the Toxicity Assessment. General sources of uncertainty pertaining to the assessment of chemical carcinogenicity include (1) high-to-low dose extrapolation, (2) the common use of a UCL (typically 95%) on the slope of the dose-response curve for cancer slope factors, (3) uncertainty in whether a particular chemical is in fact a human carcinogen, and (4) uncertainty in the applicability of the no-threshold model of carcinogenesis, particularly for chemicals that may not act as mutagens. As discussed in Section 5.5.3, most chemical CSFs are based on carcinogenic effects observed at relatively high dose rates in test animals that have been extrapolated to lower dose rates in humans. The underlying assumption is that even a very low level of exposure carries some risk of carcinogenesis.

General sources of uncertainty pertaining to the assessment of systemic toxicity include (1) the application of UFs on the dose-response data, (2) relying on a single "critical effect" to measure toxicity, and (3) toxicological interactions among the various COPCs. Uncertainty factors are used to account for several possible sources of uncertainty in developing an RfD including

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extrapolating from the NOAEL or LOAEL to a chronic RfD, variability in sensitivity in the human population, interspecies variability between humans and test animals, and inadequate dosing periods in a critical study. These UFs (and related metabolism factors) are designed to introduce a protective bias in the toxicity criteria such that the potential for adverse effects in sensitive human subpopulations will not be underestimated. There is also considerable uncertainty, and usually a protective bias, in the screening-level practice adopted in this assessment of summing hazard quotients across chemicals to estimate a hazard index.

With respect to radionuclides, uncertainties may relate to the estimation of radiation dose as well as to the assessment of carcinogenic risk associated with any particular dose. One of the primary distinctions between the toxicity criteria used in this assessment to quantify radiation dose and radionuclide cancer risk is that the former pertain only to adults whereas the latter are applicable for use in estimating cancer risks for a general population composed of adults and children. Therefore, there is less confidence in the estimates of radiation dose for scenarios that include child receptors than in scenarios related strictly to adult exposures. Because infants and young children have proportionally larger organ masses relative to their body size, organ-specific radiation doses may be underestimated for these receptors. There is also an important distinction to be made between chemical and radionuclide CSFs. Although chemical CSFs are commonly calculated as the 95% UCL on the slope of the dose-response curve, radionuclide CSFs reflect an average estimate of the lifetime risk of cancer. Although chemical and radionuclide cancer risk estimates have been summed in this assessment, the intentional bias associated with chemical CSFs does not strictly allow for simple summation of chemical and radionuclide cancer risks. Additionally, many chemical CSFs are based on animal studies and therefore incorporate uncertainties that do not pertain to radionuclide CSFs, which are based on human epidemiological studies.

Several of the metal COPCs addressed in the risk assessment (i.e., zinc, manganese, copper, total chromium, selenium) are also essential micronutrients required by the body for normal functioning. The body normally exerts a degree of homeostatic control over the body burdens of these metals following ingestion exposures, such that in order for systemic toxicity to manifest the body's control mechanisms must be overwhelmed or incapacitated. Thus, chronic toxicity related to relatively low dose rates above the daily requirement may be unrealistic, although at somewhat higher exposure rates there may be concern for subpopulations with genetic predispositions towards problems in homeostatic regulation of one of these metals.

The applicability of the generic uncertainties described in the previous paragraphs to particular results in this assessment varies. As discussed in Sections 5.7.2 through 5.7.6, there are some few instances where two or more chemicals contribute in a roughly equal manner to a hazard index substantially above 1.0. Specifically, this is the case for the some waste sites and for background HI for the Rural-Residential and CTUIR scenarios. The potential additivity of HQ values is discussed in the results sections for these scenarios. It may be concluded that chemical hazard has been overestimated by perhaps a factor of 2 to 3 by the summation of chemical-specific HQ values in these cases.

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There are numerous instances where chemicals and radionuclides both contribute significantly to a cancer risk result. The most common occurrence is the summing of cancer risks via arsenic and radionuclides in soil in the Rural-Residential and CTUIR exposure scenarios. Also, excluding the organic chemical results, arsenic and potassium-40 are both important drivers of cancer risk via fish ingestion. In the case of fish ingestion, as discussed below, the principal uncertainty is likely to be related to the chemical form (and associated toxicity) of arsenic in fish tissue. Although the different bases of the chemical and radionuclide slope factors makes the general summation of these cancer risks suspect, arsenic is (like ionizing radiation) a known human carcinogen and has a slope factor based on human epidemiological data. Therefore, uncertainty introduced by the addition of chemical (arsenic) and radionuclide cancer risks is not as large in this assessment as might more generally be the case.

As discussed in Section 5.5.3, EPA/630/R-03/003F has recently published guidance for adjusting cancer potency estimates for childhood exposures to mutagenic carcinogens. Among the 12 chemicals listed in Table 1b of EPA/630/R-03/003F that EPA calls out as having been identified as mutagens, the four PAHs (benzo(a)pyrene, dibenzanthracene, dimethylbenz(a)anthracene, and 3-methylcholanthrene) are of particular relevance in this assessment because carcinogenic PAHs are among the COPCs addressed in the human health risk assessment. For reasons described in Section 5.5.3, a quantitative adjustment of the CSFs for carcinogenic PAHs was not performed for child receptors in this assessment. The adjustment factors for CSFs described in EPA/630/R-03/003F are 10 for ages 0 to 2 years, and 3 for ages 2 to 16 years. Over a 30-year residential RME exposure duration, these changes amount to an increase in estimated lifetime cancer risk of approximately 2.5 times. This potential underestimation, while not insignificant per se for carcinogenic PAHs, is not substantial for the overall risk assessment because these compounds are only key chemical contributors to carcinogenic risk via the fish ingestion exposure pathway. Unfortunately, issues relating to the analytical data quality for PAHs in fish tissue (see Section 5.7.9.1) overwhelm uncertainty on the scale of a factor of 2 or 3.

It is important to recognize that EPA's guidance related to childhood exposure to mutagens is contained in a companion document to *Guidelines for Carcinogen Risk Assessment* (EPA/630/P-03/001F). In these guidelines, EPA distinguishes between mutagenic and nonmutagenic carcinogens. Although CSFs for both types of carcinogens are currently developed using a no-threshold model of dose-response (see Section 5.5.3), EPA/630/P-03/001F stresses the importance of differentiating these mechanistically different types of carcinogens because there is likely to be a threshold dose for the latter. Hence, a complete implementation of EPA's cancer risk assessment guidelines (EPA/630/P-03/001F) may have the effect of reducing the estimated cancer incidence risk for chemicals with laboratory evidence of carcinogenicity that are not mutagens.

Arsenic in fish tissue is potentially a major contributor to potential cancer risk and chemical hazard in the exposure scenarios involving fish consumption. Arsenic in fish tissue is generally not present in a toxic elemental form. Arsenic in fish tissue is more commonly present as organic species such as monomethylarsenic acid or dimethylarsenic acid, which are considered to be far less toxic than inorganic arsenic. The percentage of inorganic arsenic in fish tissue has been estimated at approximately 10% (FDA 1993). In a survey of fish contamination in the

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Columbia River, EPA Region 10 reports the average percentage of inorganic arsenic in anadromous fish species such as salmon and steelhead as about 1% (EPA 910-R-02-006). For resident fish species including sucker and sturgeon, the average was about 9% (EPA 910-R-02-006). Hence, arsenic-related cancer risks may be overestimated by approximately a factor of 10 when applied to resident fish or a factor of about 100 when applied to salmon and steelhead.

5.8 HUMAN HEALTH RISK ASSESSMENT RESULTS FOR GROUNDWATER EXPOSURES

Unfiltered groundwater data collected under the SAP (DOE/RL-2005-42) are used to represent current groundwater constituent concentrations. As described in DOE/RL-2005-42, representative groundwater monitoring wells were selected for this sampling by evaluating existing analytical data from the Hanford Site Groundwater Program and by selecting monitoring wells that spatially represent each operational area. The groundwater data employed in this assessment, collected at 64 monitoring wells, represent only a small subset of the available groundwater data collected over time in the 100 Area and 300 Area. For example, recent groundwater sampling in the vicinity of the former Sodium Dichromate Transfer Facility (100-D-12) has revealed chromium concentrations significantly higher than any captured in the groundwater data used in this assessment. Similarly, trichloroethene was recently detected in an area east of the 316-3 South Process Ponds near borehole 399-3-18 at concentrations higher than has normally been measured in monitoring wells in the 300 Area, but there are no detected concentrations of this analyte in the data set used in this risk assessment. Therefore, these groundwater risk results should be interpreted as semiquantitative estimates for the purpose of establishing the approximate magnitude of potential groundwater-related risks relative to the risks presented in Section 5.7.

Background risk calculations for groundwater employ the geometric mean value for each analyte from Table ES-1 of the Hanford Site groundwater background report (DOE/RL-96-91). This value is employed for both the RME and CTE background risk calculations. Therefore, the RME and CTE calculations are differentiated only by the behavioral variables. This is in essence also the case for the "total risk" groundwater calculations. Since there are only one to two groundwater samples at any well, the RME and CTE exposure point concentration values are essentially identical for all but a handful of well/analyte combinations. Using a 90th percentile for background from Table ES-1 would be likely to overestimate background risks in the RME calculations relative to the RME calculation using the site well data.

5.8.1 CTUIR Scenario Groundwater Results

Cancer Risk

The range of groundwater pathways risk results for the CTUIR scenario (radionuclides + chemicals) is from 1E-04 to >1E-02, with one well showing a result of zero. The median cancer risk value across all wells is 2E-03, and the average is 8E-03, reflecting the skewness in the results where risks at certain wells are considerably elevated relative to the majority. For just

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chemicals, the range is zero to $>1E-02$, and for just radionuclides, from zero to $1E-02$. The range of cancer risk values (radionuclides + chemicals) for all 64 monitoring wells is shown in Figure 5-43, where calculated values in excess of $1E-02$ are simply shown as $1E-02$. Groundwater risk results are tabulated according to operational area, and then ranked by magnitude, in Table 5-66.

Among specific wells, the highest cancer risks ($1E-02$ and higher) are primarily associated with hexavalent chromium and sweat lodge inhalation (B8778, B8753, A4570, B8750, 199-N-80, A4647, C4670, 199-K-22, and A4600). As discussed in Section 5.3.9, Harris and Harper (2004) provides models for calculating air concentrations of both volatile and nonvolatile constituents in the sweat lodge. The cancer risk at A4614, however, is related primarily to Aroclor-1254 via dermal absorption while bathing. Cancer risks approaching $1E-02$ are also seen from strontium-90 (A9910, A4679) via water ingestion (3:1 from domestic versus sweat lodge uses). Other analytes contributing to cancer risks in the range of $1E-3$ to $3E-3$ include uranium-234 (A5044 and 399-4-9; sweat lodge inhalation), uranium-238 (A5044 and 399-4-9; sweat lodge inhalation), and arsenic (A4675; domestic ingestion and sweat lodge inhalation). Across all exposure pathways (domestic water ingestion, domestic water inhalation, domestic water dermal absorption, sweat lodge water ingestion, sweat lodge water inhalation, and sweat lodge water dermal absorption), exposure via inhalation in the sweat lodge contributes more than 70% of total groundwater risks for approximately one-half of the 64 wells shown in Table 5-66.

A major factor in the unusually high sweat lodge inhalation cancer risks is the protective bias inherent in the models for calculating air-phase constituent concentrations. These models do not account for any air exchange in the sweat lodge during the 1-hour exposure period. For volatiles and semivolatiles, the total mass of constituents used in the 4 L of water is assumed to be present in sweat lodge air throughout the exposure period. For nonvolatile (inorganic chemicals and radionuclides) constituents, the mass contained in that quantity of water necessary to achieve saturated air is assumed to be present throughout the exposure period. By contrast, the air exchange rate employed by EPA (2001b) for modeling VOC concentrations in air during bathing is 12 exchanges per hour.

Background cancer risk values for detected chemicals and radionuclides in each well for the CTUIR scenario (radionuclides + chemicals) range between $4E-08$ to $4E-04$, with one well showing a result of zero. This range of values is well below the range associated with total cancer risk; hence, ILCR values differ only slightly from the total risk values shown in Table 5-66. The ILCR for radionuclides and chemicals, calculated as total cancer risk minus background risk for each well, are shown in Table 5-67.

Radiation Dose

The range of groundwater pathways radiation dose results for the CTUIR scenario is from 0.68 to 840 mrem/yr, with one well showing a result of zero. The median dose is 25 mrem/yr, and the average is approximately 70 mrem/yr, which shows (as with cancer risk) a high degree of skewness in the results across wells. The range of radiation dose values for all 64 monitoring

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wells is shown in Figure 5-44. Groundwater dose results are tabulated according to operational area, and then ranked by magnitude, in Table 5-68.

With only two exceptions (wells A9910 and A4679), dose rates above 45 mrem/yr are associated predominantly with the sweat lodge inhalation exposure pathway. The higher dose rates are almost entirely dominated by uranium-233/244 and uranium-238. However, in wells A9910 and A4679, doses are mostly a function of ingestion of strontium-90 in domestic water. The highest radiation dose related to an analyte other than isotopic uranium or strontium-90 is 18 mrem/yr for carbon-14 in well C4670, mostly via ingestion of domestic water. As discussed for the CTUIR groundwater cancer risk results, the major factor in the unusually high sweat lodge inhalation doses is the protective bias inherent in the models for calculating air-phase constituent concentrations.

Background radiation dose values for detected radionuclides in each well for the CTUIR scenario range between 0.0015 to 15 mrem/yr, with one well showing a result of zero. For all but 6 of the 64 wells sampled, background dose is approximately 15 mrem/yr and is almost entirely related to uranium-233/244 and uranium-238 via the sweat lodge inhalation exposure pathway. Where this is not the case, it is simply because these isotopes were not sampled in the particular well. The incremental dose for radionuclides, calculated as total dose minus background dose for each well, is shown in Table 5-69. For wells where background dose exceeds total dose, a dose of 0 mrem/yr is shown.

Chemical Hazard

As described in Section 5.6.2, chemical hazard is calculated separately for adult and child receptors. In the case of the groundwater exposure pathways for the CTUIR scenario, however, exposure in the sweat lodge is defined only for an adult receptor (Harris and Harper 2004). In all but 1 of the approximately 15 cases where child HIs (calculated solely for domestic water uses) exceed those for adults, the difference is usually about 25% or less. For this reason, and because adult HIs exceed those of children in 55 of the 64 wells, adult HI values are presented here.

The range of groundwater pathways adult chemical HI results for the CTUIR scenario is from 0.45 to 340, with one well showing a result of zero. However, the highest calculated value (560) is for the child at well A4614, as noted above. The median adult HI across wells is 5.5 and the average is 18, which shows (as with cancer risk and dose) a high degree of skewness in the results across wells. The range of HI values for all 64 monitoring wells is shown in Figure 5-45. The adult HI of 310 at well A4614, rather than the child HI of 560, is shown to improve the readability of the other results. Groundwater HI results are tabulated according to operational area, and then ranked by magnitude, in Table 5-70. For the one exception, well A4614 in the 100-H Area where HIs are driven by dermal absorption of Aroclor-1254 while bathing, the child HI is tabulated instead of the adult value.

With only two exceptions (wells A4614 and A4653), HIs above 10 are dominated by the sweat lodge inhalation exposure pathway. These exceptions, Aroclor-1254 in well A4614 and heptachlor epoxide in well A4653, are related to child exposure and dermal absorption while bathing. Otherwise, the higher HIs are almost entirely dominated by inorganic chemicals and the

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sweat lodge inhalation exposure pathway. The dominant chemicals are hexavalent chromium and manganese. As discussed for the CTUIR groundwater cancer risk results, the major factor in the unusually high sweat lodge inhalation doses is the protective bias inherent in the models for calculating air-phase constituent concentrations.

Background HI values for detected chemicals in each well for the CTUIR scenario range between 0.9 to 6.2, with one well showing a result of zero. The ratio of background HI to total HI is shown in Table 5-71. For the single well (A9910) where background HI exceeded total HI, this ratio is shown as 1.0.

5.8.2 Rural-Residential Scenario Groundwater Results

Cancer Risk

The range of RME groundwater pathways risk results for the Rural-Residential scenario (radionuclides + chemicals) is from 4E-06 to 6E-03, with one well showing results of zero. The median RME cancer risk value across all wells is 2E-04, and the average is 3E-04, indicating that the results are considerably less skewed than was the case in the CTUIR scenario. The range of CTE risk results is from 4E-07 to 1E-03, with one well showing results of zero. For just chemicals, the RME and CTE ranges are 8E-06 to 6E-03 and 2E-06 to 1E-03, respectively, with five wells showing results of zero. For just radionuclides, the RME and CTE ranges are from 1E-06 to 2E-03, and 1E-07 to 2E-04, respectively, with one well showing results of zero. Because domestic groundwater RME exposure parameter values differ by perhaps a factor of two at most between Rural-Residential and CTUIR scenarios, the differences in cancer risks between these scenarios is primarily a function of the sweat lodge exposure pathways. In other words, the Rural-Residential cancer risk results are approximately equivalent to the CTUIR results, minus the contribution of exposures in the sweat lodge.

The range of cancer risk values (radionuclides + chemicals) for all 64 monitoring wells is shown in Figure 5-46. The RME value of 6E-03 for well A4614 in the 100-H Area is truncated in the figure in order to improve the scale for viewing other results. Groundwater risk results are tabulated according to operational area, and then by well ID, in Table 5-72.

Rural residential cancer risks are associated almost exclusively with the water ingestion and dermal absorption exposure routes. Because volatile chemicals were not detected in the groundwater samples, tritium is the only constituent contributing to inhalation pathway risks, which were consistently below 1E-07 for both RME and CTE calculations.

The highest calculated groundwater risks (well A4614 in the 100-H Area) are related to dermal absorption of Aroclor-1254. The exposure point concentration of Aroclor-1254 in this well (8.3 µg/L) is within the solubility limit for this compound (43 µg/L; companion database to EPA/530-R-05-006). The other instances of cancer risks exceeding 1E-03 are related to ingestion of strontium-90 (wells A9910 and A4679). However, the number of instances where Aroclor-1254 and strontium-90 are significant contributors to cancer risks is limited. Among chemicals contributing to RME cancer risks above 1E-04, arsenic (via ingestion) and bis(2-ethylhexyl)phthalate (primarily via dermal absorption) are by far the most common. Heptachlor

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epoxide (A4653, A4587, A4647, A4665, and A4669), carbon-14 (C4670 and A4660), and radium-228 (A4650) also contribute to RME cancer risks at or above $1E-04$ in certain wells.

Although they are important contributors to calculated groundwater cancer risks, there is some uncertainty regarding the analytical results for arsenic and radium-228. The range of detected arsenic exposure point concentrations in monitoring wells where it was measured – 3.5 to $14 \mu\text{g/L}$ – is well above the geometric mean value for Hanford Site background of $1.83 \mu\text{g/L}$ (DOE/RL-96-91). Even more pronounced, the detected radium-228 values in the three wells where it was measured (2.0 pCi/L at 199-K-22, 2.3 pCi/L at A4679, and 5.3 pCi/L at A4650) are approximately 100 times larger than the background geometric mean value of 0.032 pCi/L (DOE/RL-96-91).

Background RME groundwater pathways cancer risk values for detected chemicals and radionuclides in each well for the Rural-Residential scenario (radionuclides + chemicals) range between $8E-09$ to $4E-05$, with one well showing a result of zero. For the CTE calculations, these background values range between $7E-10$ to $6E-06$. These range of values are well below the range associated with total cancer risk; hence, ILCR values differ only slightly from the total risk values shown in Table 5-72. The ILCR for radionuclides and chemicals, calculated as total cancer risk minus background risk for each well, are shown in Table 5-73.

Radiation Dose

The range of RME groundwater pathways radiation dose results for the Rural-Residential scenario is from 0.2 to 150, with one well showing results of zero. The median RME dose is 1.5 mrem/yr , and the average is approximately 6 mrem/yr , which shows a high degree of skewness in the RME dose results across wells. The range of CTE risk results is from 0.1 to 92, with one well showing results of zero. The range of radiation dose results for all 64 monitoring wells is shown in Figure 5-47. Groundwater doses are tabulated according to operational area, and then by well ID, in Table 5-74.

As indicated in Figure 5-47, there are only a handful of wells where annual radiation dose exceeds the 15 mrem/yr threshold. These wells include A9910, A4679, and A5044 (RME calculation only). For wells A9910 and A4679, dose is a function entirely of ingestion of strontium-90 in drinking water. At well A5044, dose is related to ingestion of uranium-233/234 and uranium-238.

Background RME radiation dose values for detected radionuclides in each well for the Rural-Residential scenario range between 0.0006 to 0.37 mrem/yr , with one well showing a result of zero. For the CTE calculations, the range is 0.0004 to 0.22 mrem/yr . For all but 6 of the 64 wells sampled, background dose is approximately 0.3 and 0.2 mrem/yr for the RME and CTE calculations, respectively. Background dose is almost entirely related to uranium-233/244 and uranium-238 via the water ingestion exposure pathway. Where this is not the case, it is simply because these isotopes were not sampled in the particular well. The incremental dose for radionuclides, calculated as total dose minus background dose for each well, is shown in Table 5-75. For wells where background dose exceeds total dose, a dose of 0 mrem/yr is shown.

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Chemical Hazard

As described in Section 5.6.2, chemical hazard is calculated separately for adult and child receptors. At each of the groundwater wells, RME child HIs (calculated solely for domestic water uses) exceed those for adults by a factor of approximately 50% to 75%. For the CTE calculations, however, adult HIs tended to be slightly higher when water ingestion dominated exposures while the reverse was true when dermal absorption was more important. Still, the difference between adult and child CTE HIs was usually about 15% or less. For simplicity, only child HI values are presented here.

The range of groundwater pathways child chemical HI results for the Rural-Residential scenario is from 0.06 to 520 for the RME calculations and from 0.02 to 290 for the CTE results, with 1 well showing a result of zero. The median RME child HI is 2.0, and the average is 11, which reflects the influence of the very high HI result at well A4614 (absent this well, the average would be 4.0). The range of HI values for all 64 monitoring wells is shown in Figure 5-48. Because the HIs at well A4614 are so far in excess of those at the other wells, they are not included in the figure. Groundwater HI results are tabulated according to operational area, and then well ID, in Table 5-76.

The high chemical HIs seen in well A4614 is related to dermal absorption of PCBs while bathing. In the remaining wells, HIs are often a function of ingestion of metals including hexavalent chromium, iron, arsenic, and thallium, as well as dermal absorption of organic chemicals including heptachlor epoxide and bis(2-ethylhexyl)phthalate.

Background RME HI values for detected chemicals in each well for the Rural-Residential scenario range between 0.04 to 3.9, with one well showing a result of zero. For the CTE calculations, the range is 0.01 to 1.3. The ratio of background HI to total HI is shown in Table 5-77. For well A4630 in the 100-H Area, background HI exceeds the well HI and the ratio is shown as 1.0.

5.8.3 Resident Monument Worker Scenario Groundwater Results

Cancer Risk

The range of RME groundwater pathways risk results for the Resident Monument Worker scenario (radionuclides + chemicals) is from 4E-06 to 4E-03, with one well showing results of zero. The median RME cancer risk value across all wells is 1E-04, and the average is 2E-04. The range of CTE risk results is from 6E-07 to 7E-04, with 1 well showing results of zero. For just chemicals, the RME and CTE ranges are 6E-06 to 4E-03 and 1E-06 to 7E-04, respectively, with one well showing results of zero. For just radionuclides, the RME and CTE ranges are 1E-06 to 2E-03 and 2E-07 to 3E-04, respectively, with 1 well showing results of zero. The range of ILCR values (radionuclides + chemicals) for all 64 monitoring wells is shown in Figure 5-49. The RME value of 4E-03 for well A4614 in the 100-H Area is truncated in the figure in order to improve the scale for viewing other results. Groundwater risk results are tabulated according to operational area, and then by well ID, in Table 5-78.

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The combinations of exposure pathways and analytes associated with cancer risks in the Resident Monument Worker scenario are analogous to those described for the Rural-Residential. With the exception of the absence of a child receptor, and the use of a slightly shorter (industrial) exposure duration, these exposure scenarios are identical with respect to groundwater exposure. For example, the highest calculated groundwater risks (well A4614 in the 100-H Area) are again related to dermal absorption of Aroclor-1254. Ingestion of arsenic and dermal absorption of bis(2-ethylhexyl)phthalate are again commonly associated with the higher calculated risk results.

Background RME groundwater pathways cancer risk values for detected chemicals and radionuclides in each well for the Resident Monument Worker scenario (radionuclides + chemicals) range between $7E-09$ to $3E-05$, with one well showing a result of zero. For the CTE calculations, these background values range between $1E-9$ to $5E-06$. These range of values are well below the range associated with total cancer risk; hence, ILCR values do not differ greatly from the total risk values shown in Table 5-78. The ILCR for radionuclides and chemicals, calculated as total cancer risk minus background risk for each well, are shown in Table 5-79.

Radiation Dose

The range of RME groundwater pathways radiation dose results for the Resident Monument Worker scenario is from 0.2 to 150, with one well showing results of zero. The median RME dose is approximately 1.5 mrem/yr, and the average is approximately 6 mrem/yr, which shows a high degree of skewness in the RME dose results driven by wells A9910 and A4679. The range of CTE risk results is from 0.1 to 92, with one well showing results of zero. The range of radiation dose results for all 64 monitoring wells is shown in Figure 5-50. Groundwater doses are tabulated according to operational area, and then by well ID, in Table 5-80.

As indicated in Figure 5-50, there are only a handful of wells where annual radiation dose exceeds the 15 mrem/yr threshold. These wells include A9910, A4679, and A5044 (RME calculation only). For wells A9910 and A4679, dose is a function entirely of ingestion of strontium-90 in drinking water. At well A5044, dose is related to ingestion of uranium-233/234 and uranium-238. The results of the radiation dose calculations for the Resident Monument Worker are identical to those described for the Rural-Residential, which differs from the Resident Monument Worker primarily in that the latter has only an adult receptor. Because radiation dose is not normalized by body weight, adult dose exceeds child dose for water ingestion because the daily ingestion rate is higher.

Background RME and CTE radiation dose values for detected radionuclides in each well for the Resident Monument Worker are identical to those described for the Rural-Residential scenario. RME values range between 0.0006 to 0.37 mrem/yr, with one well showing a result of zero. For the CTE calculations, the range is 0.0004 to 0.22 mrem/yr. For all but 6 of the 64 wells sampled, background dose is approximately 0.3 and 0.2 mrem/yr for the RME and CTE calculations, respectively. Background dose is almost entirely related to uranium-233/244 and uranium-238 via the water ingestion exposure pathway. Where this is not the case, it is simply because these isotopes were not sampled in the particular well. The incremental dose for radionuclides,

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calculated as total dose minus background dose for each well, is shown in Table 5-81. For wells where background dose exceeds total dose, a dose of 0 mrem/yr is shown.

Chemical Hazard

Chemical hazard is calculated for the adult receptor in the Resident Monument Worker scenario. The range of groundwater pathways chemical HI results for the Resident Monument Worker scenario is from 0.04 to 280 for the RME calculations and from 0.02 to 180 for the CTE results, with one well showing a result of zero. Removing the outlier of well A4614, the median RME HI across all wells is approximately 1.2, and the average is approximately 2.5 (including well A4614 skews the mean HI to approximately 6.5). The range of HI values for all 64 monitoring wells is shown in Figure 5-51. Because the HIs at well A4614 are so far in excess of those at the other wells, they are not included in the figure. Groundwater HI results are tabulated according to operational area, and then by well ID, in Table 5-82.

The high chemical HIs seen in well A4614 are related to dermal absorption of Aroclor-1254 while bathing. In the remaining wells, HIs are often a function of ingestion of metals including hexavalent chromium, iron, arsenic, and thallium, as well as dermal absorption of organic chemicals including heptachlor epoxide and bis(2-ethylhexyl)phthalate.

Background RME HI values for detected chemicals in each well range between 0.02 and 2.5, with one well showing a result of zero. For the CTE calculations, the range is 0.01 to 1.5. The ratio of background HI to total HI is shown in Table 5-83. For well A4630 in the 100-H Area, background HI exceeds the well HI and the ratio is shown as 1.0.

5.8.4 Uncertainty Analysis for the Groundwater Assessment

As discussed in Section 5.7.9, the principal tool applied in the RCBRA for quantifying uncertainties in the risk estimates is the use of RME and CTE parameter values in the risk calculations. The range of these values for those parameters related to behavioral and/or physiological characteristics (i.e., ingestion and inhalation rates, exposure frequency) provide a measure of uncertainty related to the attributes of individual receptors within a receptor population. The use of the mean and UCL exposure concentrations in the CTE and RME calculations, respectively, provides a measure of the importance of uncertainty in the COPC concentrations in exposure media to the risk estimates. Additionally, a semiquantitative or qualitative assessment of uncertainty will be provided for other aspects of the risk assessment that affect the final estimates, including the following:

1. Uncertainty in data collection and evaluation, including analytical data quality and data representativeness
2. Uncertainty in the exposure assessment, including statistical models for the exposure point concentrations and human exposure models underlying the exposure scenarios
3. Uncertainty in the toxicity assessment, including models of chemical toxicity and radiation dosimetry upon which assessment of potential health effects are based.

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Both the quantitative and qualitative assessments of uncertainty are directed towards identifying key assumptions and parameters that contribute the most towards potentially significant human health exposures and effects.

5.8.4.1 Uncertainties Related to Data Collection and Evaluation. Uncertainty pertaining to data collection and evaluation encompasses sample collection activities, laboratory sample preparation and analysis, and data preparation and analysis.

Analytical Data Quality

Uncertainty related to laboratory sample preparation and analysis is not considered to be a significant contributor to overall uncertainty in the groundwater risk assessment results. As discussed in Section 4.0, laboratory, review, and data validation qualifiers are reported in the database supporting this risk assessment. Detection status is included as a derived field in the database, where a detect status of "TRUE" is assigned when a "U" qualifier (indicating the result was reported below the analytical detection limit) does not occur in any one of the three qualifier fields mentioned above.

The groundwater samples collected under the SAP (DOE/RL-2005-42) were employed in this groundwater risk assessment. The analytical results were evaluated against the quality criteria specified in that document. Specifically, the QAPP within the SAP (DOE/RL-2005-42) specified the analytical performance requirements for the groundwater data. All groundwater data used in this assessment met the performance requirements defined in the QAPP.

Data Evaluation: Spatial and Temporal Distributions of Contaminants

The groundwater risk calculations were conducted using groundwater sampling data collected under the SAP (DOE/RL-2005-42). There is only been a single sample event for most combinations of analyte and monitoring well. Among the 64 wells, only wells 199-H4-48, A4650, A4681, and C4670 have two samples across all analytes. Well A4587 has two additional samples for just hexavalent chromium. Because of the extremely limited number of samples, it is impossible to evaluate patterns or variability in seasonal or long-term trends in groundwater constituent concentrations in these wells. Consequently, there is a high degree of uncertainty in the time-averaged constituent concentrations used in the risk assessment. Exposure point concentrations at each well representing periods of time from approximately 7 to 70 years have been estimated based on data from just one or two samples. The absence of information on variability in groundwater constituent concentrations can be seen in the fact that RME and CTE values for exposure point concentrations are essentially identical for all but a handful of analyte and well combinations.

5.8.4.2 Uncertainties Related to the Exposure Assessment. Within each of the three exposure scenarios incorporating water exposure (CTUIR, Rural-Residential, and Resident Monument Worker), uncertainty in exposure parameter values is assessed by the calculation of CTE and RME risks. The CTE risks represent an individual with average levels of contact with the exposure media across the various exposure pathways. The RME calculation represent a "reasonable maximum" exposure condition of contact with the exposure media. For the CTUIR

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scenario, only a single set of exposure parameter values are provided in Harris and Harper (2004).

Differences between CTE and RME risks vary as a function of exposure scenario and health effects endpoint. Differences between RME and CTE groundwater cancer risks for the Rural-Residential and Resident Monument Worker scenarios was generally between a factor of 5 and 10. Differences between the RME and CTE radiation dose and HI values were commonly smaller, being approximately a factor of two to three in most cases. Unlike the situation observed with the waste sites risk assessment, there were no instances where specific groundwater wells had significantly larger differences between RME and CTE estimates. The reason for this is the limited number of samples at any individual well (one or two samples), which resulted in little or no differentiation between RME and CTE estimates of constituent exposure point concentrations.

In addition to differentiation between RME and CTE calculations, a second aspect of uncertainty related to the exposure assessment involves the models used to estimate exposure point concentrations in unsampled exposure media. The most important model involves the release of constituents from water into air in the CTUIR sweat lodge. Exposure to contaminants in groundwater during the sweat lodge, particularly via inhalation, was a major exposure pathway contributing to health effects via groundwater in the CTUIR exposure scenario. As discussed in Section 5.8.1, an important component of the unusually high sweat lodge inhalation cancer risk is the protective bias inherent in the models defined in Harris and Harper (2004) for calculating air-phase constituent concentrations. These models do not account for any air exchange in the sweat lodge during the 1-hour exposure period.

For nonvolatile (inorganic chemicals and radionuclides) constituents, which are important contributors to sweat lodge inhalation health effects, it is further assumed in the exposure models that these constituents will exist as an aerosol of respirable size following vaporization of the water in which they dissolved. It is not clear that in fact an inhalation pathway exists for nonvolatile constituents in groundwater. For example, such a pathway has not been recognized in EPA risk assessments for nonvolatile chemicals in domestic uses such as cooking or showering. However, it should be noted that there are some animal data suggesting potential carcinogenicity of hexavalent chromium by the oral route of exposure, although the epidemiological and toxicological data were considered inconclusive by EPA during preparation of their 1998 *Toxicological Review of Hexavalent Chromium* (EPA 1998).

In the calculation of dermal absorption of organic chemicals from water, the predicted permeability coefficient (K_p) values for highly lipophilic compounds such as PCBs may be near or outside the effective prediction domain of the K_p model. This situation introduces a large degree of uncertainty into the quantification of risks from aqueous dermal exposures. This is relevant in particular to the risk results for well A4614, where Aroclor-1254 is a major component of calculated cancer risks and hazards. The use of a fraction absorbed value in the calculation of dermal absorption of PCBs, as recommended in Exhibit B-3 of EPA/540/R/99/005, would result in a decrease of 40% to 50% in the calculated dermal pathway risks and hazards presented in this assessment.

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As described in Section 5.3.8, the VOC shower volatilization model does not account for additional contributions of VOCs to indoor air such as dishwashers and washing machines. However, pesticides and phthalates were the only classes of organic chemicals detected in the groundwater samples collected under the SAP (DOE/RL-2005-42). Therefore, potentially low bias in indoor air VOC concentrations is not a factor in the groundwater risk assessment. It is known that there are locations in the 100 Area and 300 Area where VOCs including trichloroethene are present in groundwater (PNNL-15892). Therefore, although potential risks via this exposure pathway are not quantified due to an absence of detected VOCs in the data employed, it is likely that such risks could be quantified using groundwater data from specific locations within the 100 Area and 300 Area.

5.8.4.3 Uncertainties Related to the Toxicity Assessment. General sources of uncertainty pertaining to the assessment of chemical carcinogenicity include (1) high-to-low dose extrapolation, (2) the common use of an UCL (typically 95%) on the slope of the dose-response curve for cancer slope factors, (3) uncertainty in whether a particular chemical is in fact a human carcinogen, and (4) uncertainty in the applicability of the no-threshold model of carcinogenesis, particularly for chemicals that may not act as mutagens. As discussed in Section 5.5.3, most chemical CSFs are based on carcinogenic effects observed at relatively high dose rates in test animals that have been extrapolated to lower dose rates in humans. The underlying assumption is that even a very low level of exposure carries some risk of carcinogenesis.

General sources of uncertainty pertaining to the assessment of systemic toxicity include (1) the application of UFs on the dose-response data, 2) relying on a single "critical effect" to measure toxicity, and 3) toxicological interactions among the various COPCs. Uncertainty factors are used to account for several possible sources of uncertainty in developing an RfD including extrapolating from the NOAEL or LOAEL to a chronic RfD, variability in sensitivity in the human population, interspecies variability between humans and test animals, and inadequate dosing periods in a critical study. These UFs (and related metabolism factors) are designed to introduce a protective bias in the toxicity criteria such that the potential for adverse effects in sensitive human subpopulations will not be underestimated.

The highest calculated cancer risk estimates are related to dermal absorption of Aroclor-1254, with dermal absorption of bis(2-ethylhexyl)phthalate and ingestion of arsenic more commonly associated with significantly elevated cancer risk values above 1E-04. The oral CSFs applied to dermal absorption of PCBs and bis(2-ethylhexyl)phthalate are based on rodent animal studies, although in the case of PCBs, there is fairly good evidence indicating potential carcinogenicity in humans (EPA 2007). Bis(2-ethylhexyl)phthalate is a nongenotoxic carcinogen which acts as a peroxisome proliferators (Klaassen 2001). As discussed in Section 5.7.9.3, the use of a no-threshold model may not be applicable to this mode of carcinogenicity and the associated CSF may overestimate risks related to this chemical.

5.9 ASSESSMENT OF THE 100 AREA RIVER EFFLUENT PIPELINES

Between 1943 and 1988 at the Hanford Site, pipelines extending from reactor outfall structures in the 100 Areas into the Columbia River were used to carry reactor cooling water for discharge

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to the river. Today, the effluent pipelines at the 100-B/C, 100-D, 100-H, 100-F, 100-K, and 100-N Areas remain in place on or beneath the river channel bottom. The river effluent pipelines are known or suspected to contain small amounts of residual contamination from past reactor operations.

Samples of scale from the interior surfaces and enclosed sediment of the effluent pipelines from 100-C, 100-DR, and 100-F Reactors in 1984 were analyzed for radionuclides and revealed primarily cobalt-60, cesium-137, europium-152, europium-154, and europium-155. In 1995, pipe scale and sediment from the interior of the effluent pipelines from the 100-B and 100-D Reactors were sampled and analyzed for metals as well as a larger suite of radionuclides than the 1984 sampling.

Using the 1984 and 1995 data, risk evaluations using the RESRAD and RESRAD-BUILD computer codes have been performed for an exposure scenario in which a pipeline section breaks away from the main pipeline and is washed onto the shore of the river. An evaluation of potential risks related to the river effluent pipelines as they are today, located on or beneath the river channel bottom, was conducted in 1998. Although the 1998 study indicated no potential for significant effects, the more current analysis determined that a child receptor in the Avid Recreational scenario could receive an estimated annual dose greater than 15 mrem/yr above background for 15 years beyond 2007. A complete summary of the 100 Area river effluent pipelines risk assessment, with accompanying tables, figures and references, is provided in Appendix G-2.

5.10 CONCLUSIONS

The human health risk assessment for the 100 Area and 300 Area Component of the RCBRA documented the methods and results for quantifying human exposure and associated health effects related to chemicals and radionuclides in environmental media. Human health risks were assessed for a number of hypothetical exposure scenarios that varied from low-intensity to high-intensity exposure conditions in order to provide risk managers with information on how potential risks may vary under a variety of exposure assumptions. However, the use of any particular scenario in this risk assessment does not imply any endorsement of either the scenarios or the underlying assumptions by DOE or other stakeholders with respect to future land use.

Risk assessment calculations for the Industrial/Commercial Worker, Resident Monument Worker, Rural-Residential, and CTUIR exposure scenarios were conducted for each of the 163 remediated waste sites evaluated in this assessment. Risk assessment calculations for the Recreational scenarios (Avid Angler, Avid Hunter, and Casual User), for which exposure may occur over much broader spatial scales than an individual waste site, integrated exposure over some or all of the 100 Area and 300 Area. A separate set of risk calculations were performed to assess the potential effects of exposure to groundwater within the Resident Monument Worker, Rural-Residential, and CTUIR exposure scenarios. These calculations were conducted for each of the 64 monitoring wells sampled as part of the 100 Area and 300 Area environmental sampling campaign.

Human Health Risk Assessment

As described in Section 5.7, the risk assessment results for exposures related to fish ingestion, and to a subset of six naturally occurring radionuclides¹² in soil and biota, were calculated and presented independently of the risks related to other exposure pathways and analytes. The contribution of the fish ingestion exposure pathway and of background levels of these radionuclides to calculated risks is very high. The potential impacts of residual levels of Hanford-related contamination at the individual waste sites is largely indiscernible when these results are included in the risk calculation sums.

Across the 163 remediated waste sites and the 64 monitoring wells, the range of cancer risk and radiation dose results for the Resident Monument Worker, Rural-Residential, and CTUIR scenarios were approximately equivalent for the soil-related exposure pathways (not including the thorium, radium, and potassium isotopes) and the groundwater exposure pathways. The calculated HI values were generally somewhat higher for the soil-related exposure pathways than for groundwater exposures. However, with the inclusion of potassium-40, and isotopic radium and thorium, soil-related risks for the remediated waste sites generally exceeded those calculated for groundwater exposures. For all but a relatively few remediated waste sites, cancer risk and radiation dose related to the six naturally occurring radionuclides exceeded risk and dose related to residual levels of contamination at these sites. Calculated cancer risks and HI values related to the fish ingestion exposure pathway were commonly much higher than those calculated for either the remediated waste sites or the monitoring wells. Operational area analytical results for organic chemicals in fish tissue suffered from a systematic problem related to elevated detection limits. In the case of PAHs and PCBs, this analytical data quality problem creates the appearance of significant risks via the fish ingestion pathway. It is unlikely that the calculated risks related to fish ingestion for these analytes are meaningful.

Several remediated waste sites were identified where calculated RME cancer risk and/or radiation dose levels were higher than at most other sites and elevated relative to threshold criteria for some or all intensities of exposure scenario. These include sites from the 300 Area, which were remediated to cleanup standards related to industrial land use, and some sites in the 100-F and 100-B operational areas. These sites, and the associated COPCs, include the following:

- 316-5 (arsenic, isotopic uranium, cesium-137, cobalt-60)
- 316-2 (isotopic uranium, arsenic, cobalt-60)
- 300-10 (arsenic)
- 116-F-14 (isotopic europium, cesium-137, cobalt-60)
- 316-1 (arsenic, isotopic uranium)
- 100-F-35 (strontium-90, arsenic, cesium-137)
- 100-F-37 (arsenic)
- 118-B-3 (isotopic europium, arsenic, cesium-137)
- 116-B-11 (isotopic europium, cesium-137)
- 118-F8-1 (isotopic europium, cesium-137).

¹² Potassium-40, radium-226, radium-228, thorium-228, thorium-230, and thorium-232.

Human Health Risk Assessment

Only under the Rural-Residential and CTUIR scenarios did RME HI values exceed 1.0 for any of the remediated waste sites. Some of these sites include the following:

- 100-K-33 (mercury)
- 100-K-30 (mercury)
- 128-C-1 (copper)
- 110-K-32 (mercury)
- 300-10 (arsenic).

Uncertainties related to the human health risk assessment tend to results in protective biases more frequently, and to a greater degree, than biases that may result in underestimation of risks. Some of the key protective biases inherent in the risk calculations include the following:

1. The use of a basement excavation model for accessing subsurface contamination that assumes the worst-case location and orientation of a basement relative to the historical footprint of the waste site;
2. The use of screening-level models with protective assumptions to model transport of chemical and radionuclides among different environmental media for the purpose of calculating exposure point concentrations;
3. The use of chemical cancer slope factors and systemic toxicity criteria with protective assumptions related to dose-response and human sensitivity to effects; and,
4. The use of historical (undecayed) radionuclide activity data to evaluate potential exposures at some unspecified future time.

Figure 5-1. Plan View of Basement Excavation Scenario.

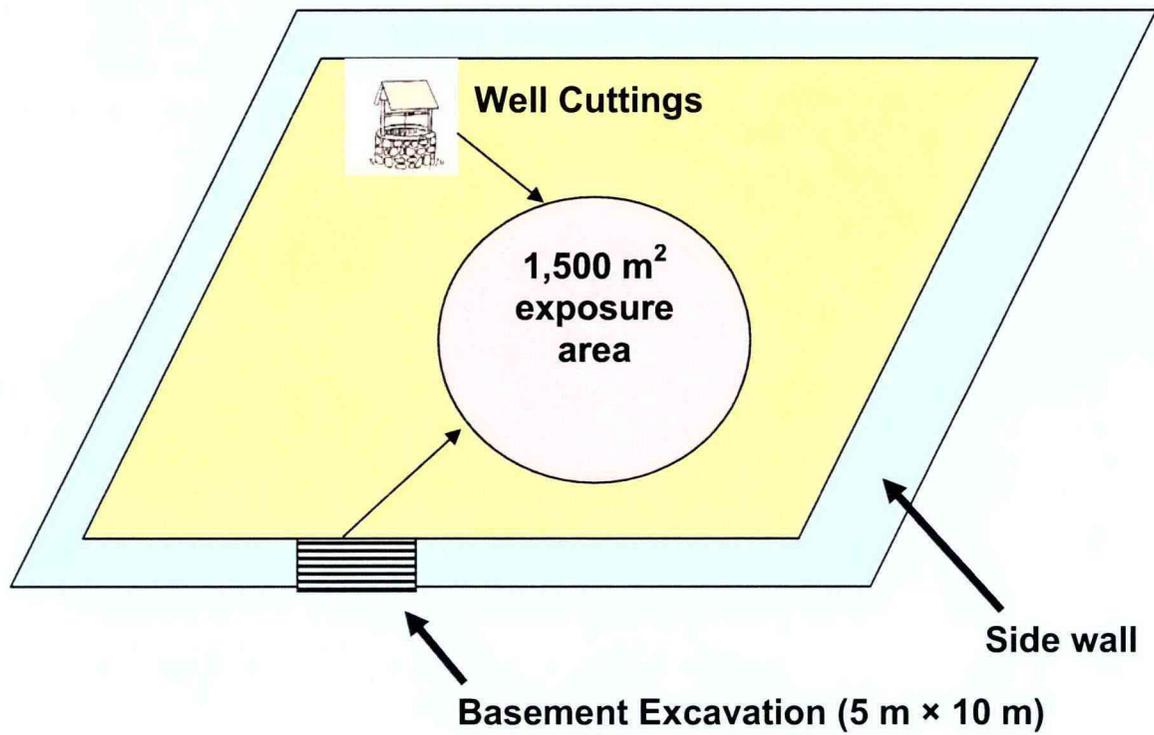


Figure 5-2. Side View of Basement Excavation Scenario.

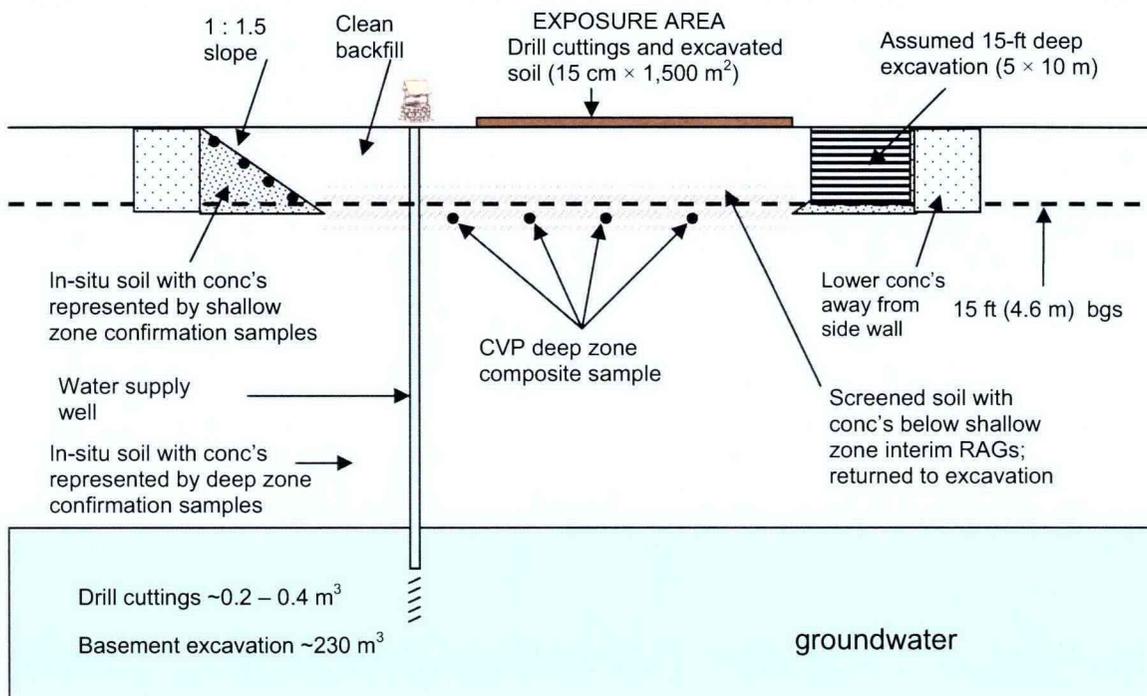


Figure 5-3. Plot of Henry's Constant vs. Stripping Efficiency for VOCs.

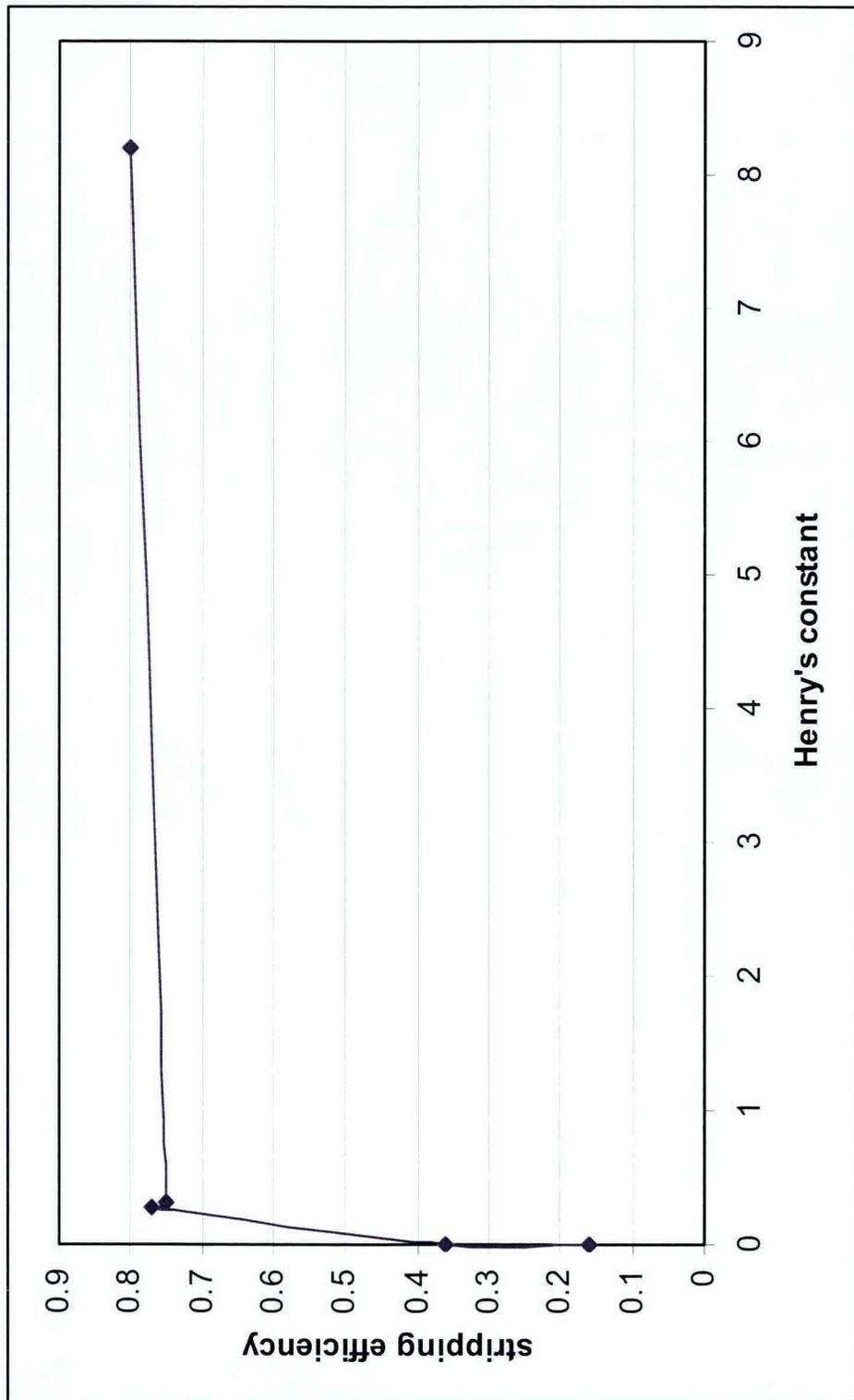


Figure 5-4. Protocol for Selection of Route-Specific Toxicity Criteria for Organic Chemicals.

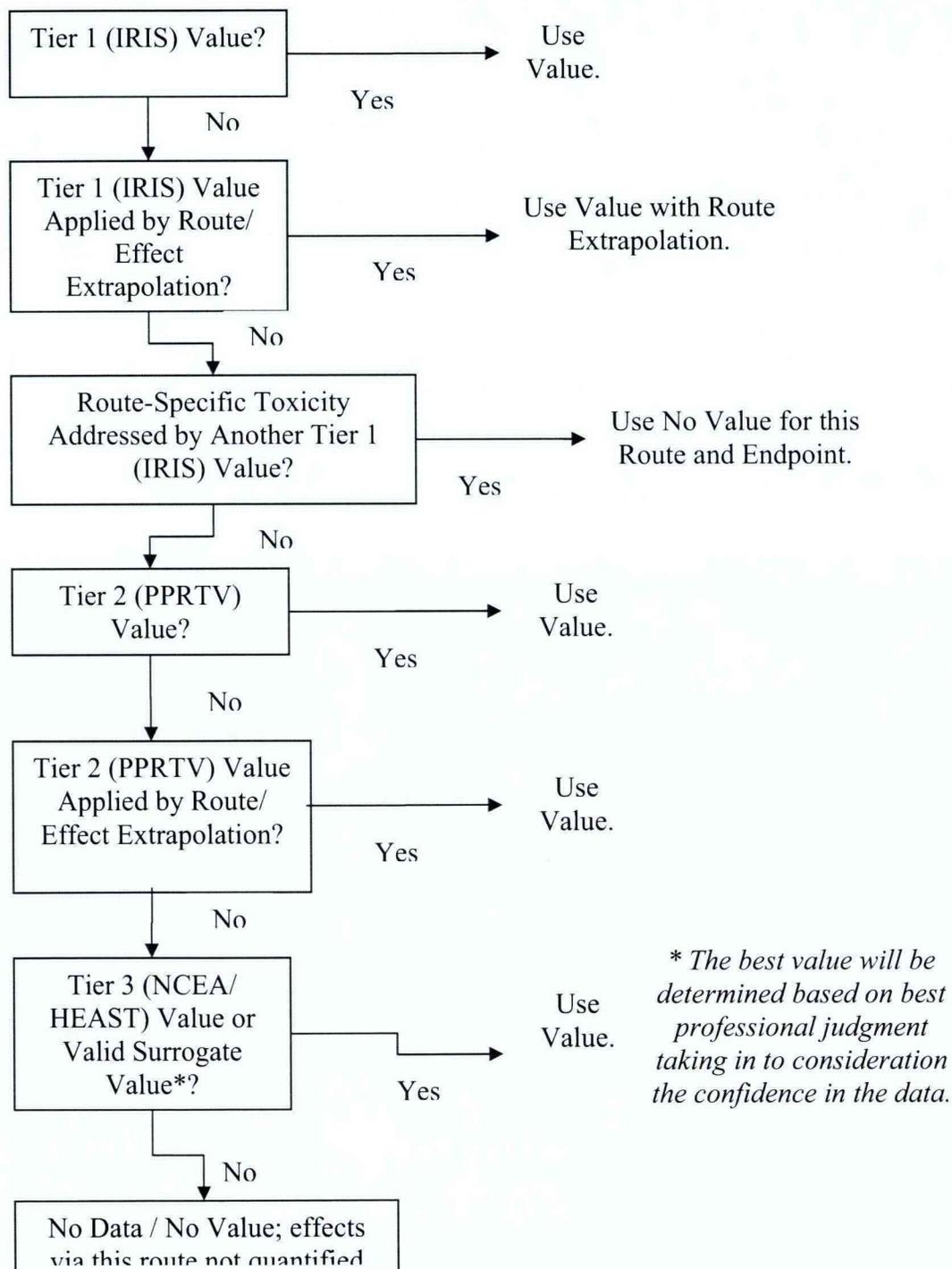


Figure 5-5. Protocol for Selection of Route-Specific Toxicity Criteria for Inorganic Chemicals.

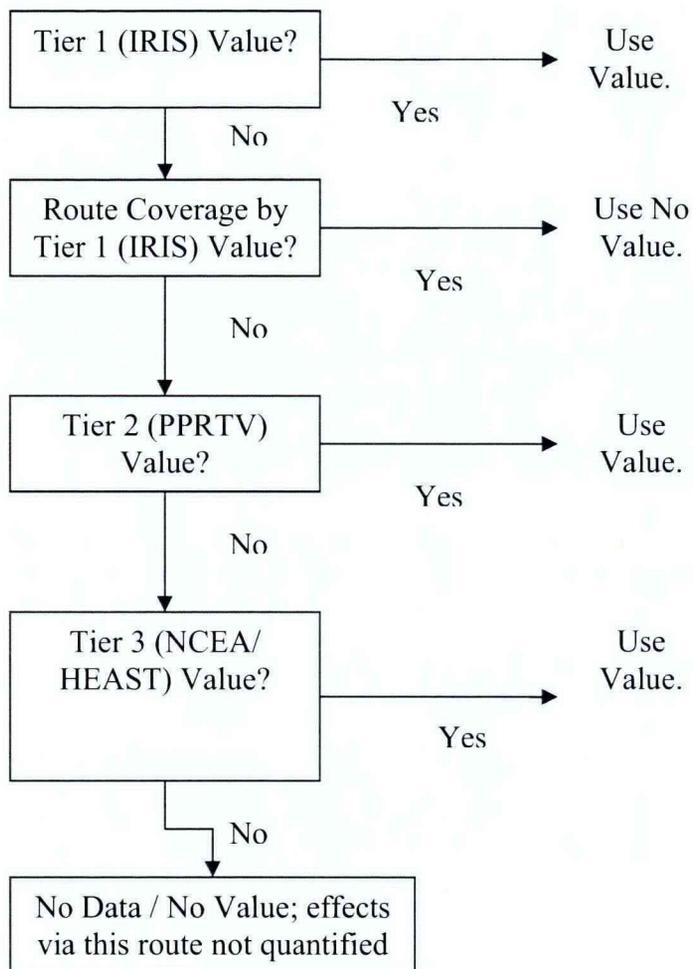
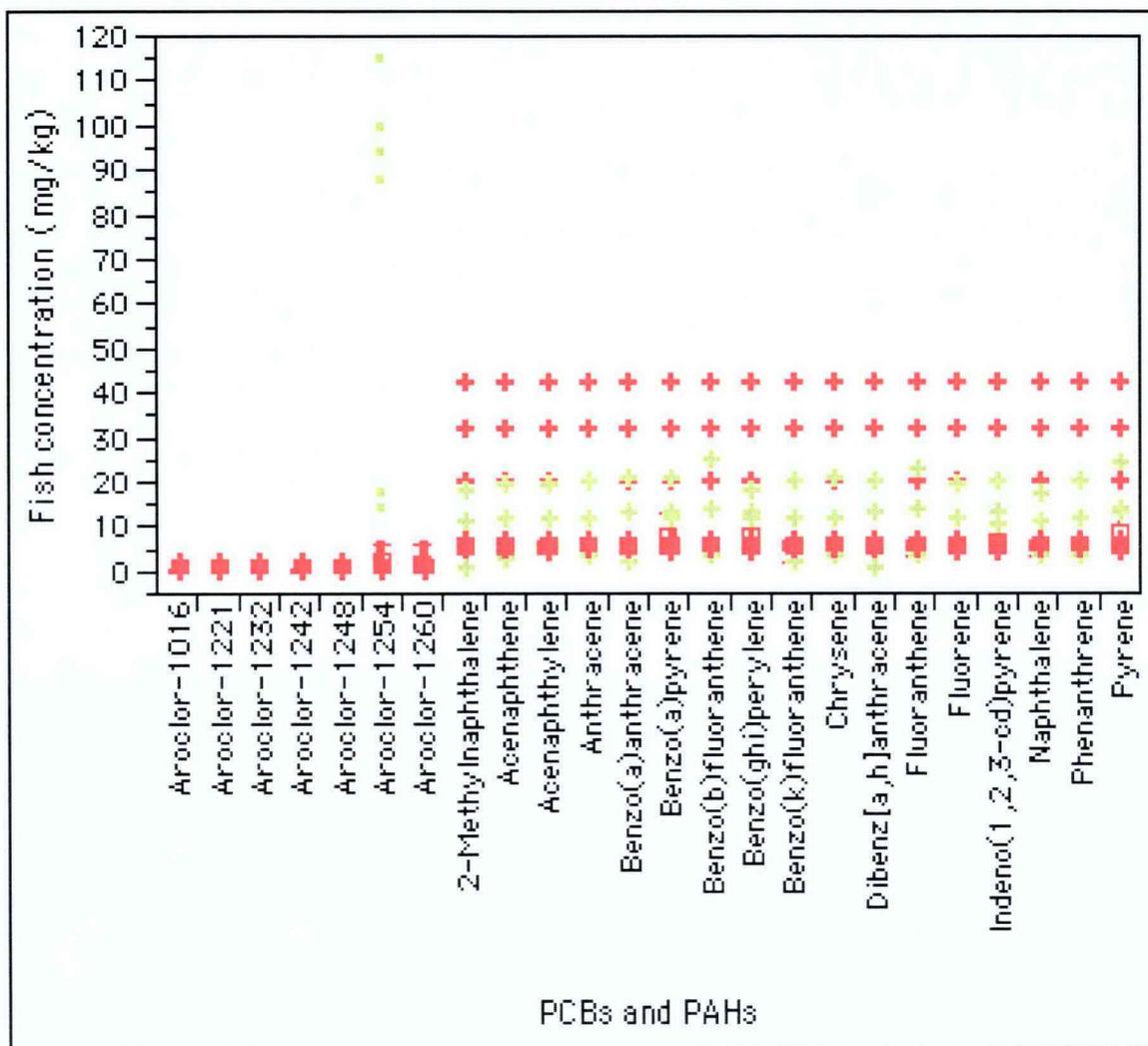


Figure 5-6a. Range of PCB and PAH Values in Fish Tissue Data.



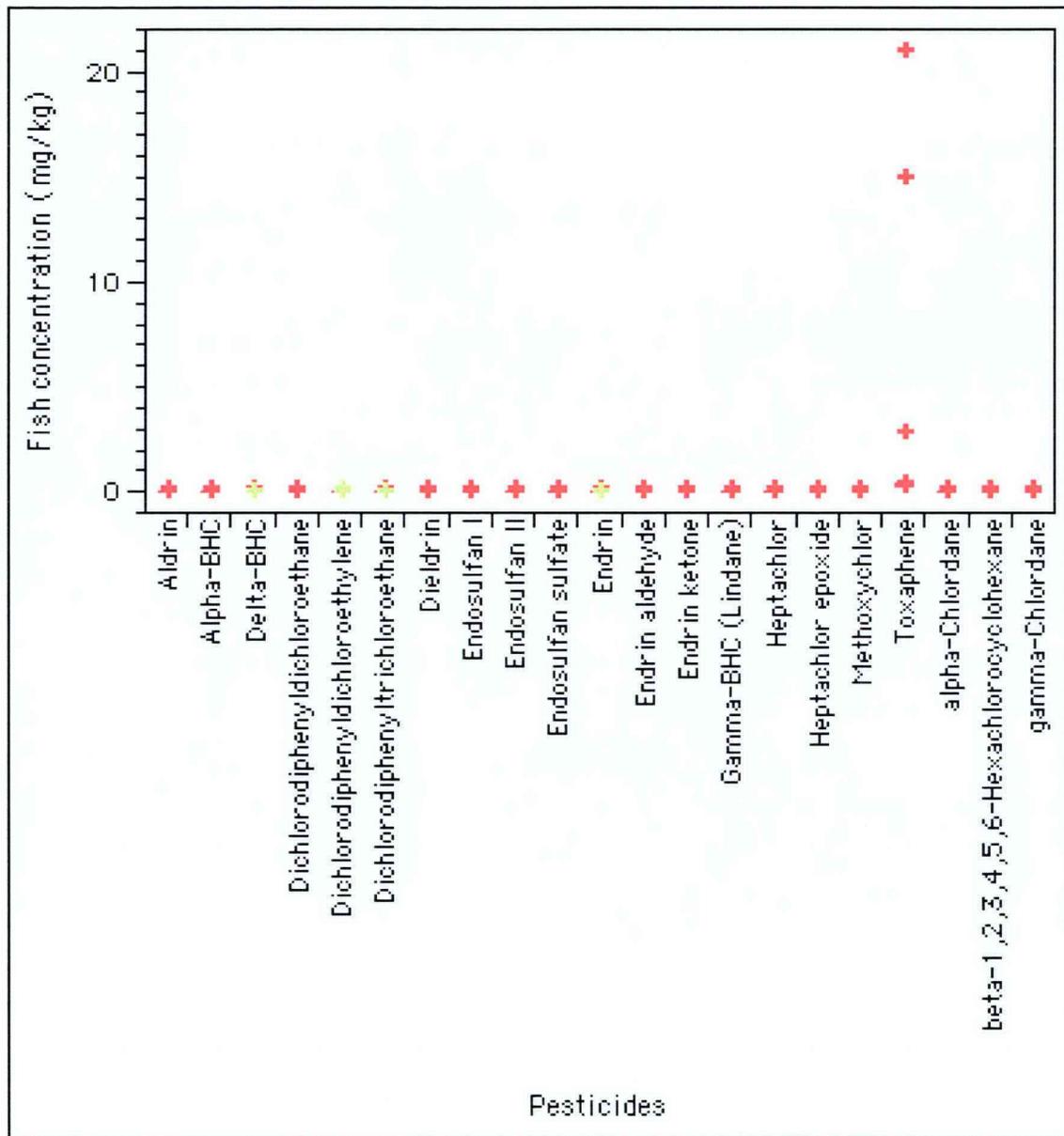
data_source_type

- BC_PILOT
- + RCBRA

detect_status

- FALSE
- TRUE

Figure 5-6b. Range of Pesticide Values in Fish Tissue Data.



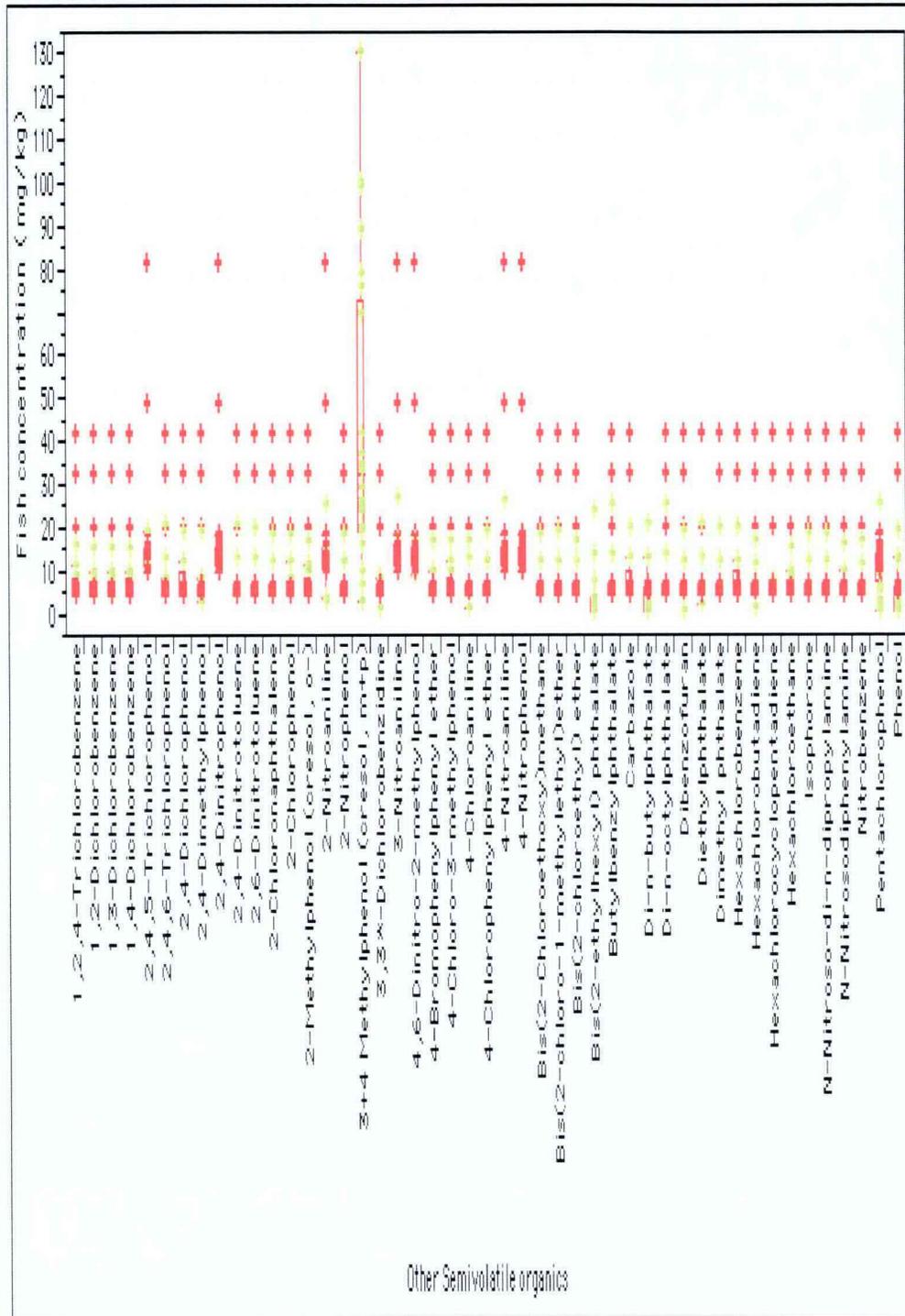
data_source_type

- BC_PILOT
- + RCBRA

detect_status

- FALSE
- TRUE

Figure 5-6c. Range of SVOC Values in Fish Tissue Data.



data_source_type

- BC_PILOT
- + RCBRA

detect_status

- FALSE
- TRUE

Figure 5-7. Rural Residential Scenario Cancer Risks; All Waste Sites.

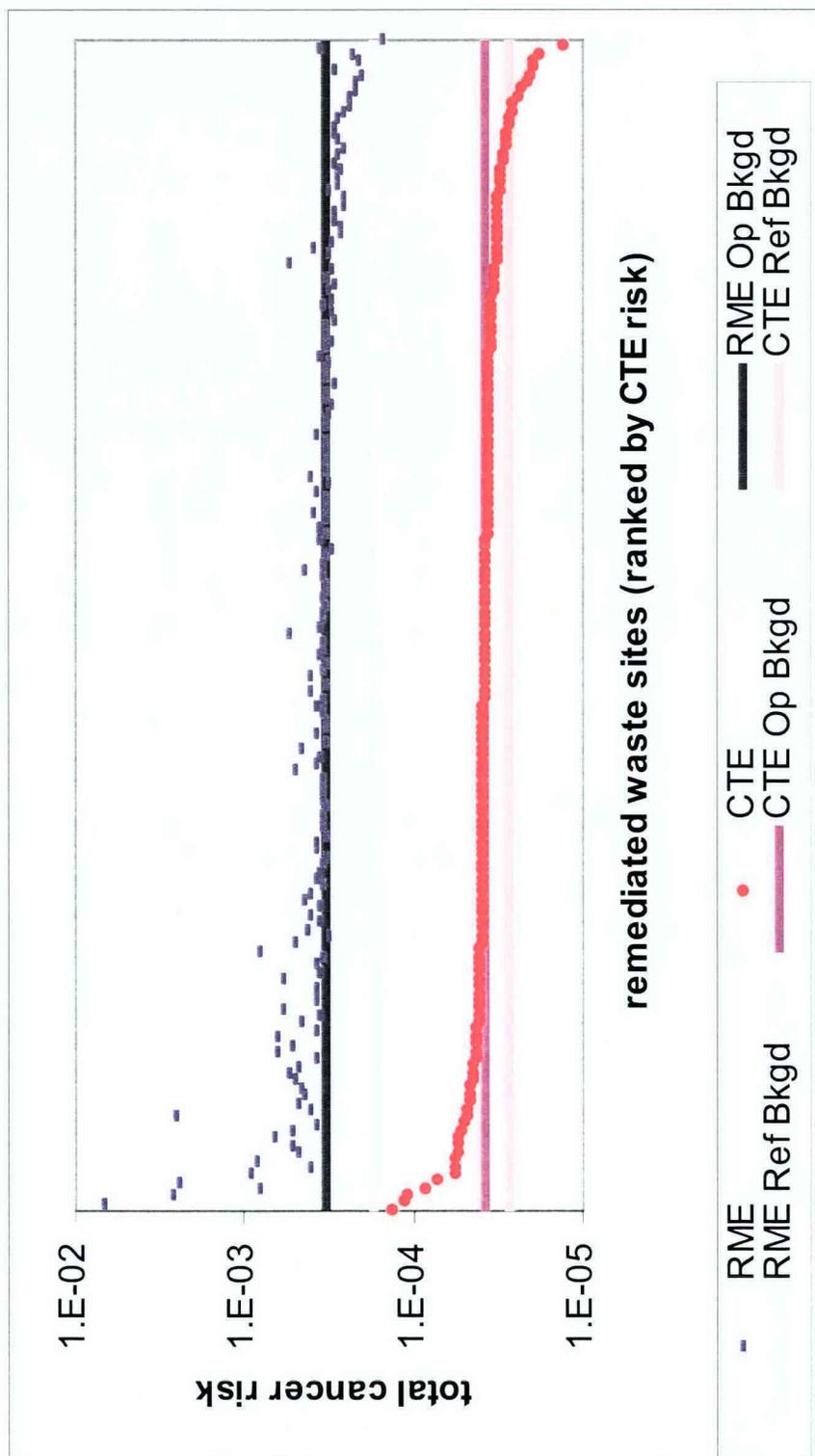


Figure 5-8. Rural Residential Scenario Total Radiation Dose; All Waste Sites.

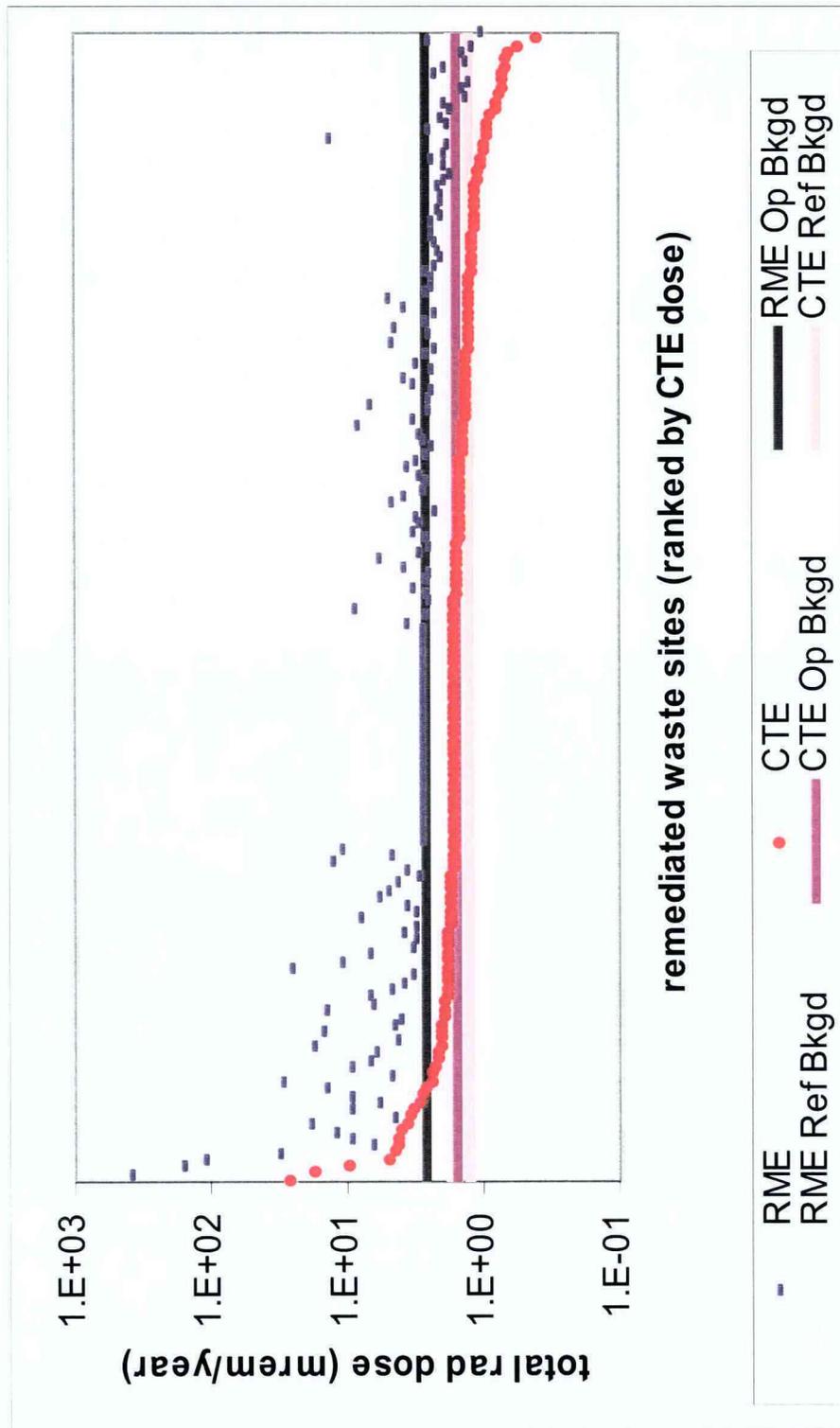


Figure 5-9. Rural Residential Scenario Total Child Hazard Index; All Waste Sites.

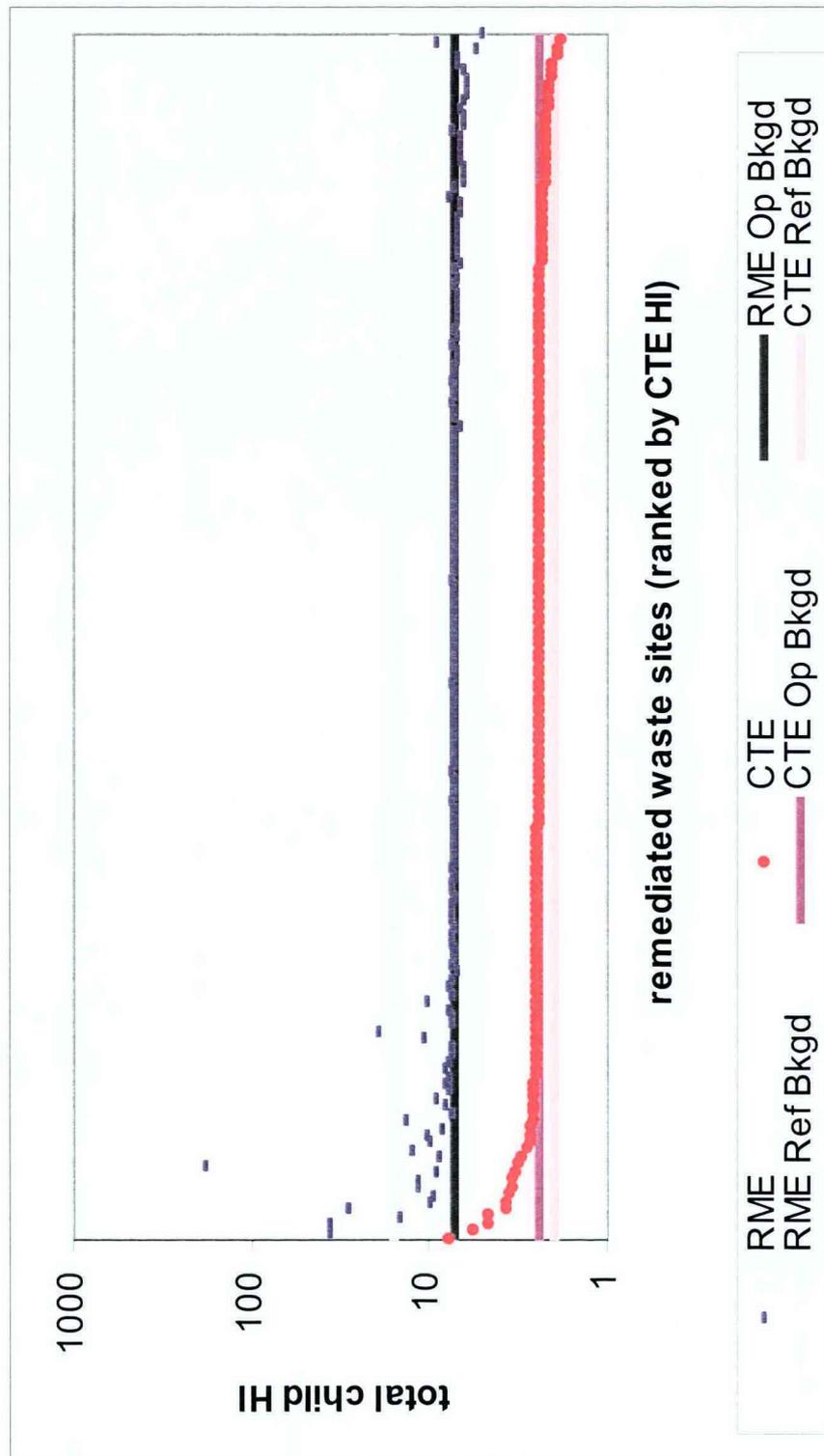


Figure 5-10. CTUIR Scenario “Local Area Only” Cancer Risks; All Waste Sites.

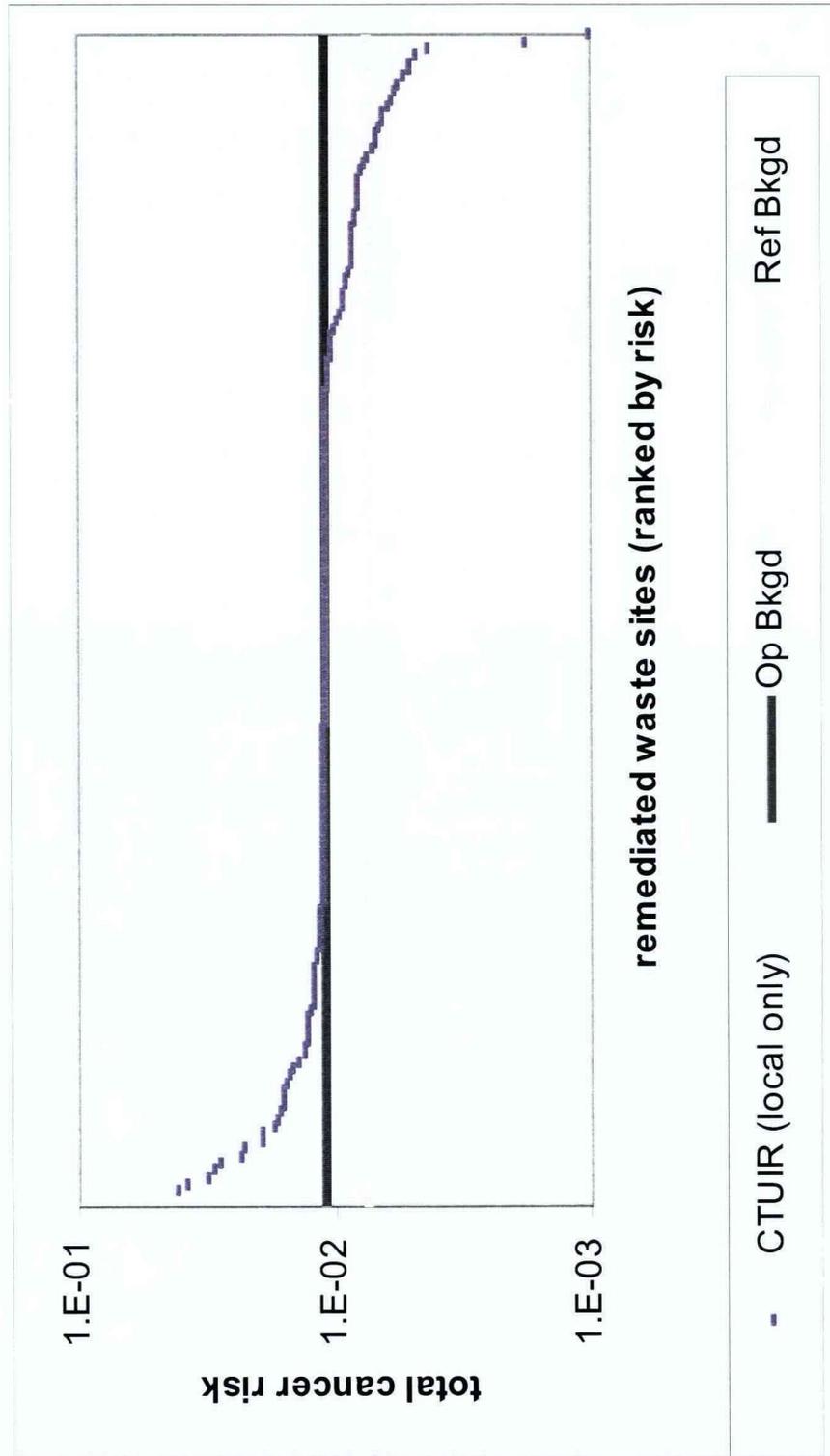


Figure 5-11. CTUIR Scenario “Local and Broad Areas” Cancer Risks; All Waste Sites.

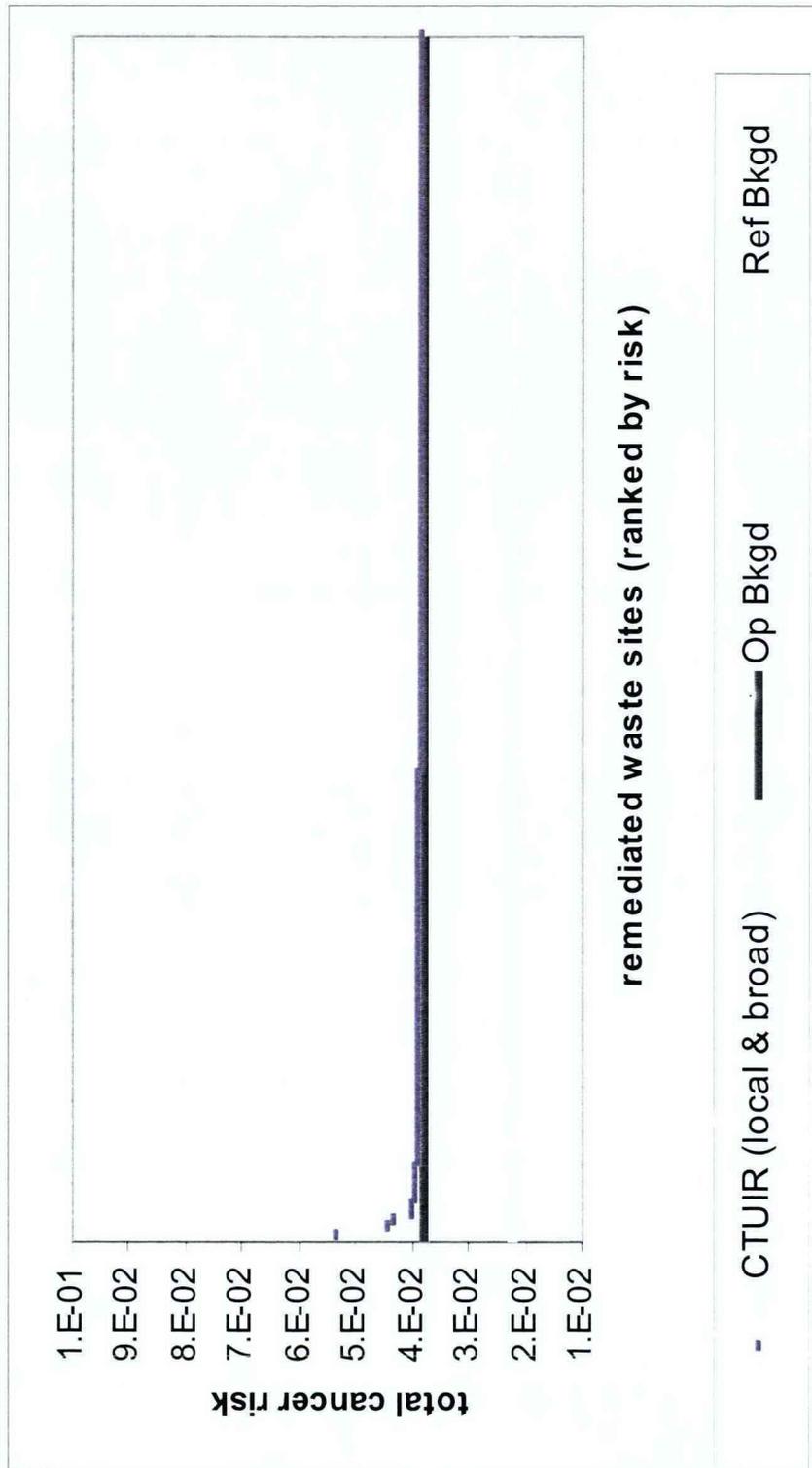


Figure 5-12. CTUIR Scenario “Local Area Only” Total Radiation Dose; All Waste Sites.

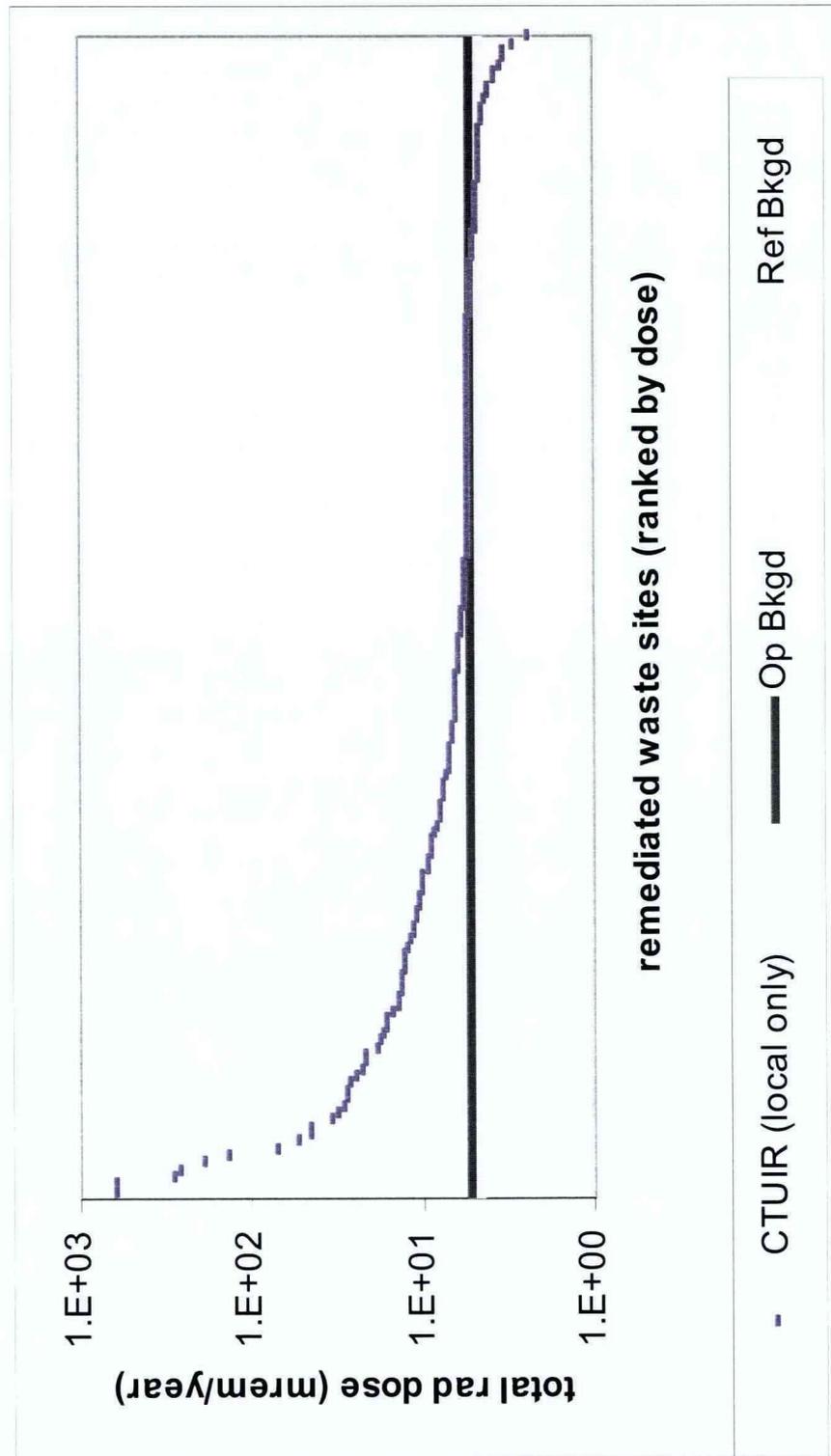


Figure 5-13. CTUIR Scenario “Local and Broad Areas” Total Radiation Dose; All Waste Sites.

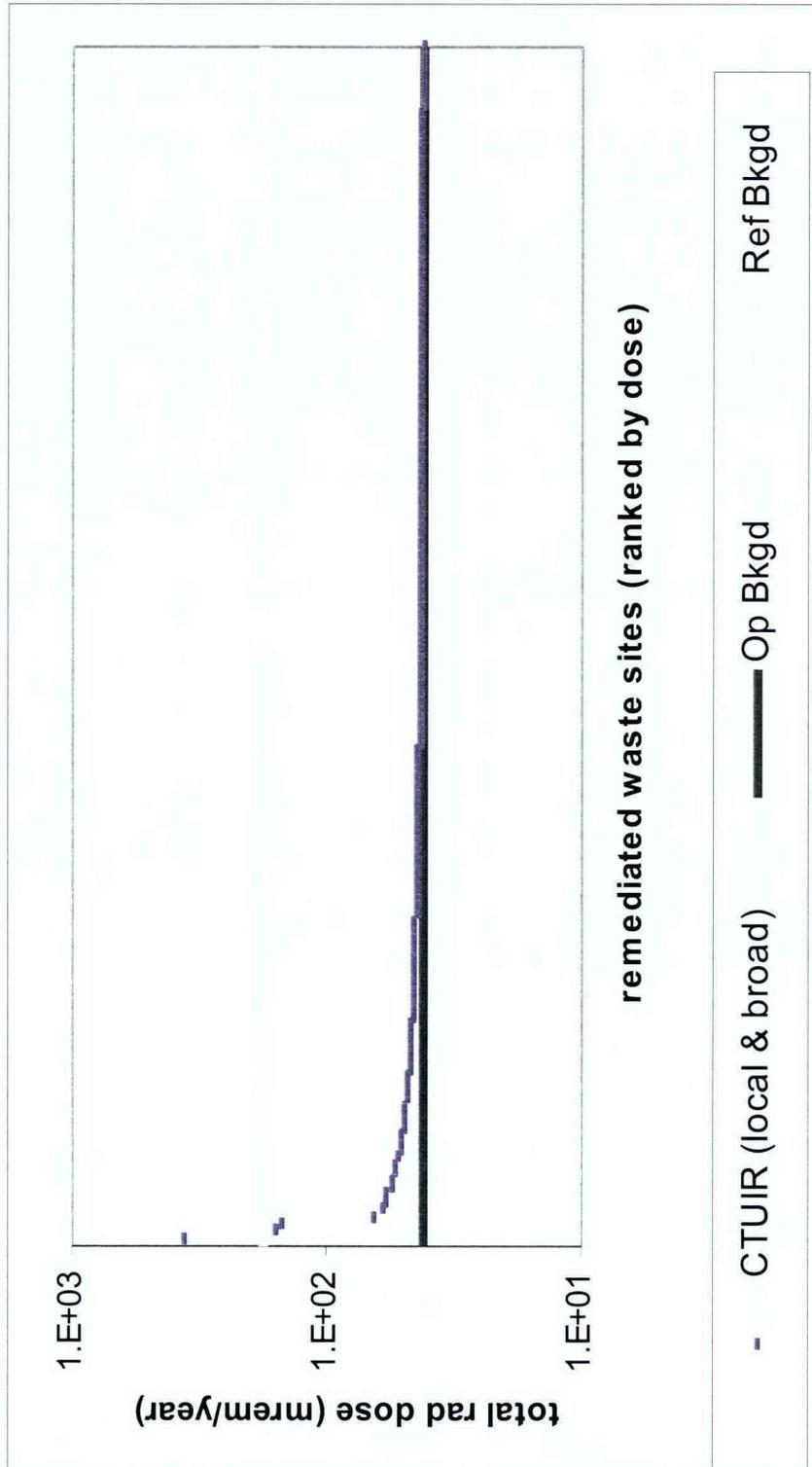


Figure 5-14. CTUIR Scenario “Local Area Only” Total Child Hazard Index; All Waste Sites.

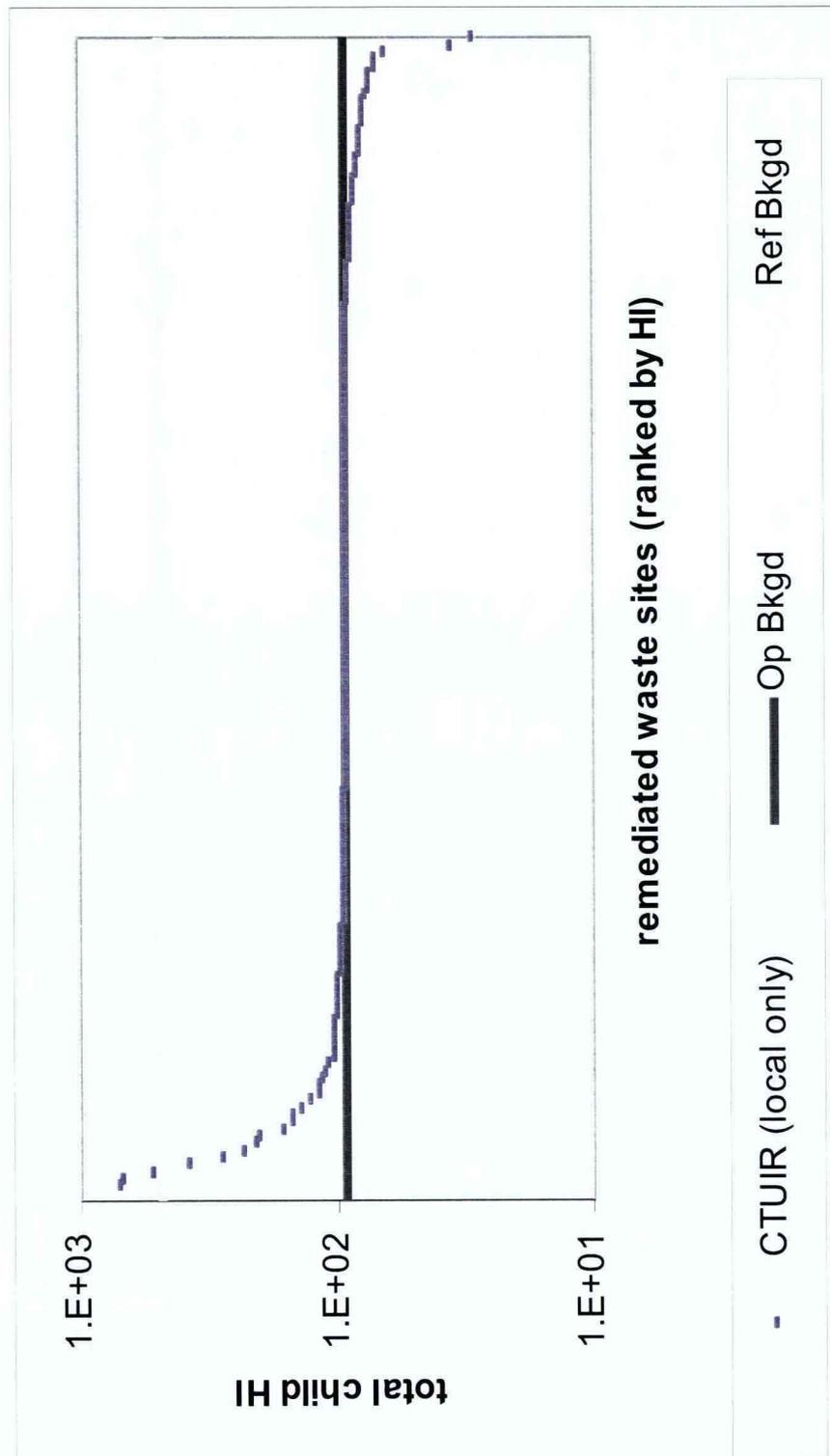


Figure 5-15. CTUIR Scenario “Local and Broad Areas” Total Child Hazard Index; All Waste Sites.

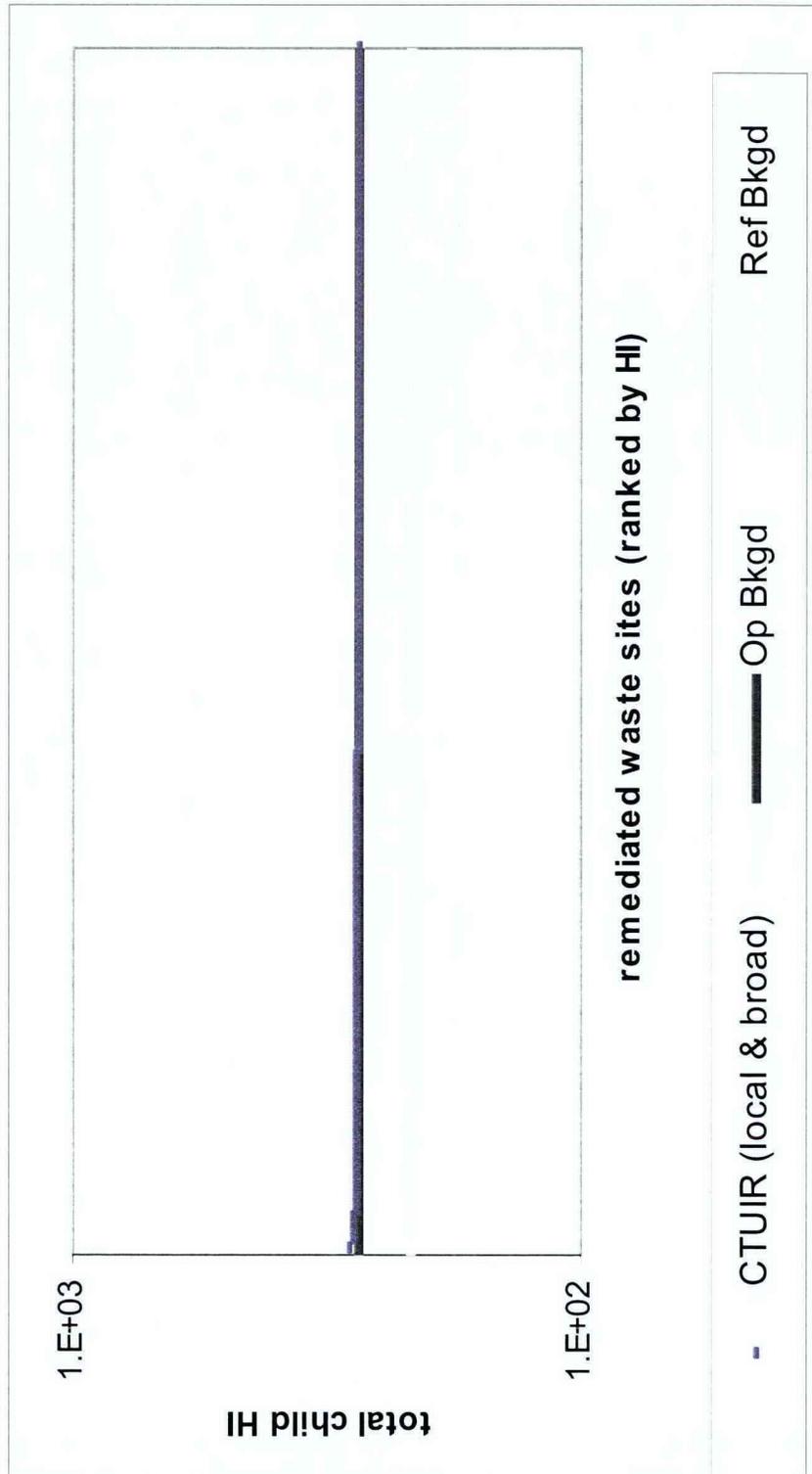


Figure 5-16. Resident Monument Worker Scenario Cancer Risks; All Waste Sites.

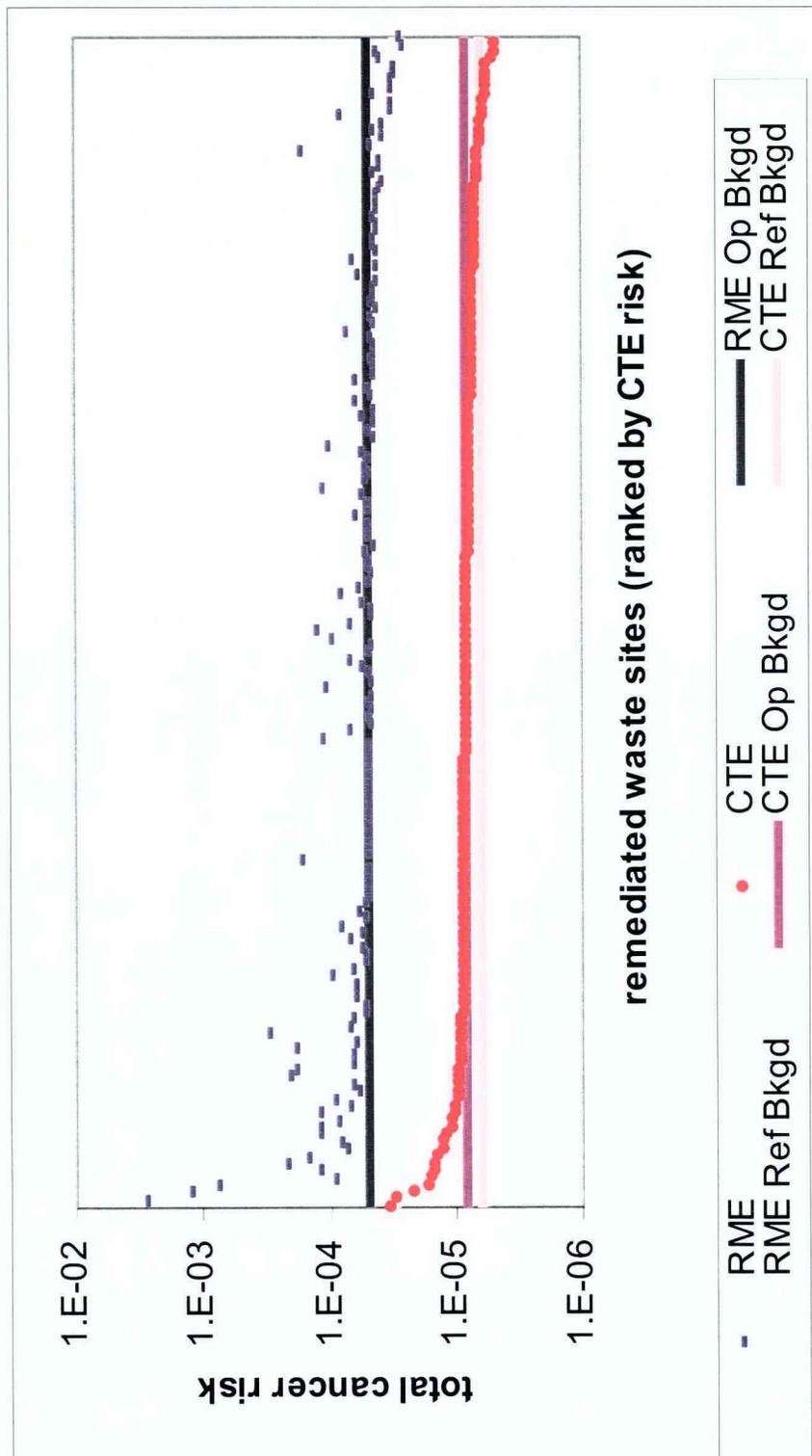


Figure 5-17. Resident Monument Worker Scenario Total Radiation Dose; All Waste Sites.

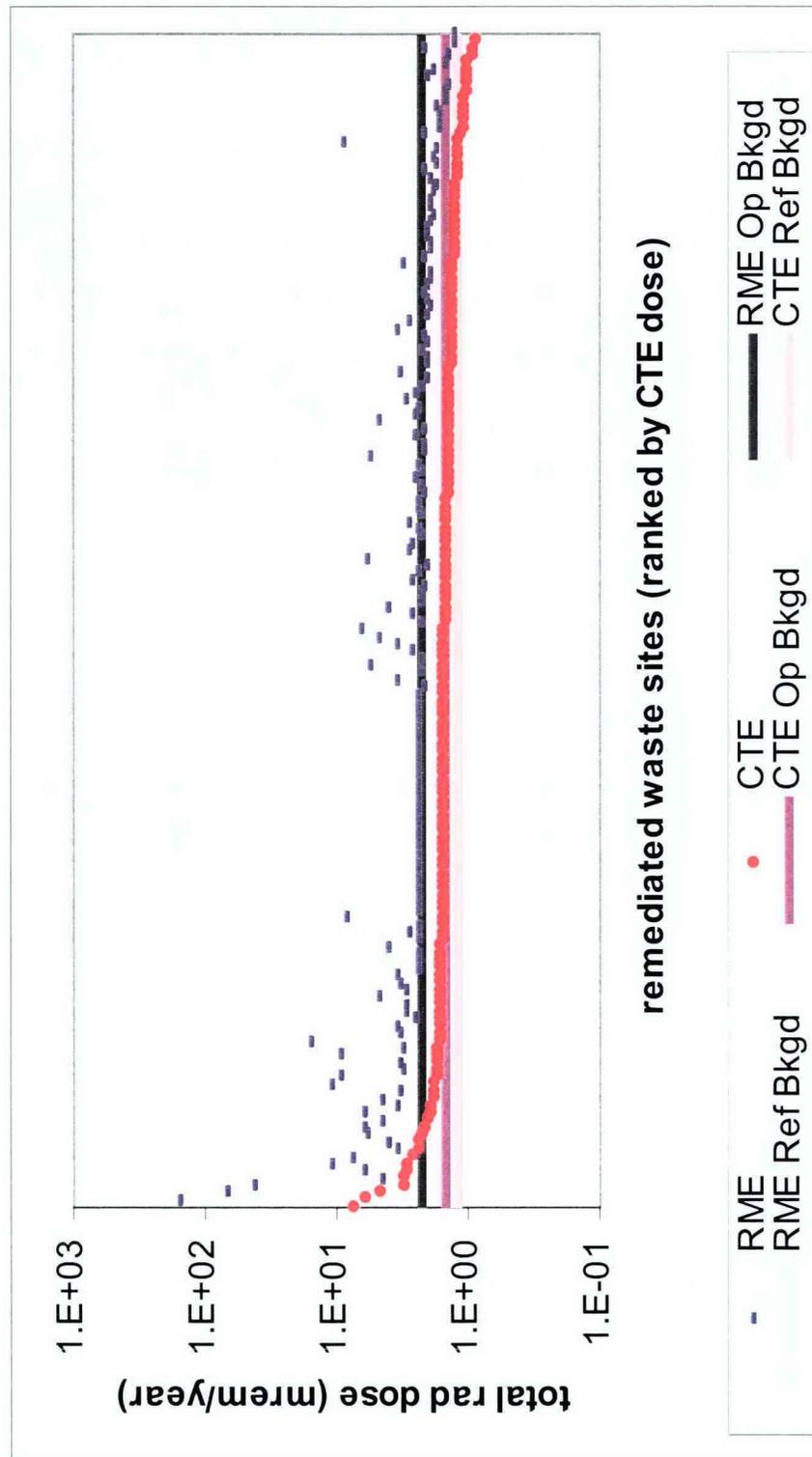


Figure 5-18. Resident Monument Worker Scenario Total Hazard Index; All Waste Sites.

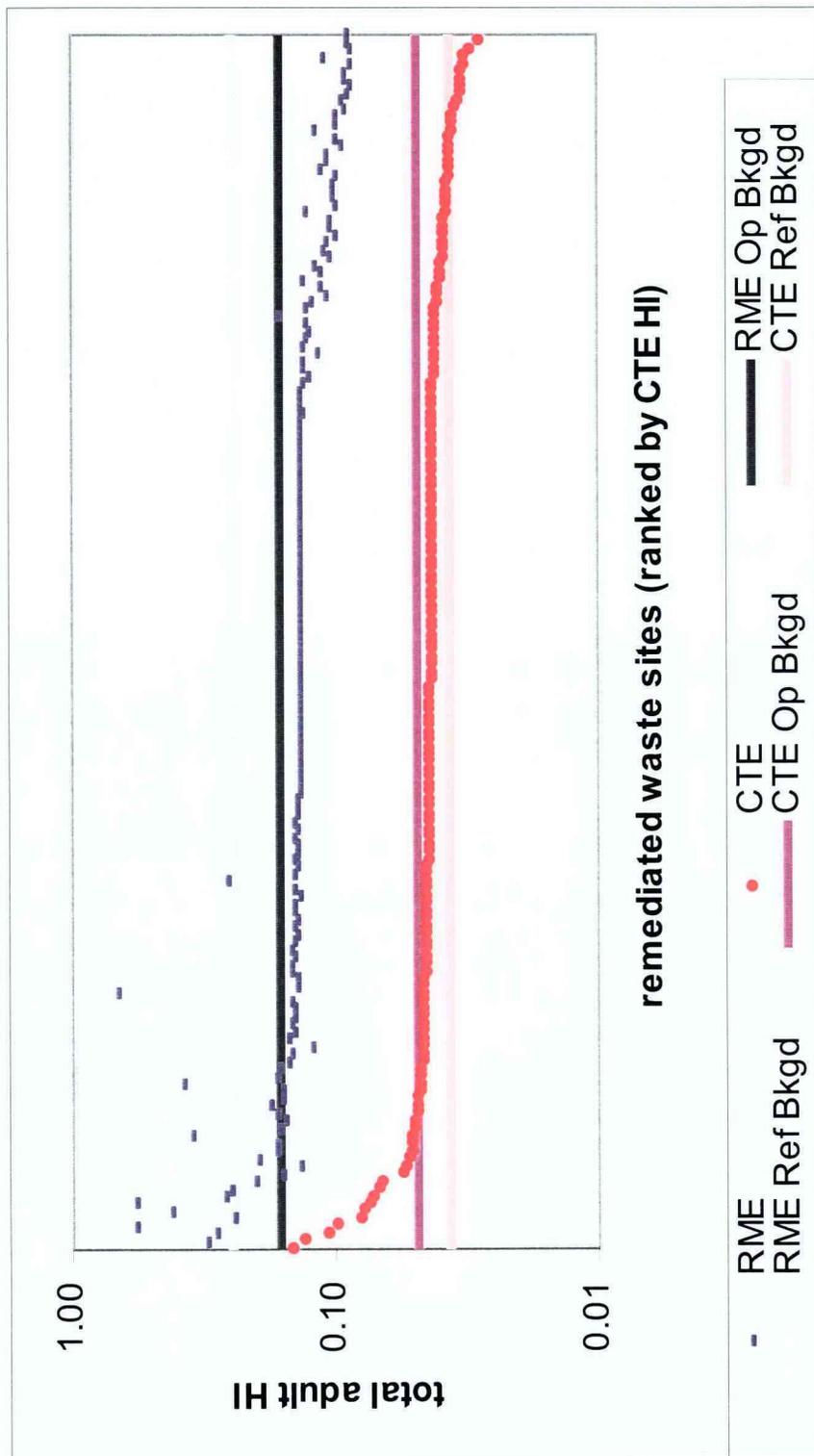


Figure 5-19. Industrial / Commercial Scenario Total Cancer Risks; All Waste Sites.

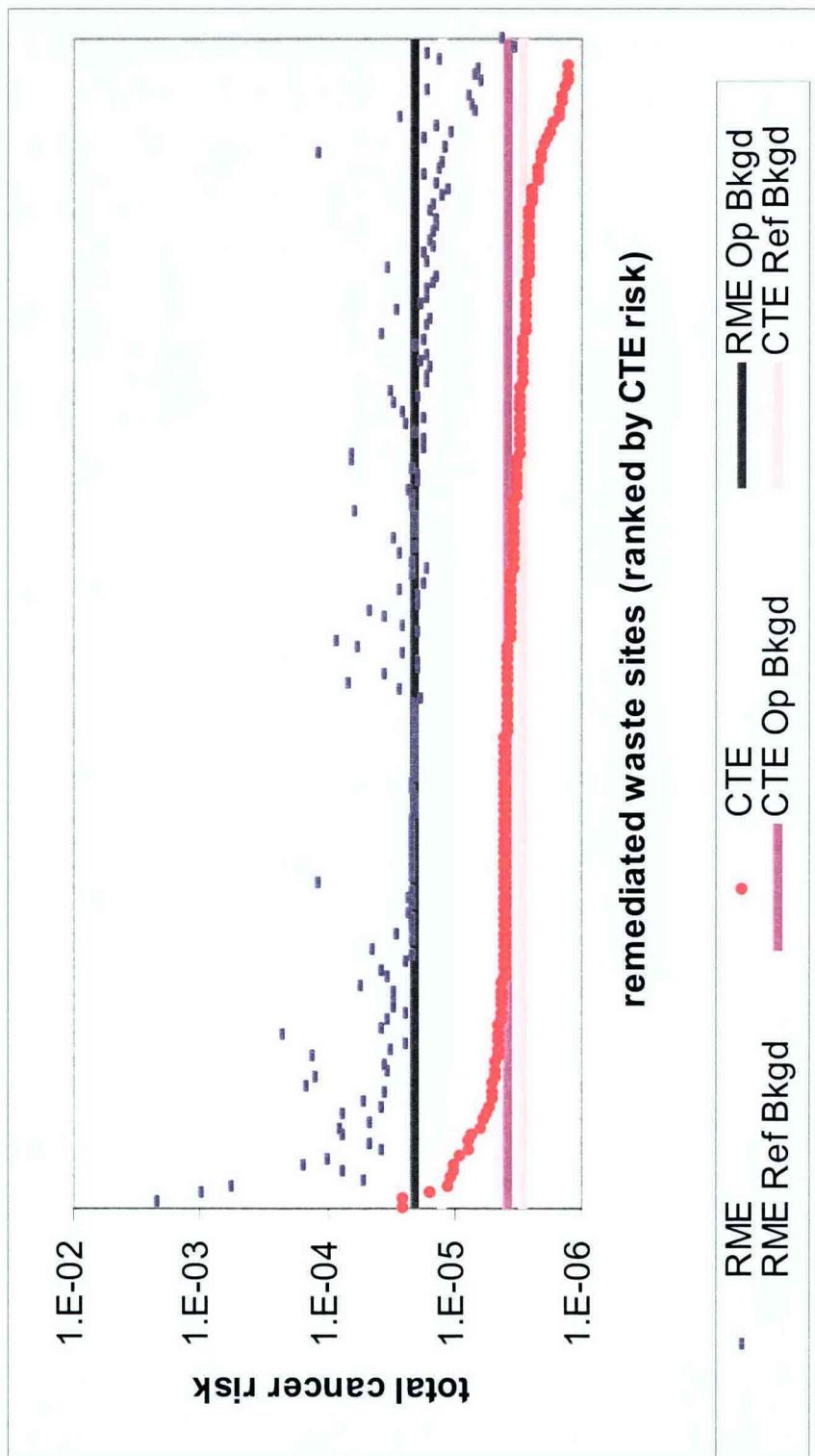


Figure 5-20. Industrial / Commercial Scenario Total Radiation Dose; All Waste Sites.

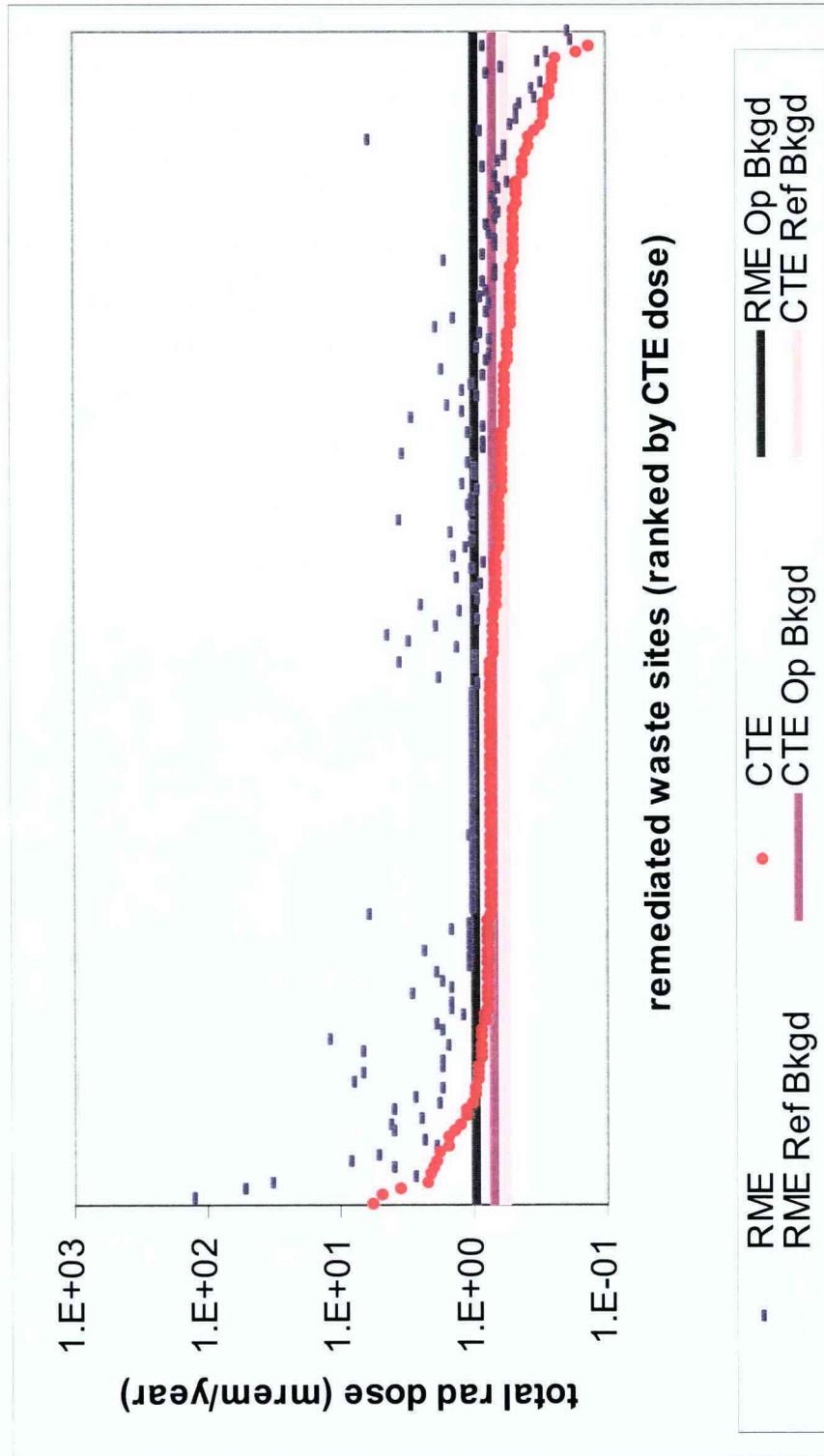


Figure 5-21. Industrial / Commercial Scenario Total Hazard Index; All Waste Sites.

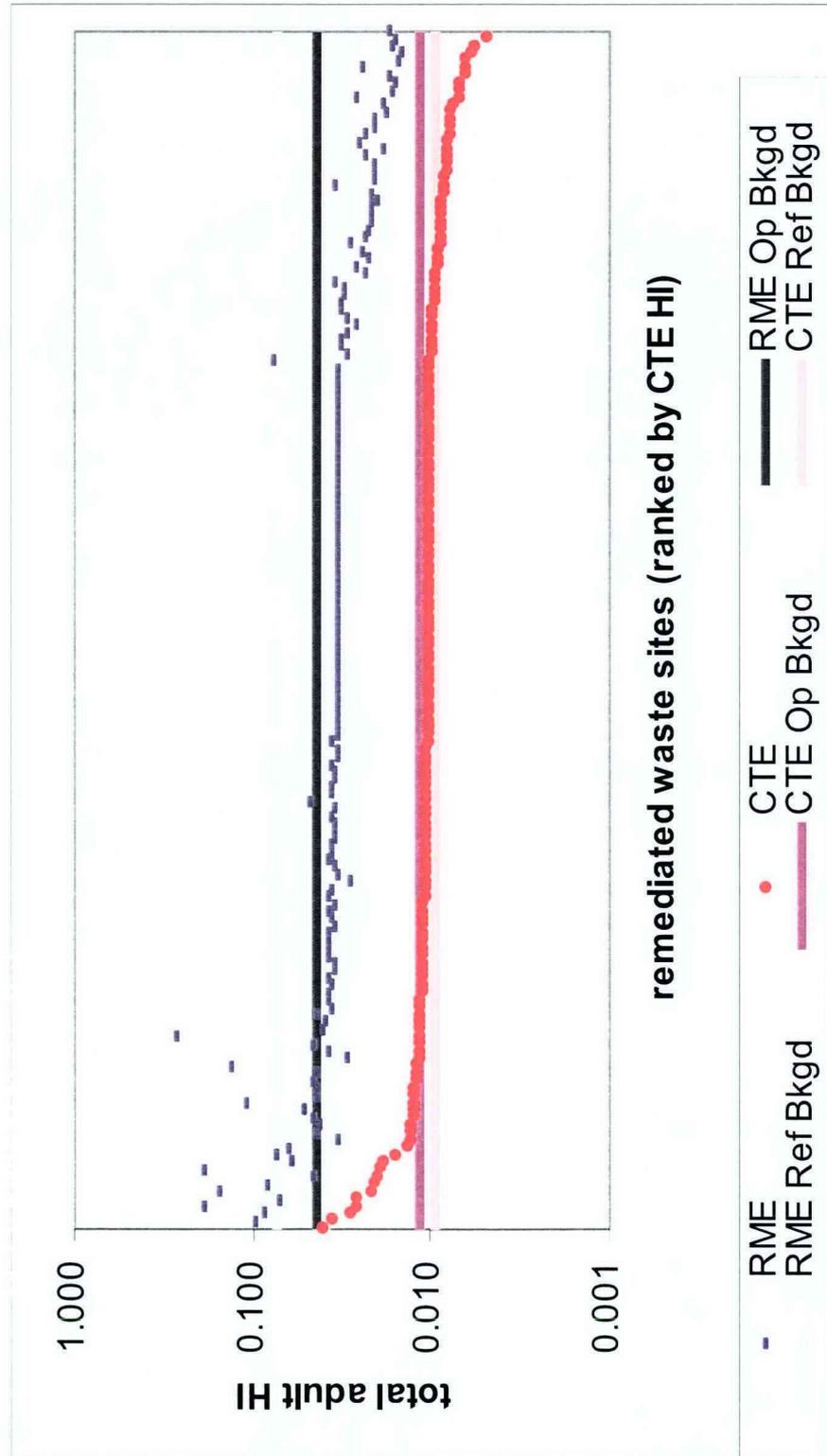


Figure 5-22a. Casual User and Avid Hunter Total Cancer Risk.

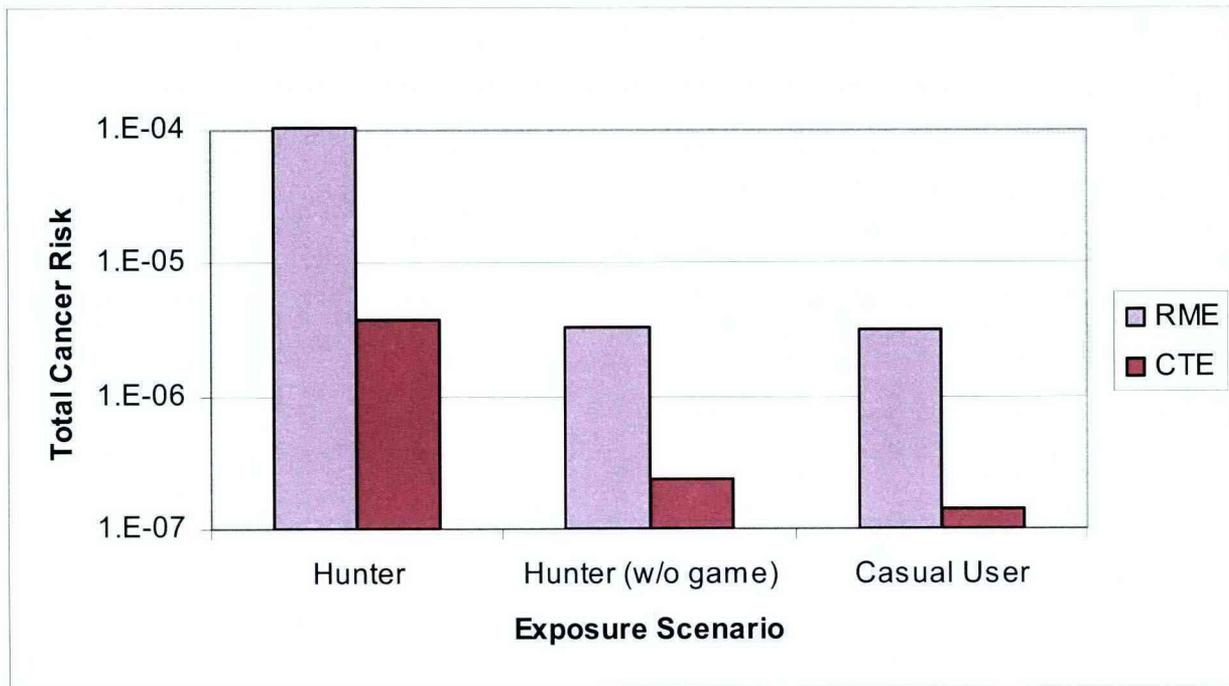


Figure 5-22b. Casual User and Avid Hunter Background Cancer Risk.

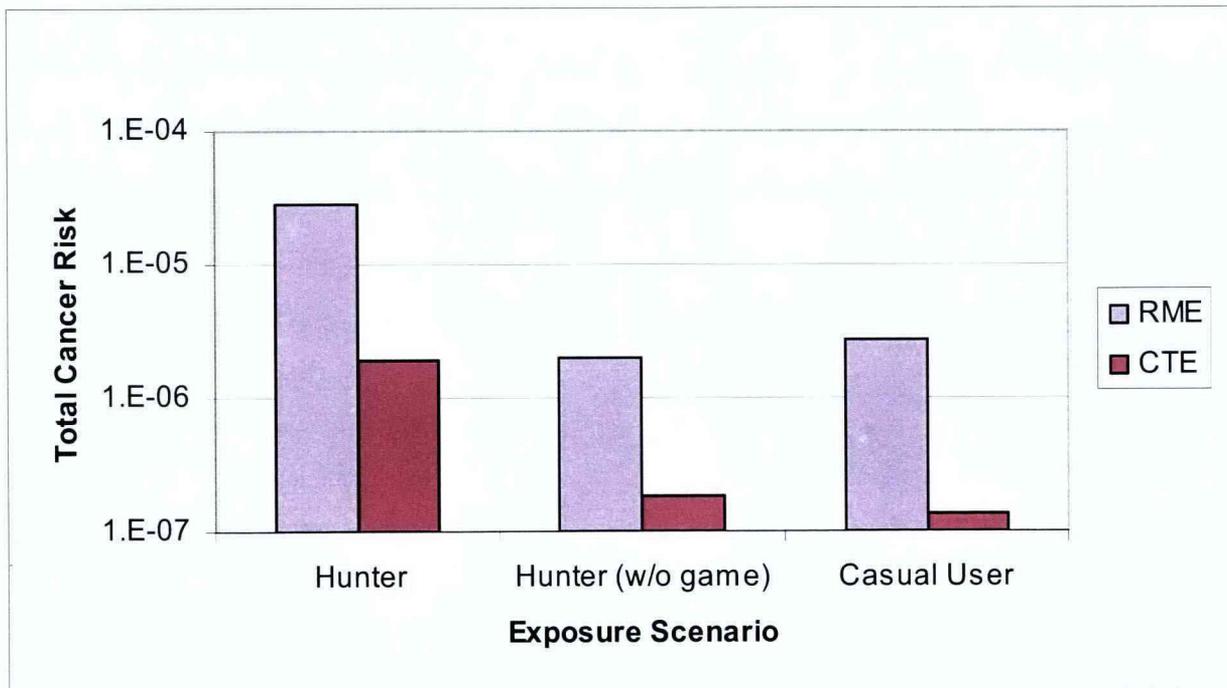


Figure 5-23. Avid Angler Cancer Risks for Sediment Exposures.

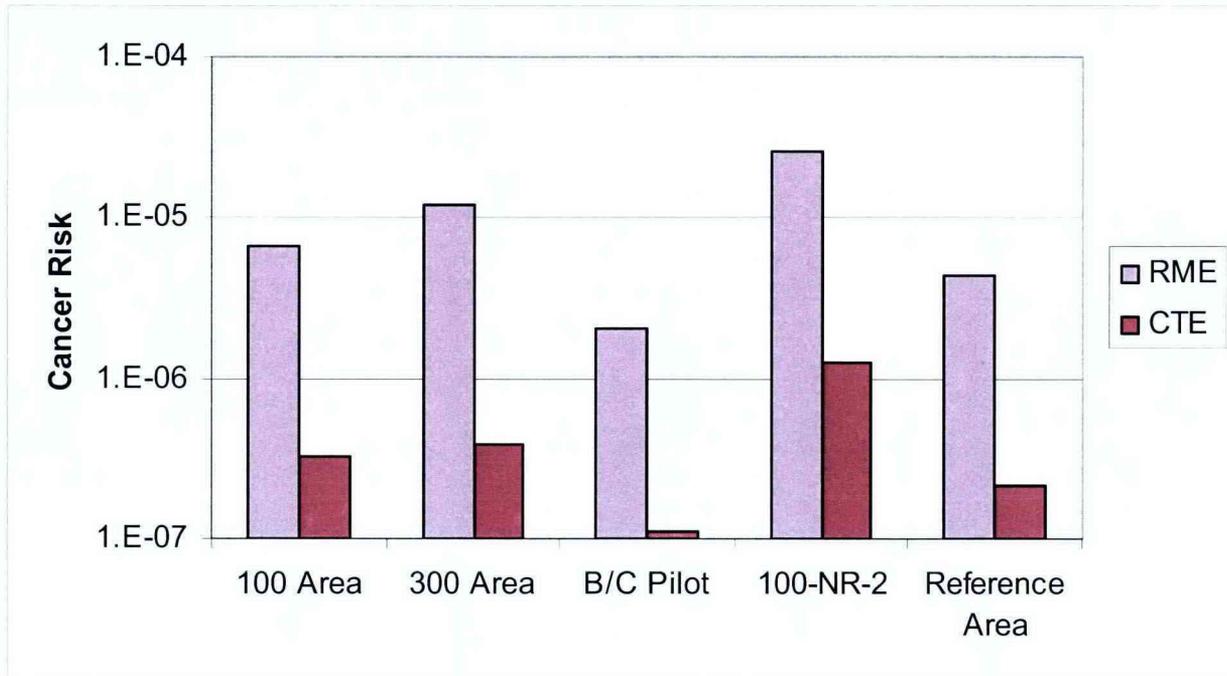


Figure 5-24a. Casual User and Avid Hunter Total Radiation Dose.

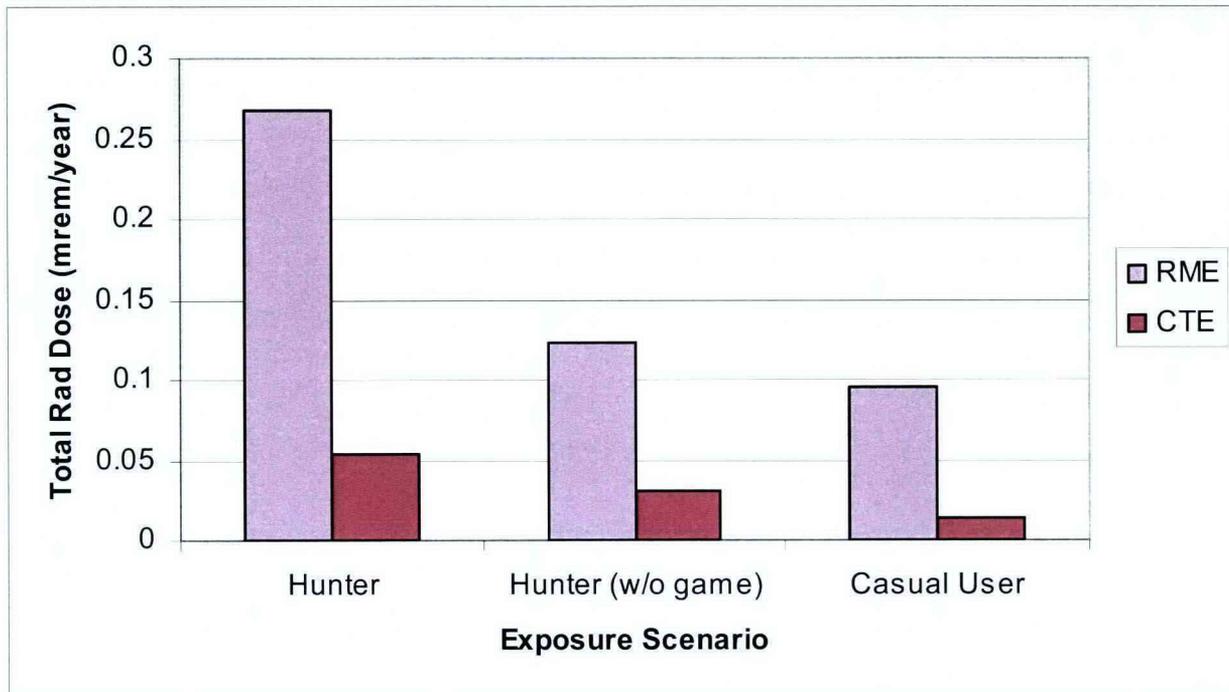


Figure 5-24b. Casual User and Avid Hunter Background Radiation Dose.

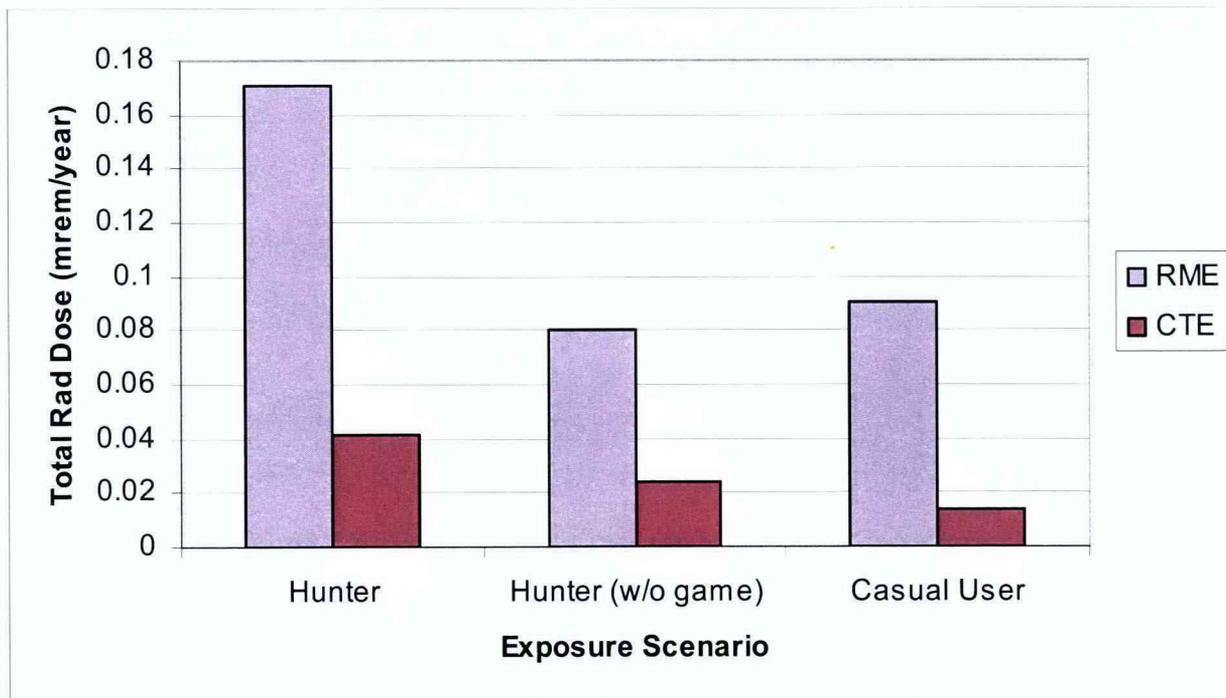


Figure 5-25. Avid Angler Radiation Doses for Sediment Exposures.

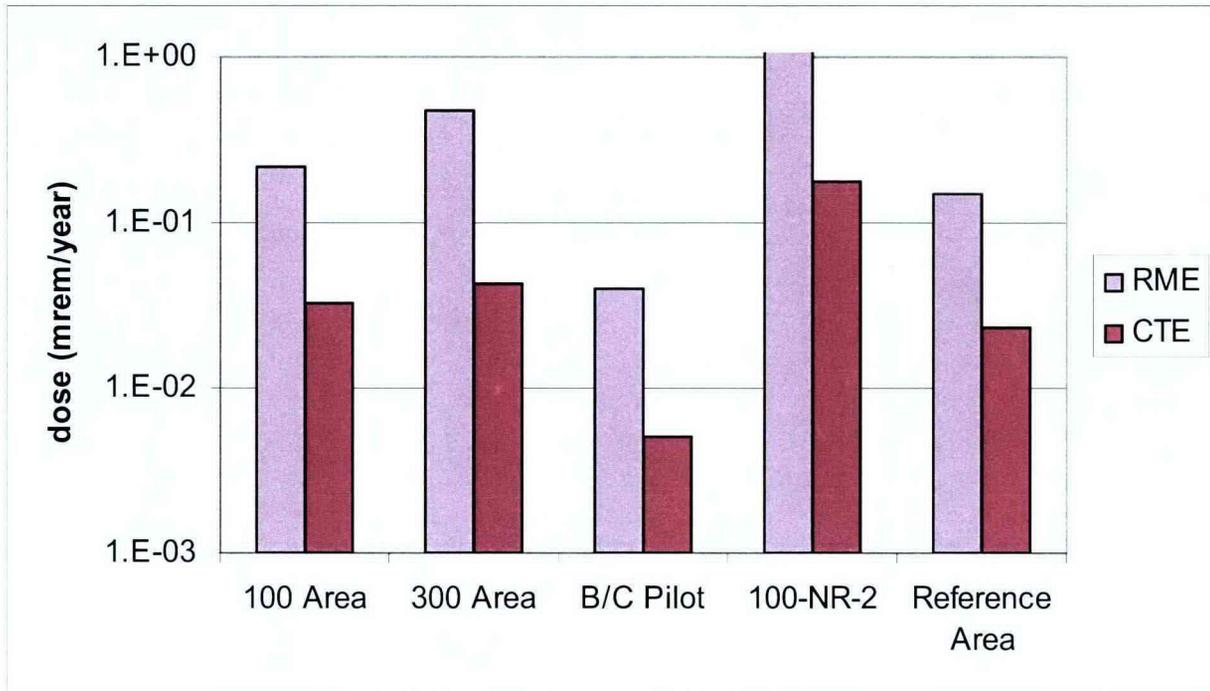


Figure 5-26a. Casual User and Avid Hunter Total Child Hazard Index.

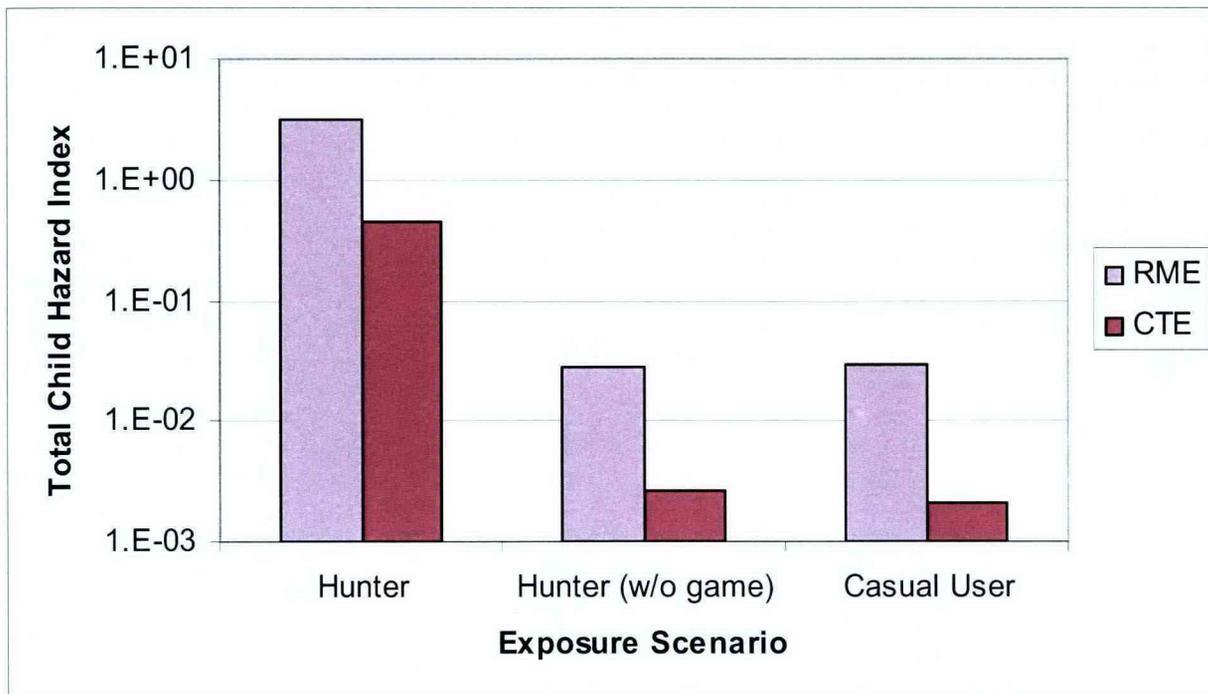


Figure 5-26b. Casual User and Avid Hunter Background Child Hazard Index.

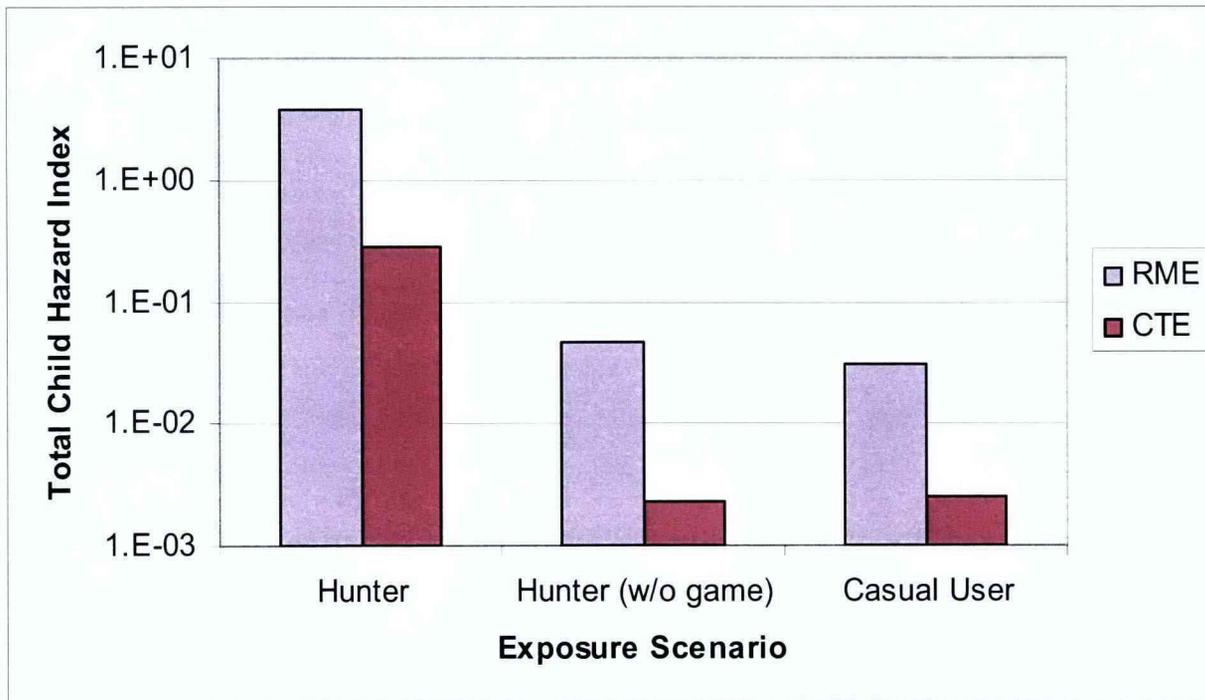


Figure 5-27. Avid Angler Hazard Indices for Sediment Exposures.

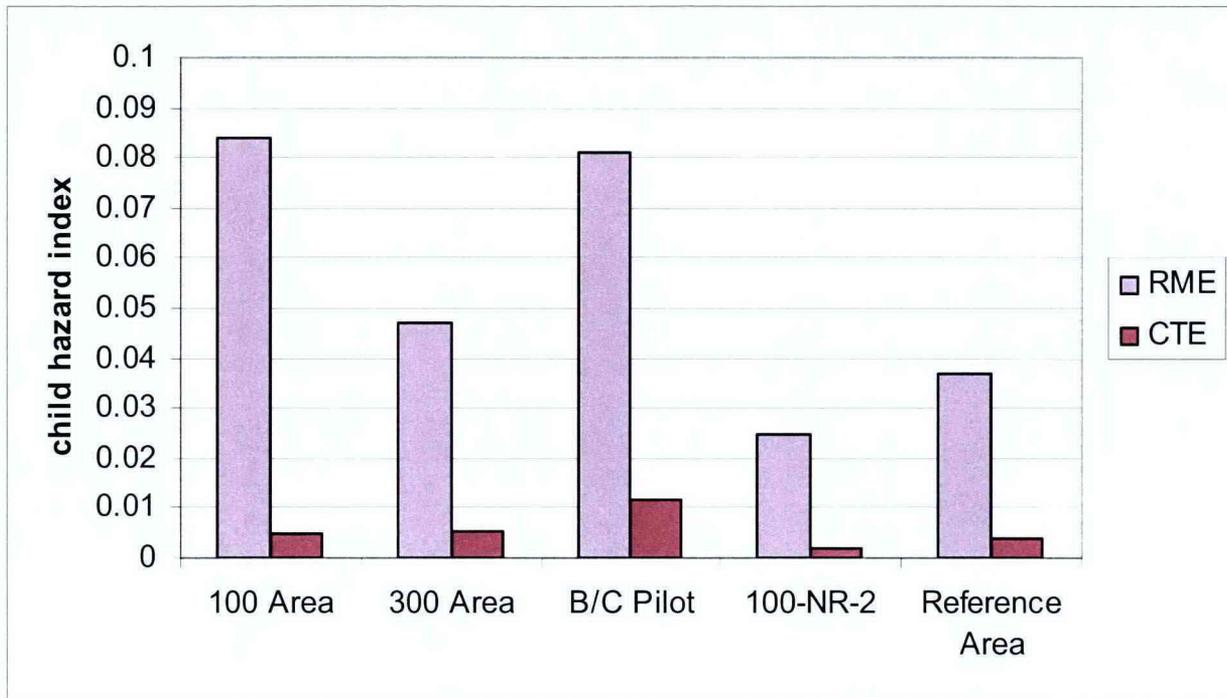


Figure 5-28. Avid Angler Cancer Risks for Fish Ingestion.

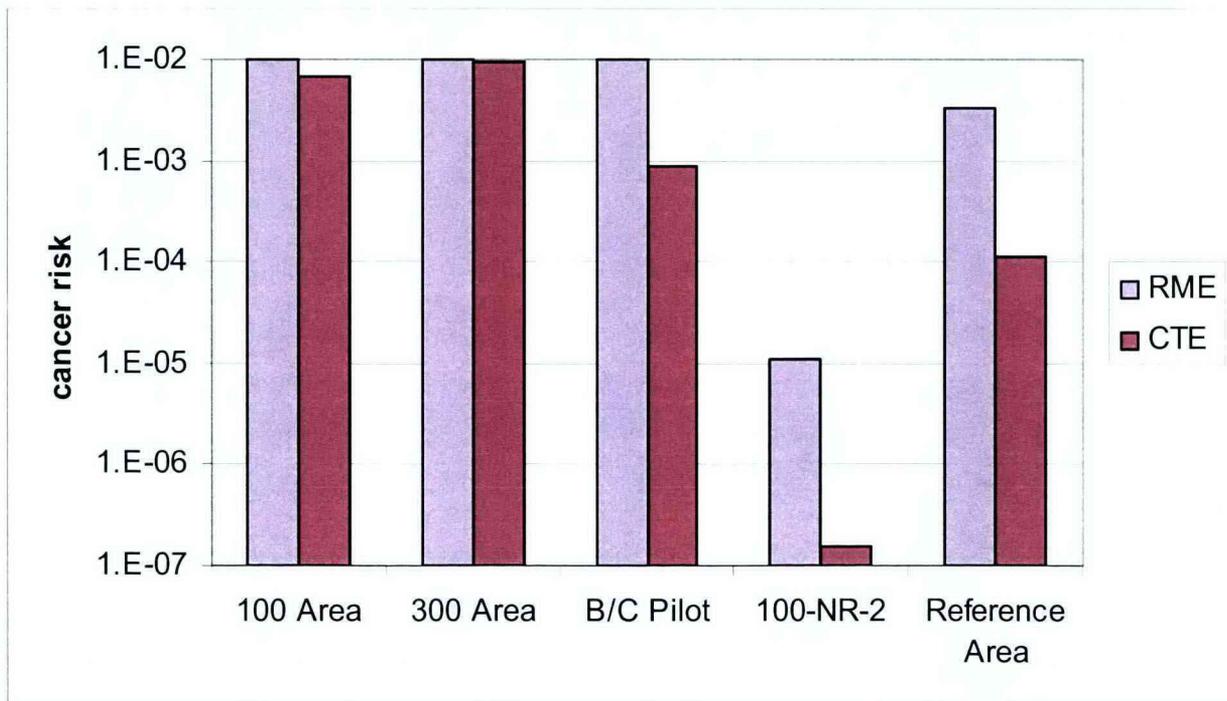


Figure 5-29. Rural Resident Cancer Risks for Fish Ingestion.

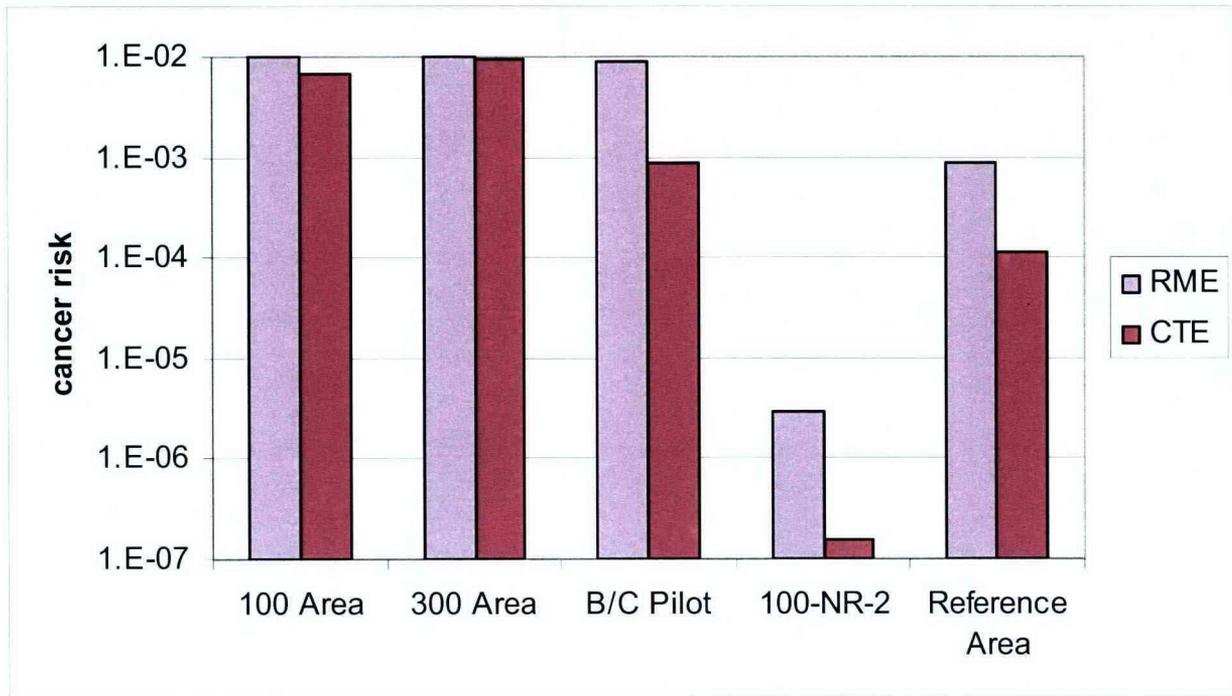


Figure 5-30. CTUIR Cancer Risks for Fish Ingestion.

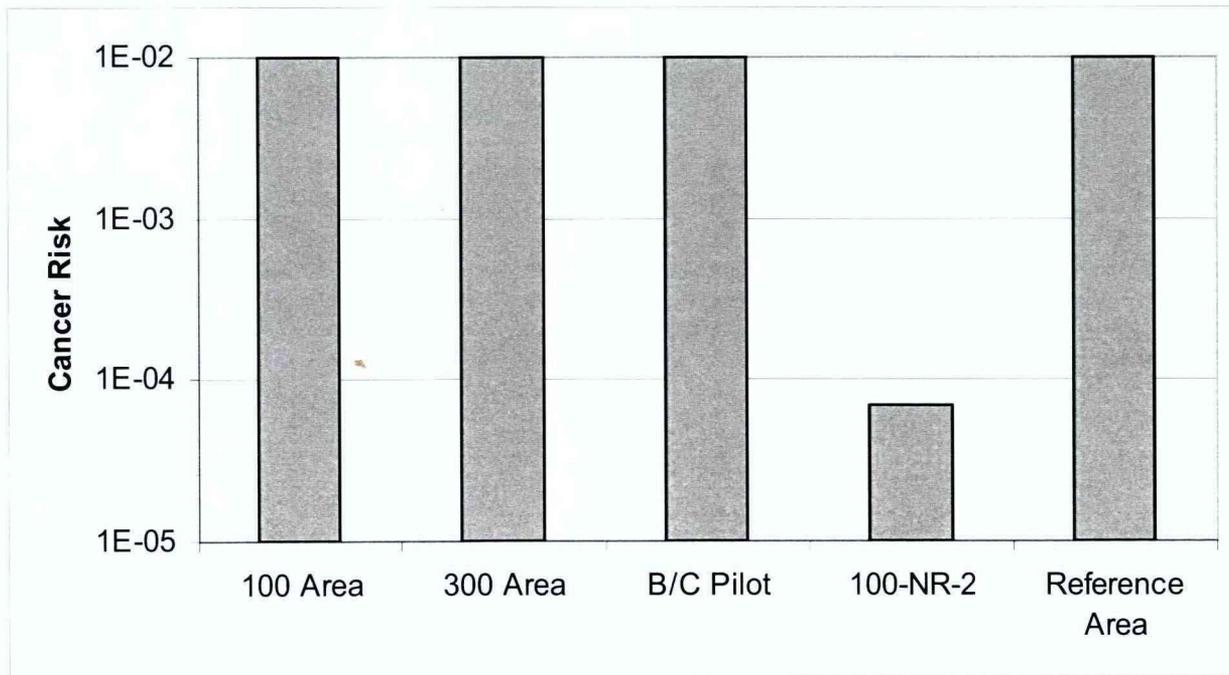


Figure 5-31. Avid Angler Radiation Doses for Fish Ingestion.

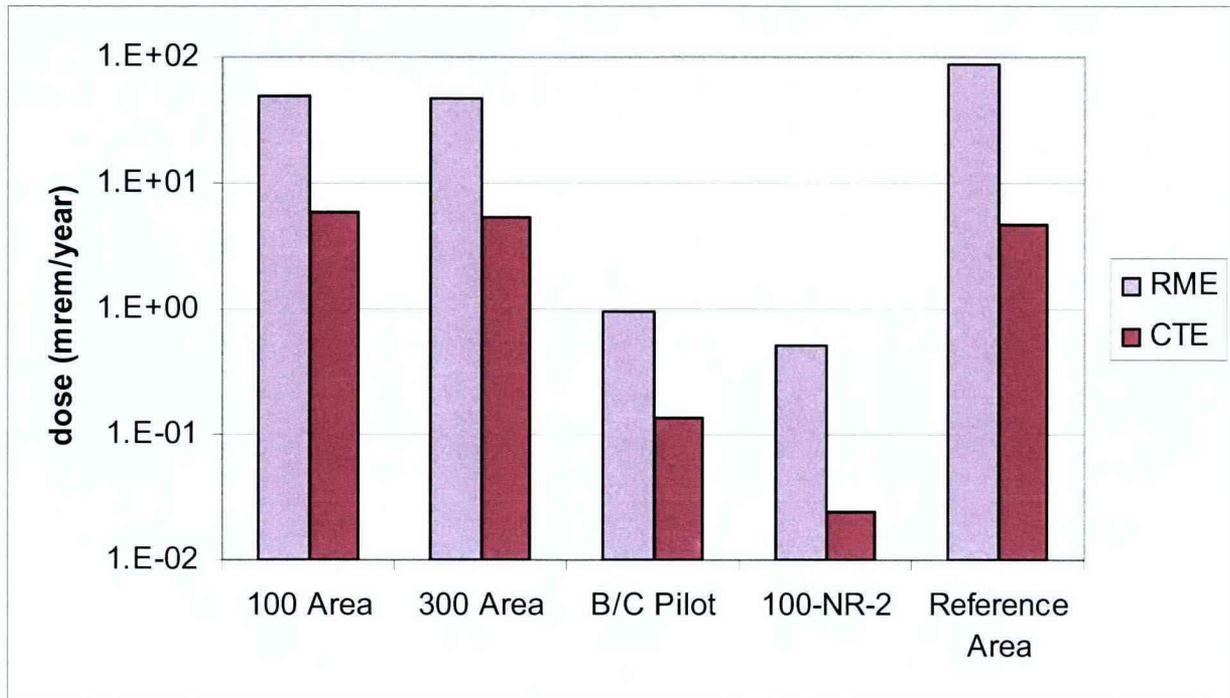


Figure 5-32. Rural Resident Radiation Doses for Fish Ingestion.

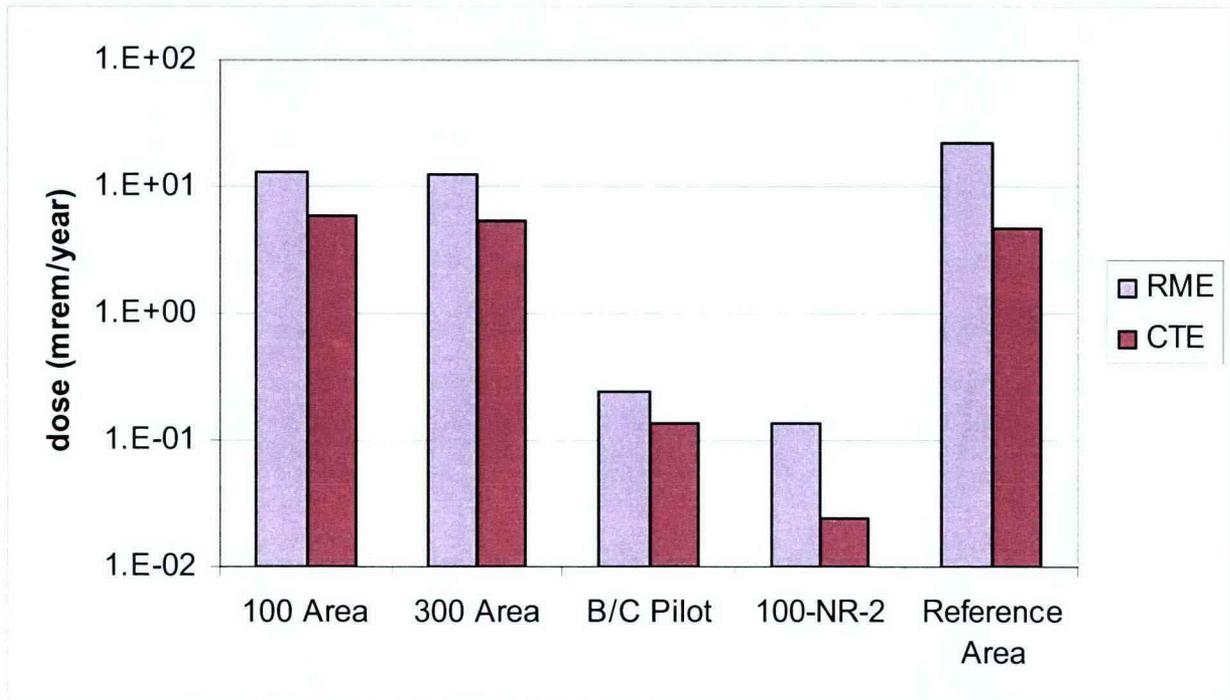


Figure 5-33. CTUIR Radiation Doses for Fish Ingestion.

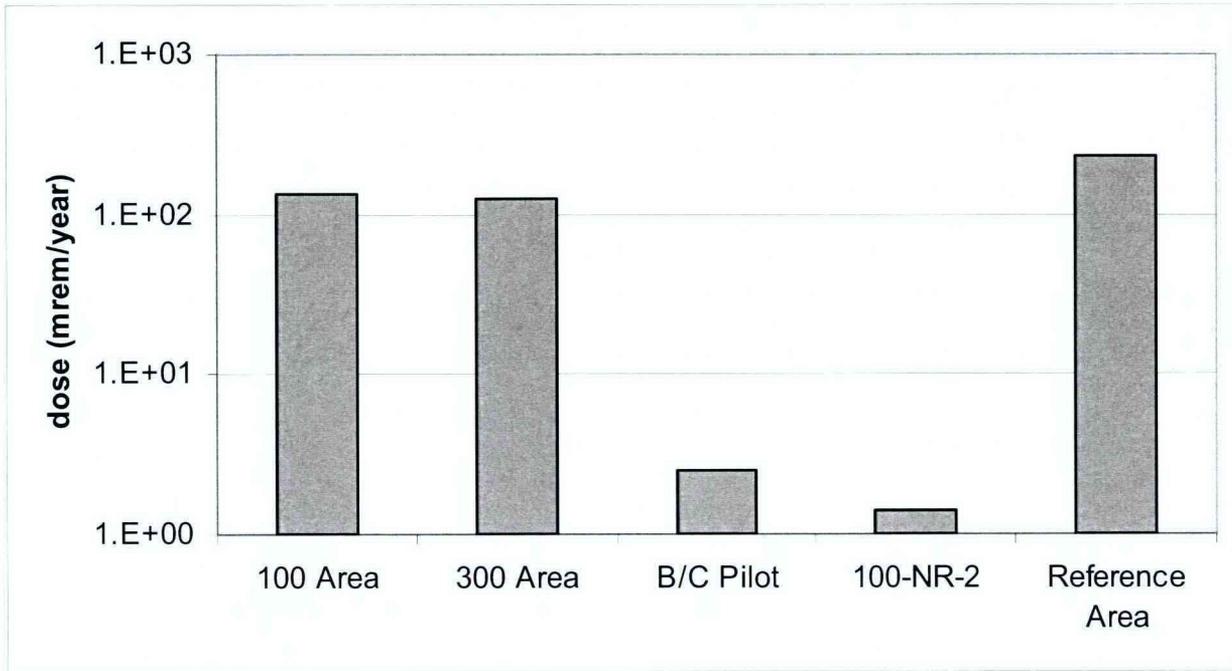


Figure 5-34. Avid Angler Hazard Indices for Fish Ingestion.

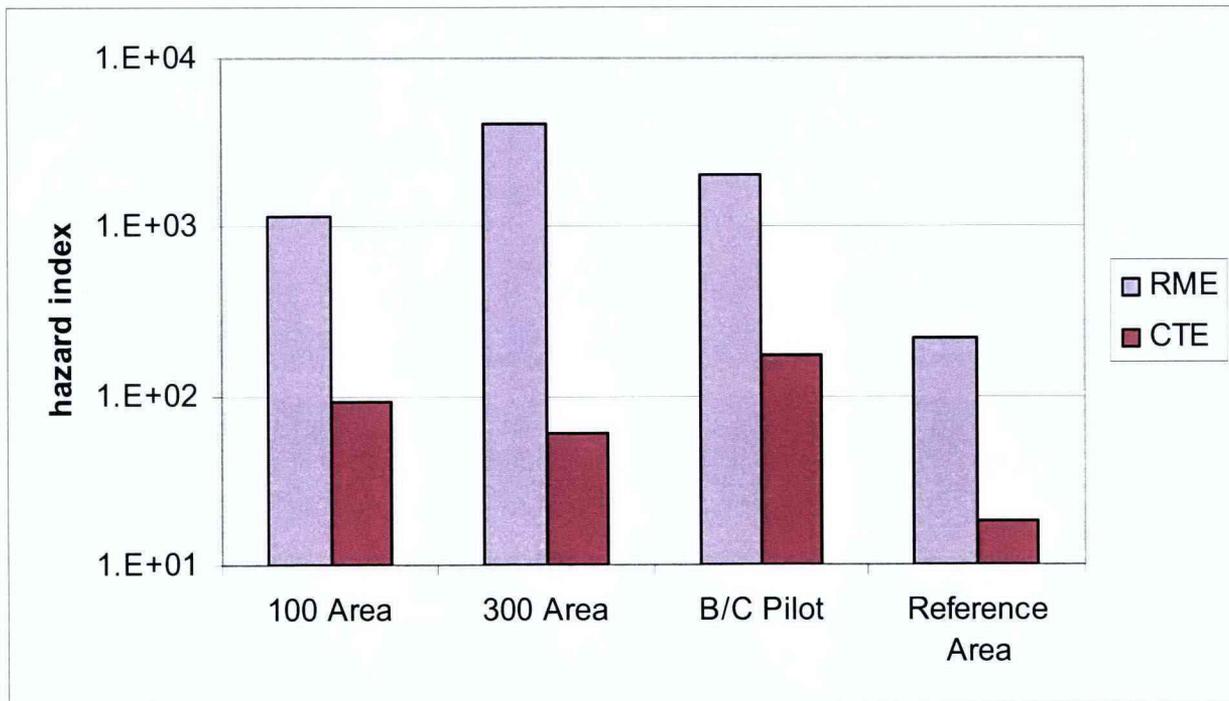


Figure 5-35. Rural Resident Hazard Indices for Fish Ingestion.

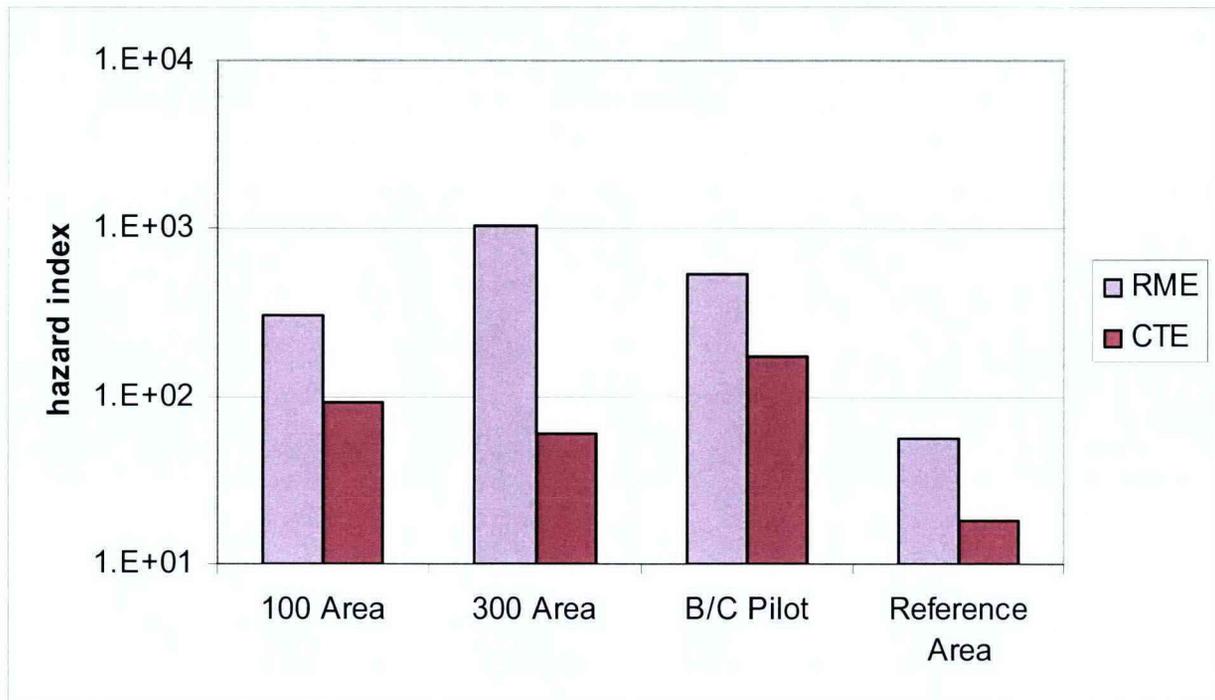


Figure 5-36. CTUIR Hazard Indices for Fish Ingestion.

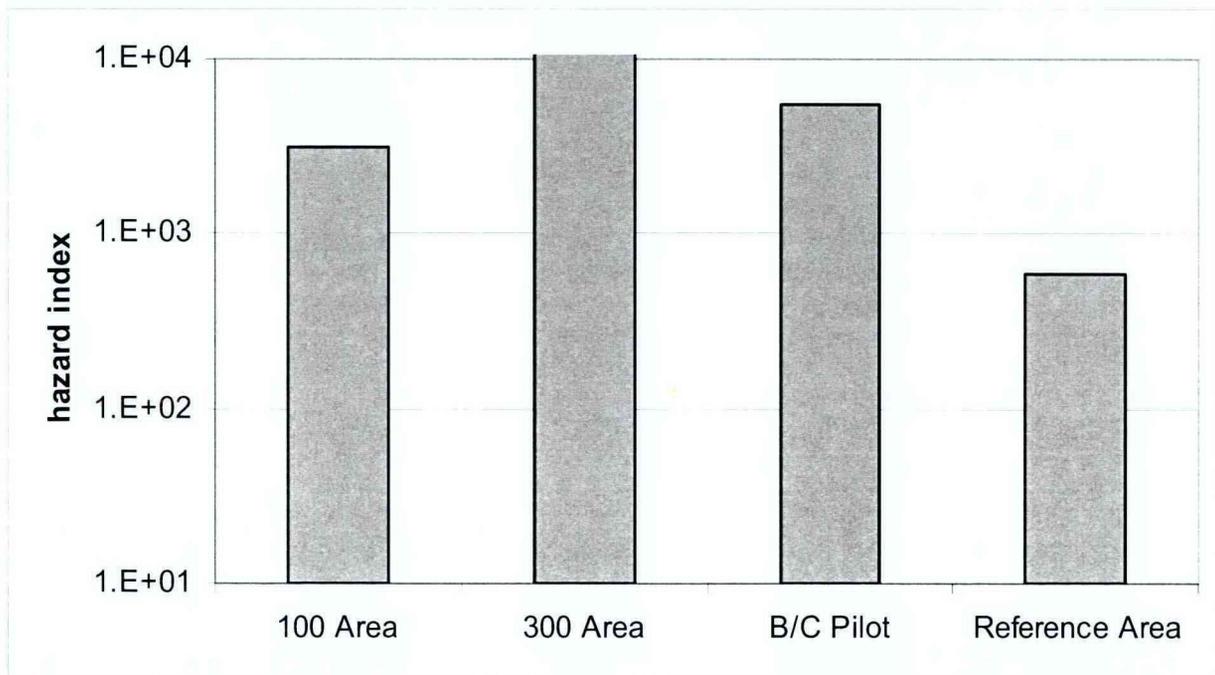


Figure 5-37. Cancer Risks Related to Potassium-40, Isotopic Radium, and Isotopic Thorium – Scenarios Other than Recreational.

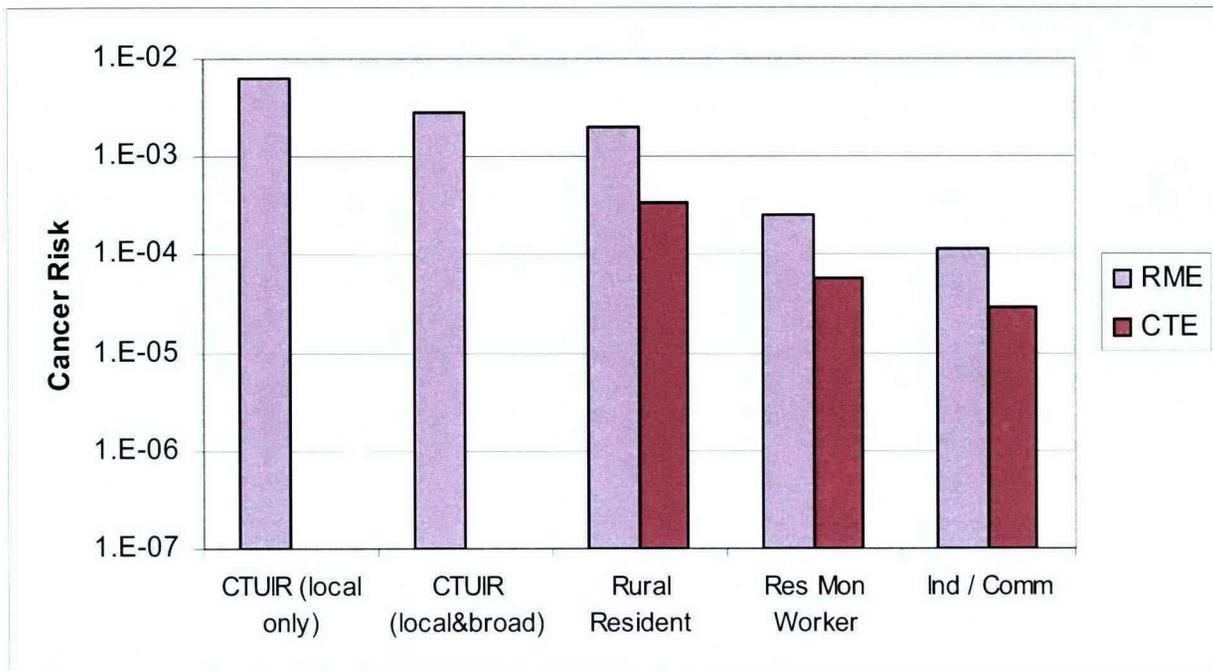


Figure 5-38. Radiation Doses Related to Potassium-40, Isotopic Radium, and Isotopic Thorium – Scenarios Other than Recreational.

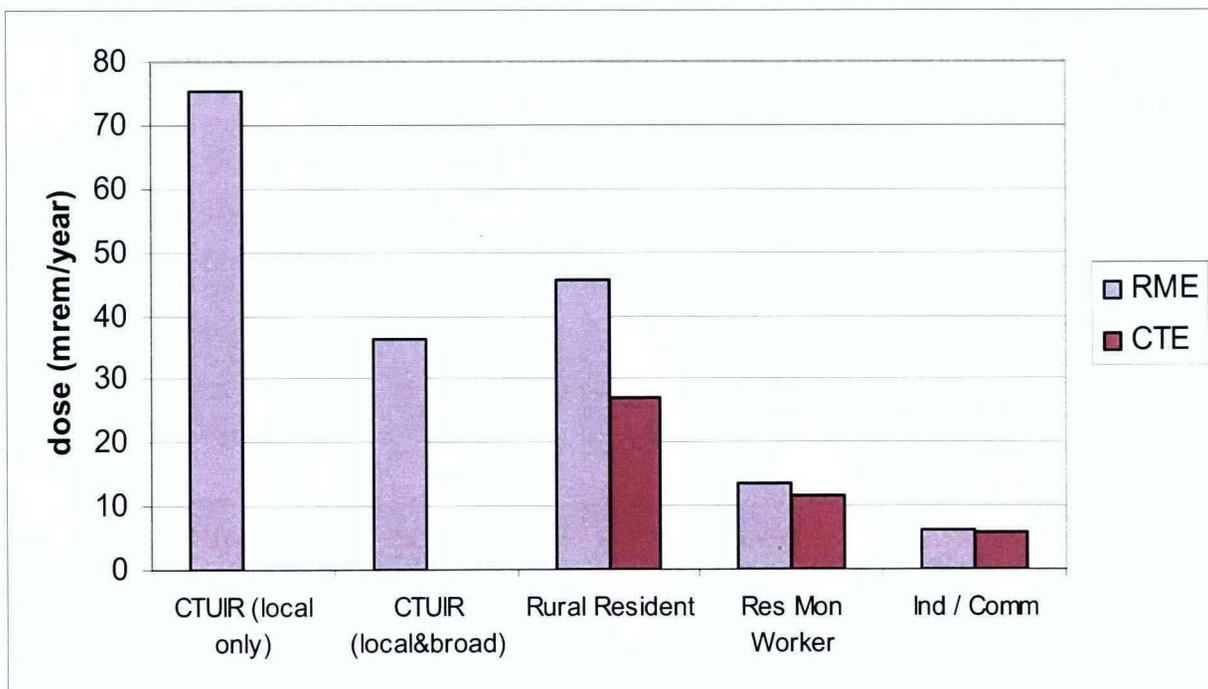


Figure 5-39. Cancer Risks Related to Potassium-40, Isotopic Radium, and Isotopic Thorium – Recreational Scenarios.

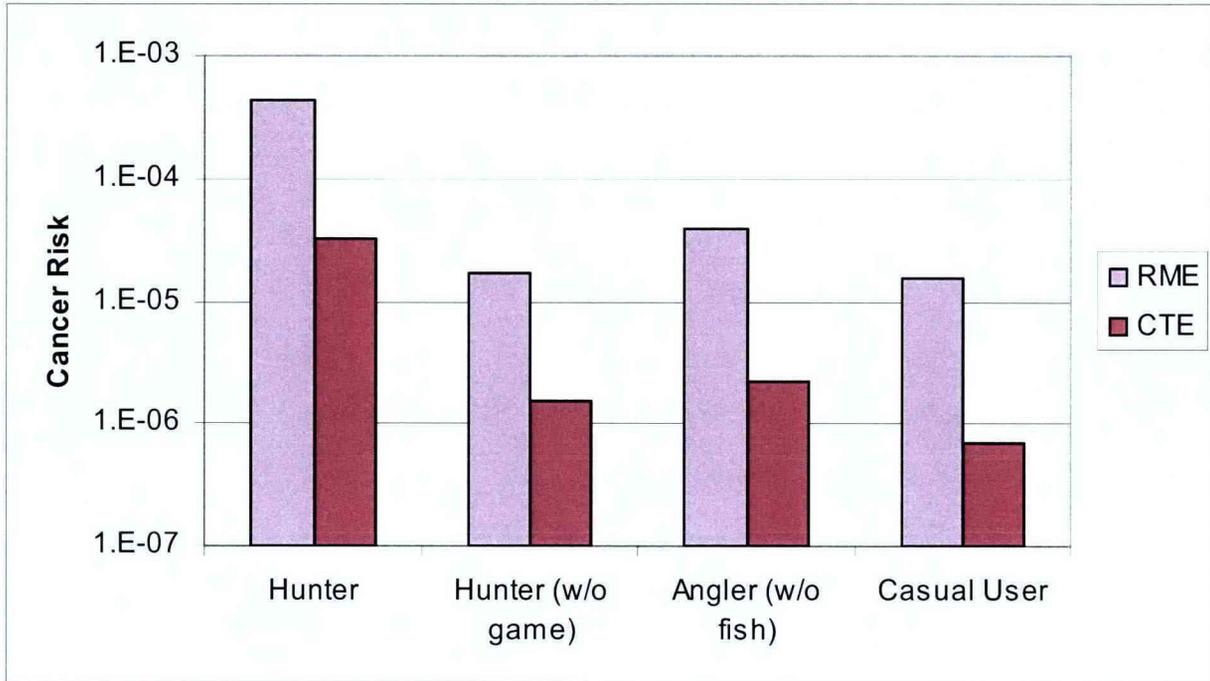


Figure 5-40. Radiation Doses Related to Potassium-40, Isotopic Radium, and Isotopic Thorium – Recreational Scenarios.

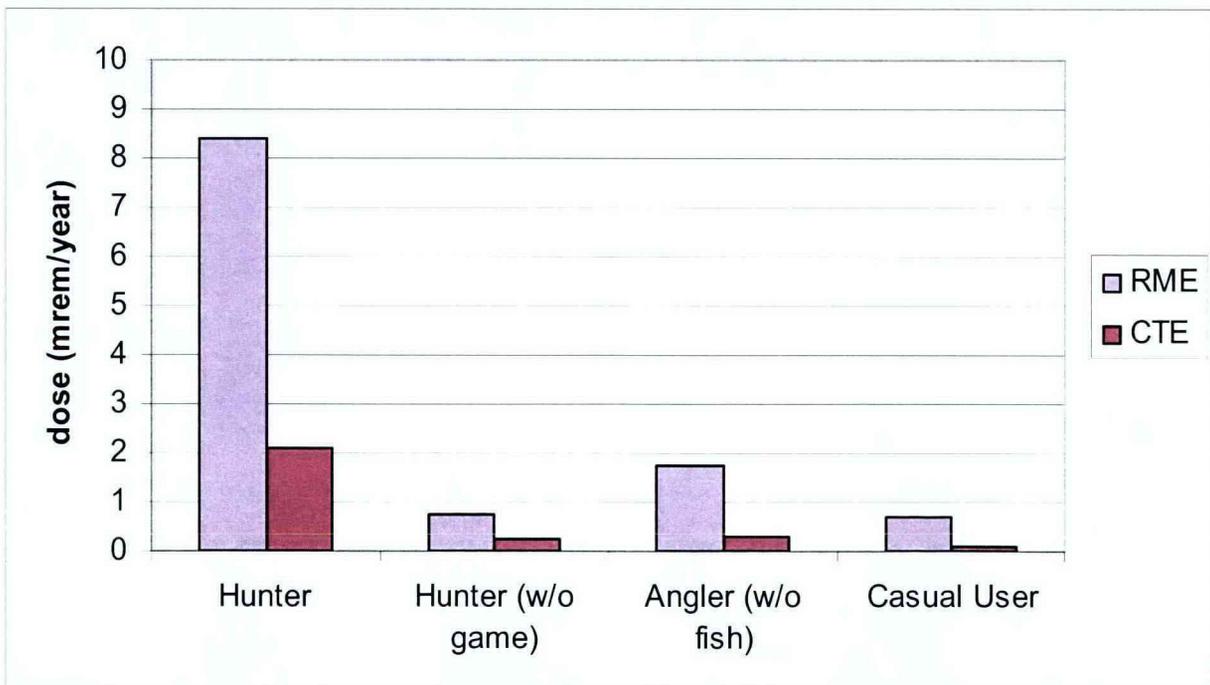


Figure 5-41. Cancer Risks Related to Potassium-40, Isotopic Radium, and Isotopic Thorium – Fish Ingestion Exposure Pathway.

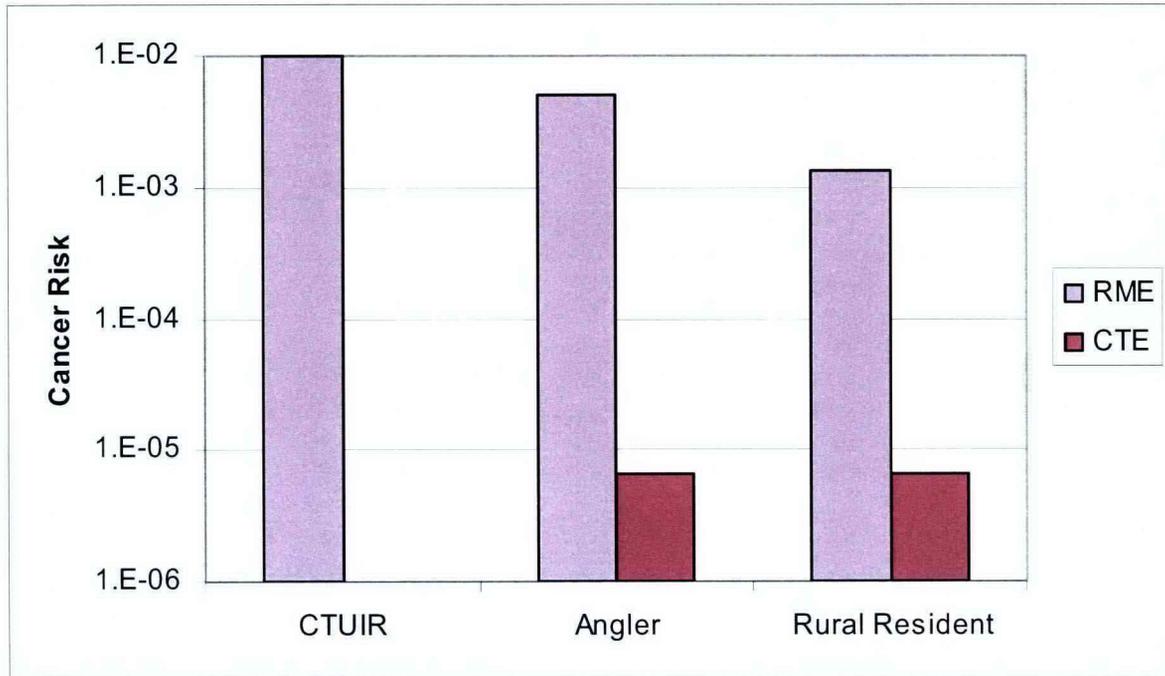


Figure 5-42. Cancer Risks Related to Potassium-40, Isotopic Radium, and Isotopic Thorium – Fish Ingestion Exposure Pathway.

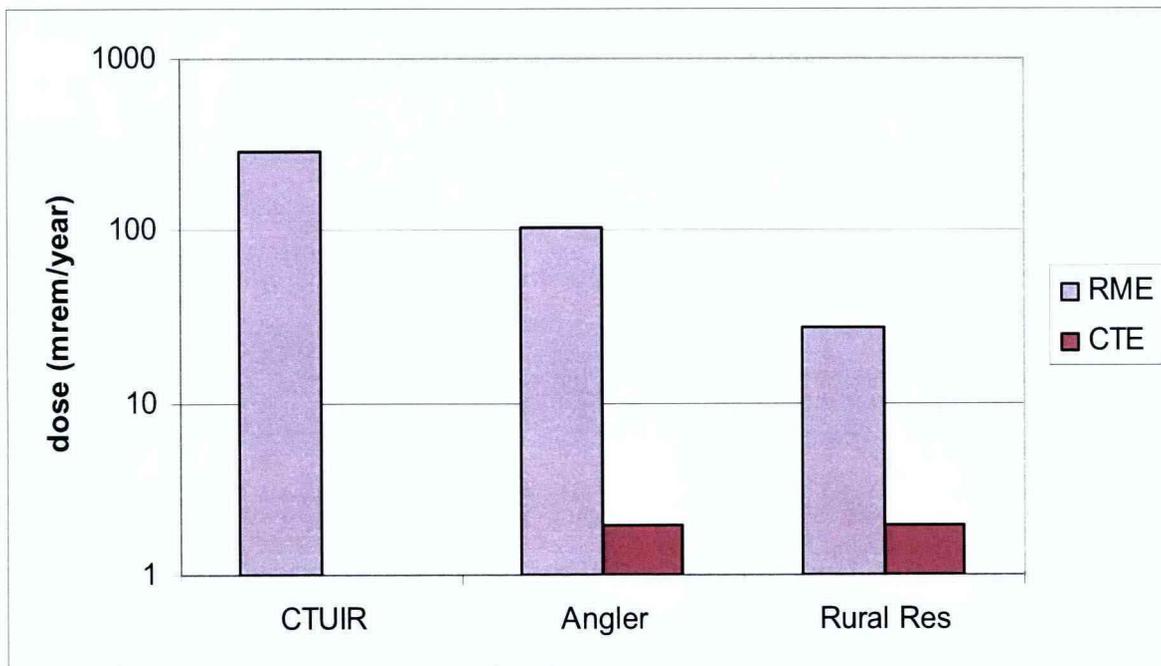


Figure 5-43. CTUIR Scenario Groundwater Cancer Risks.

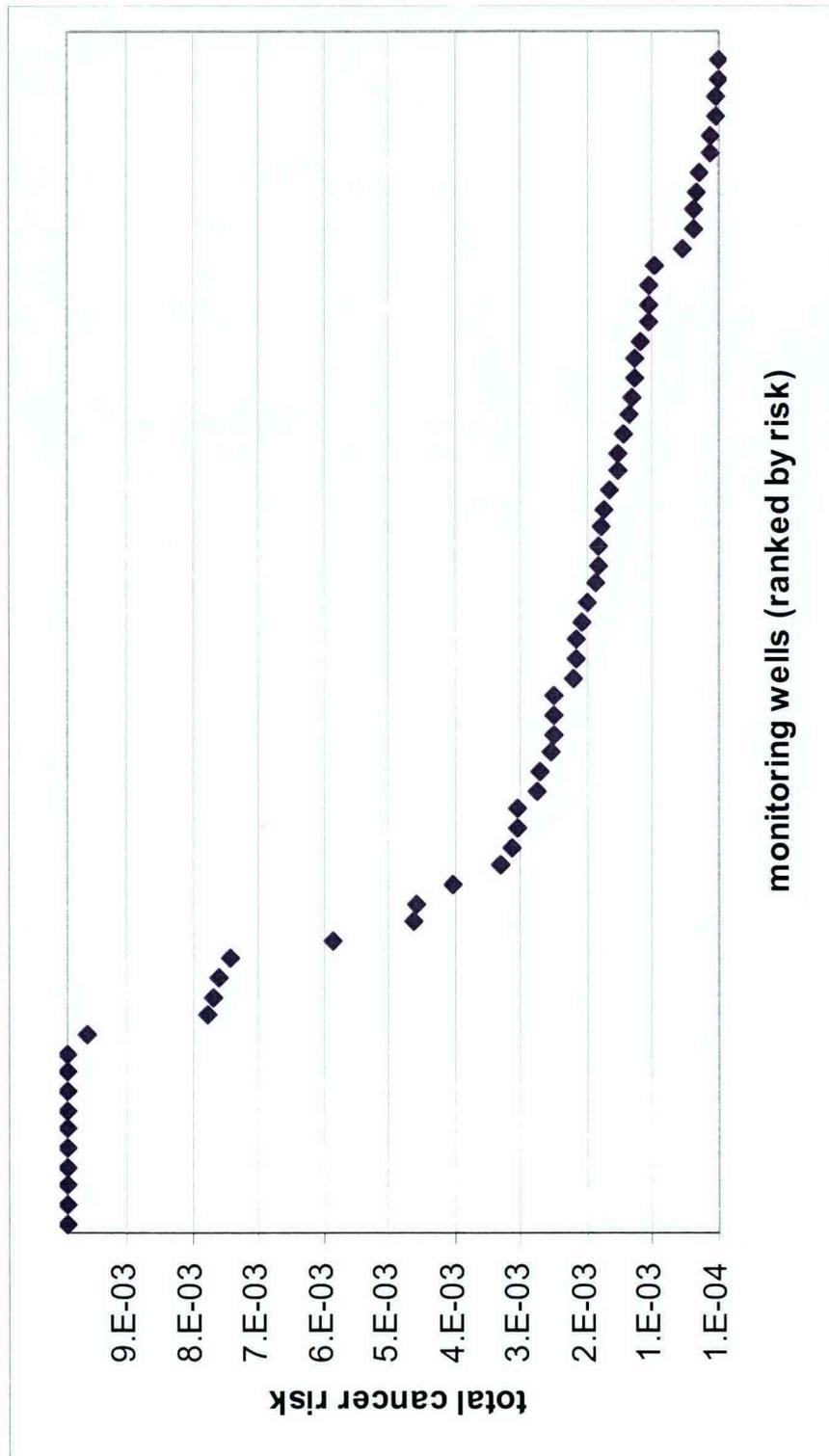


Figure 5-44. CTUIR Scenario Groundwater Radiation Dose.

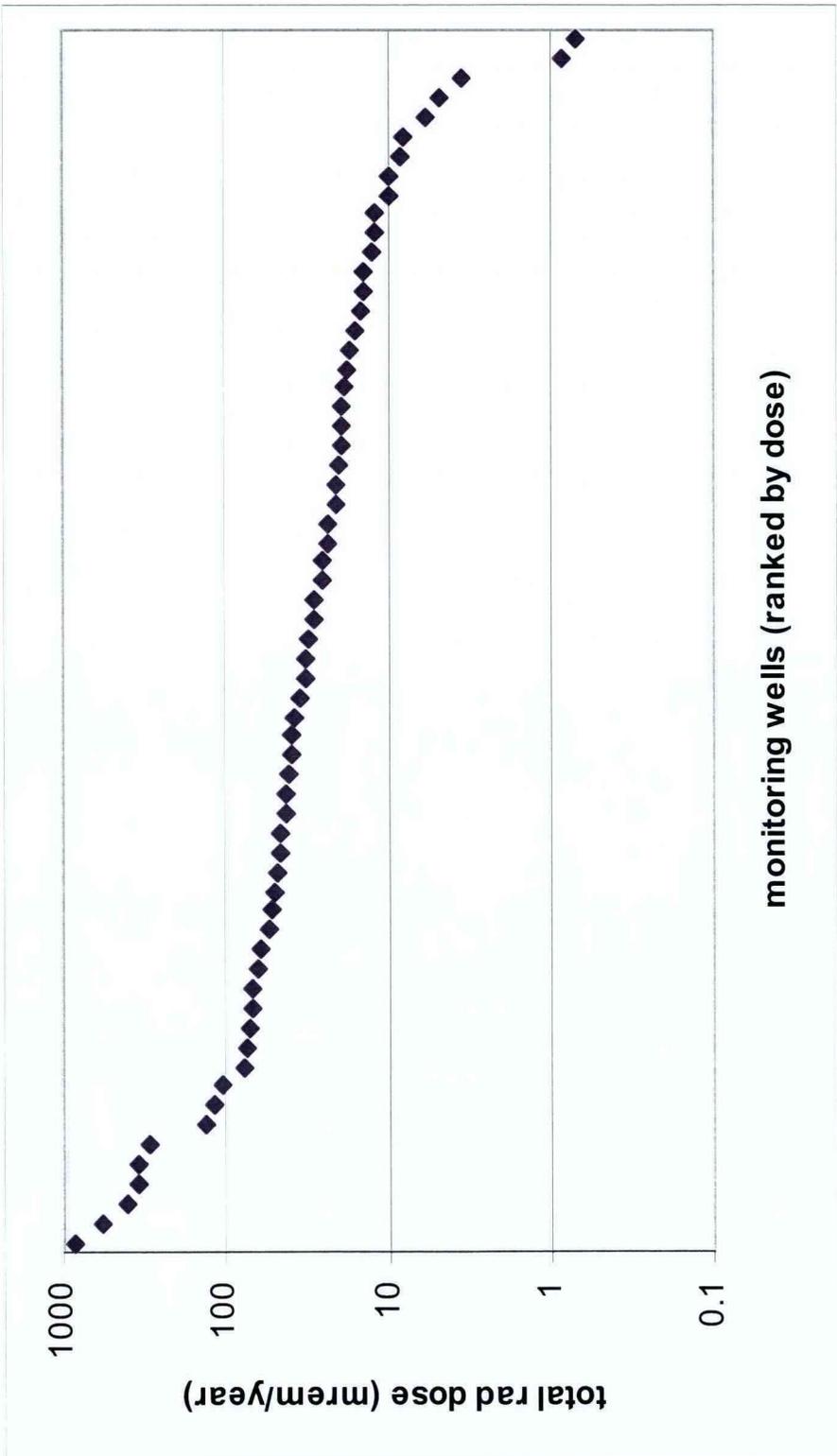


Figure 5-45. CTUIR Scenario Groundwater Adult Chemical Hazard Index.

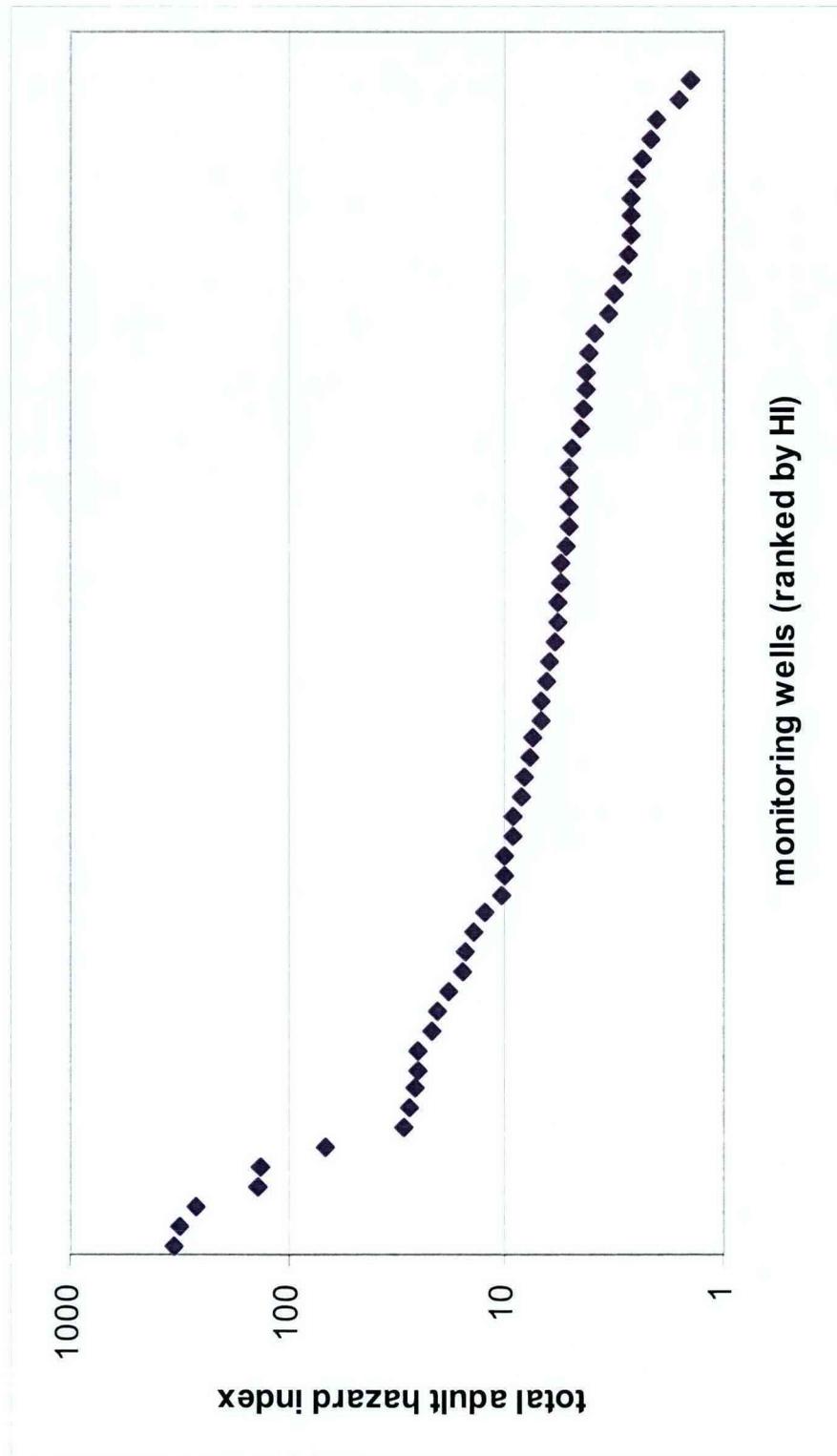


Figure 5-46. Rural Residential Scenario Groundwater Cancer Risks.

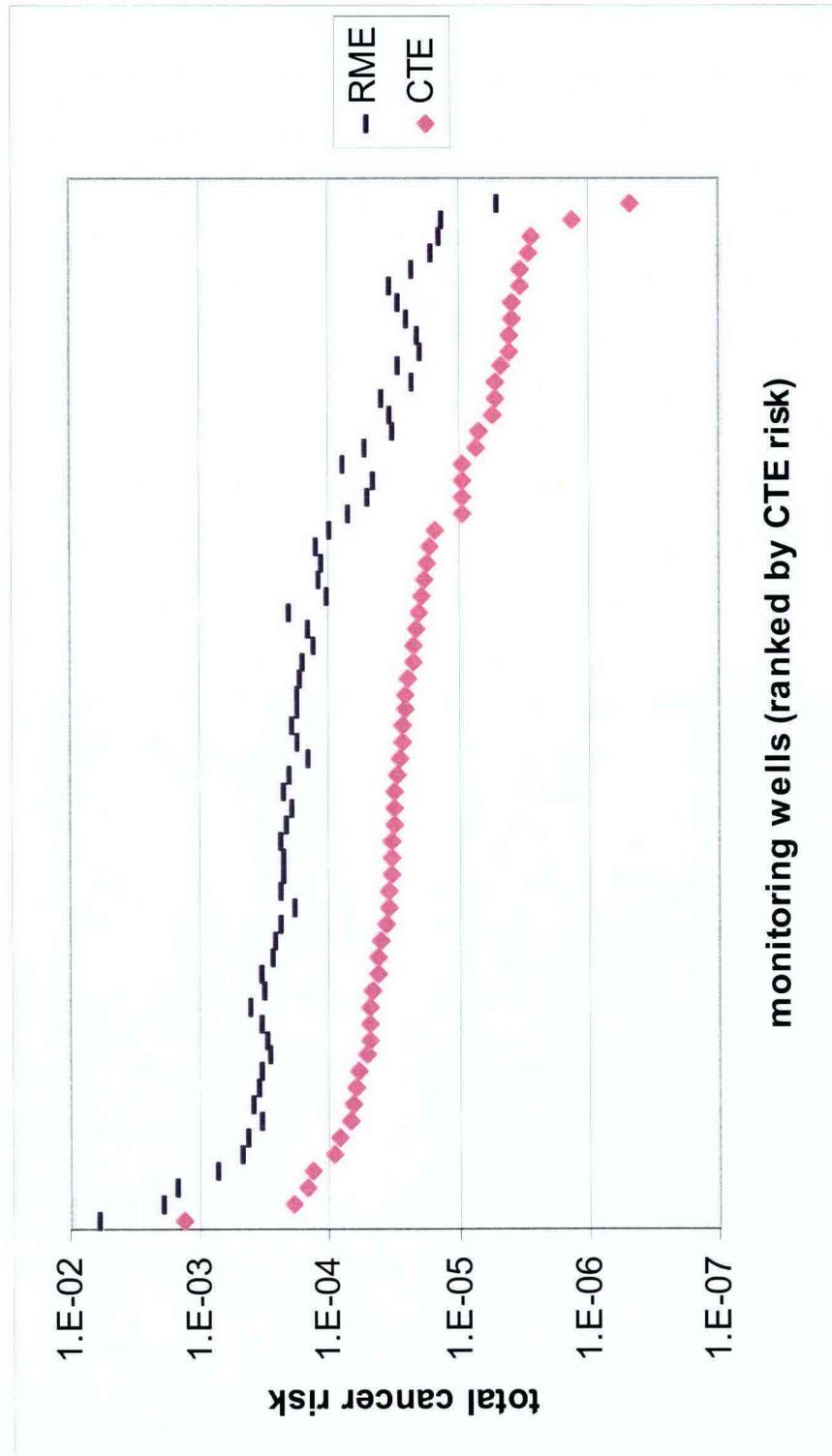


Figure 5-47. Rural Residential Scenario Groundwater Dose.

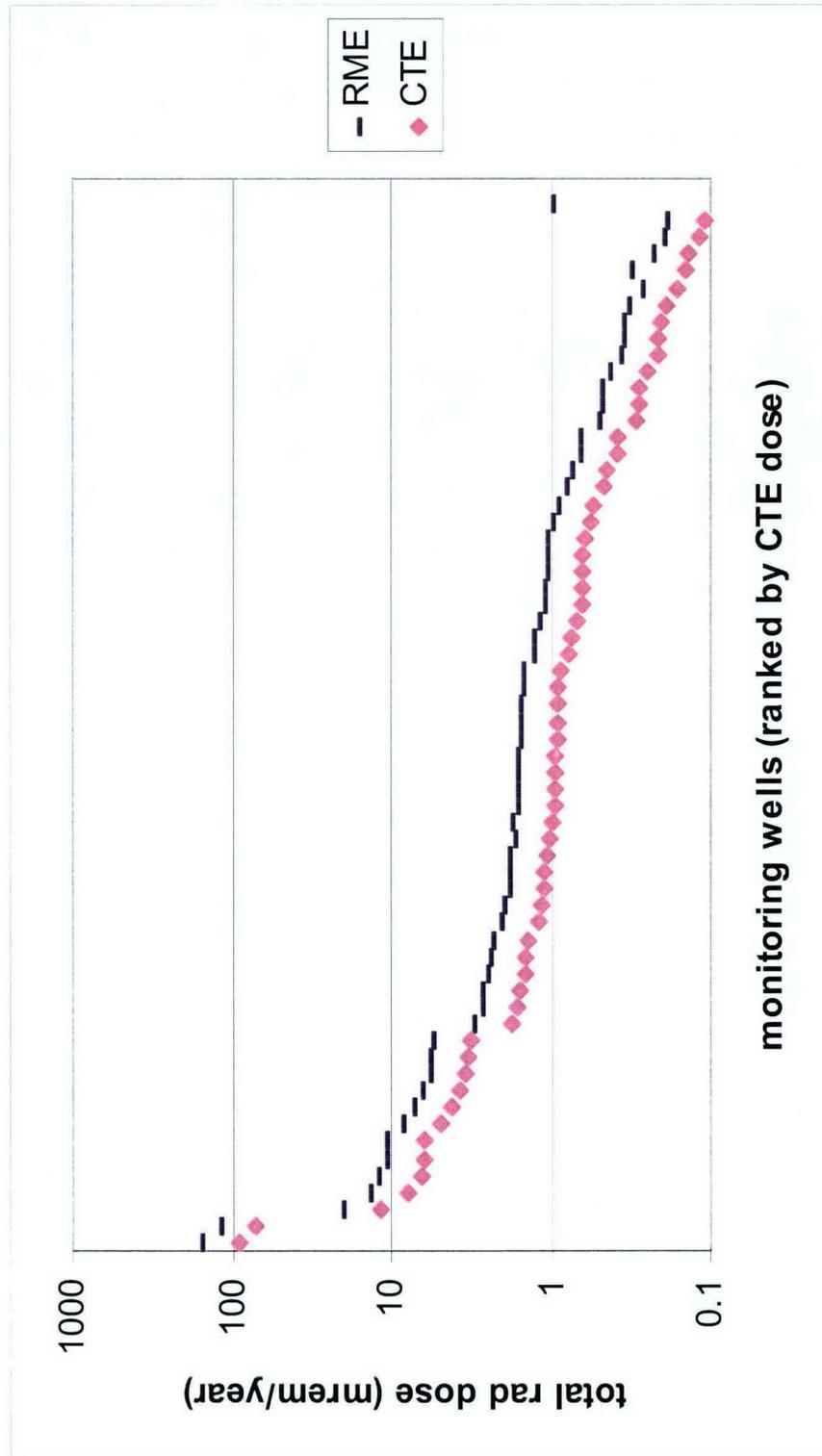


Figure 5-48. Rural Resident Scenario Groundwater Child Chemical Hazard Index.

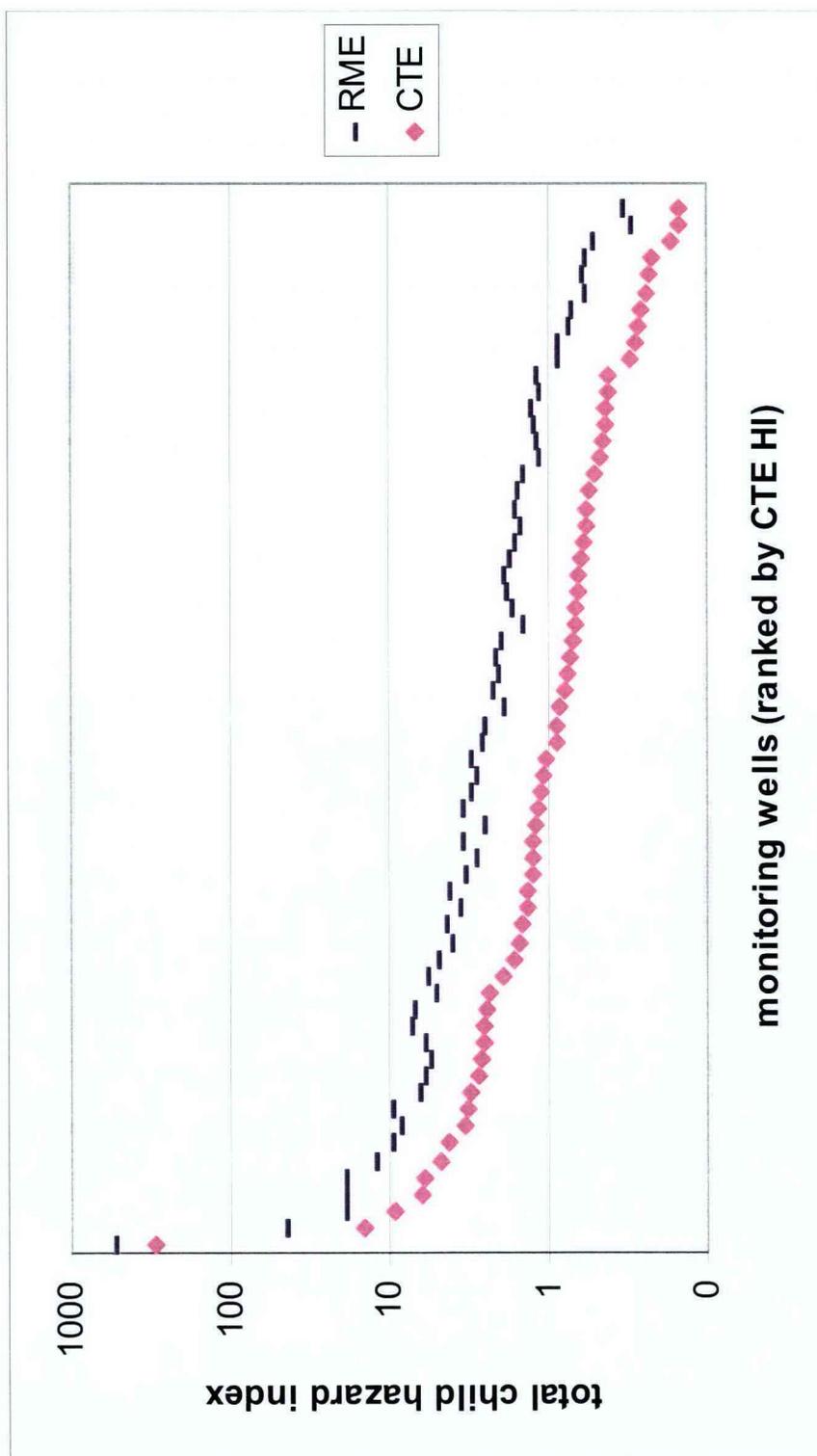


Figure 5-49. Resident Monument Worker Scenario Groundwater Cancer Risks.

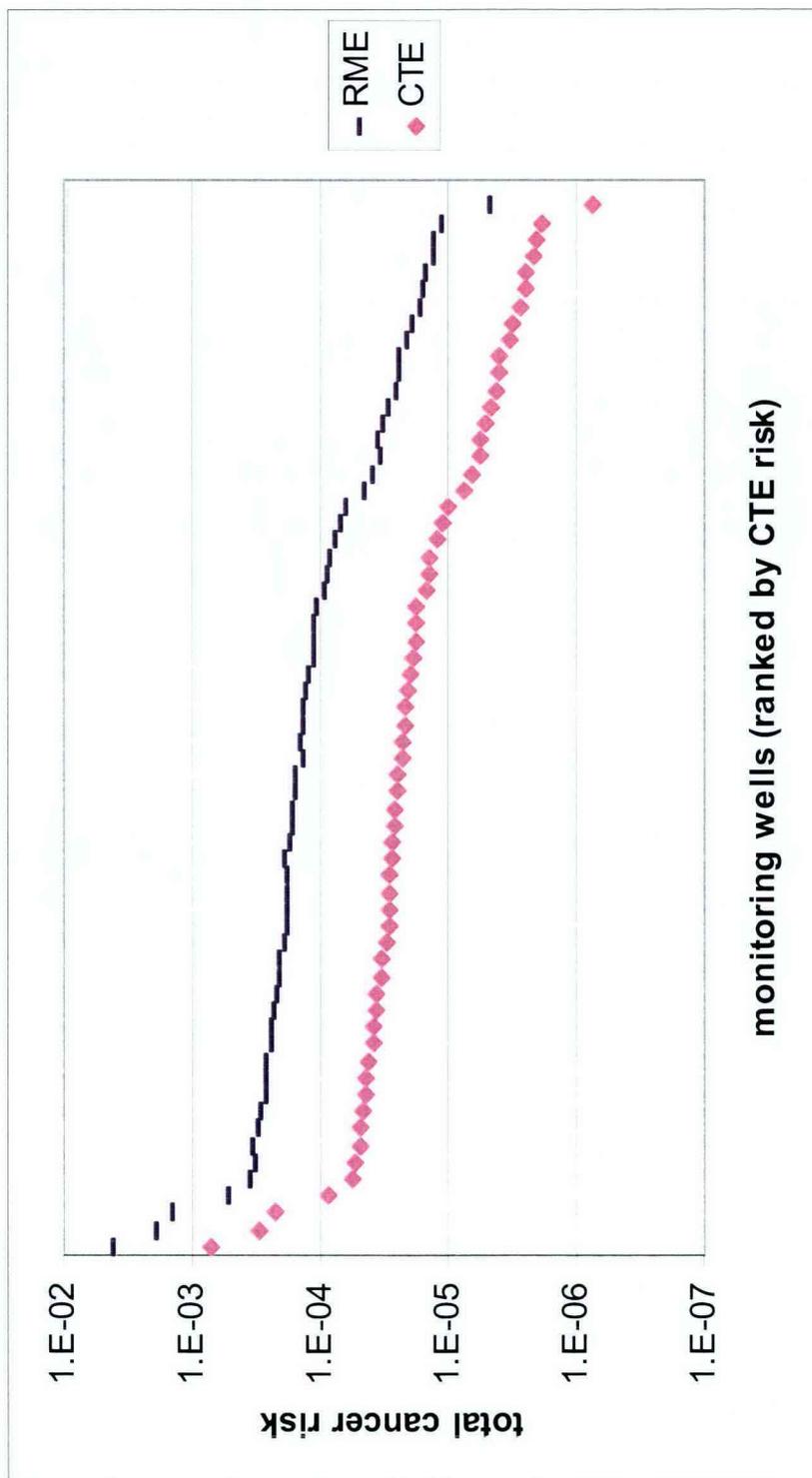


Figure 5-50. Resident Monument Worker Scenario Groundwater Dose.

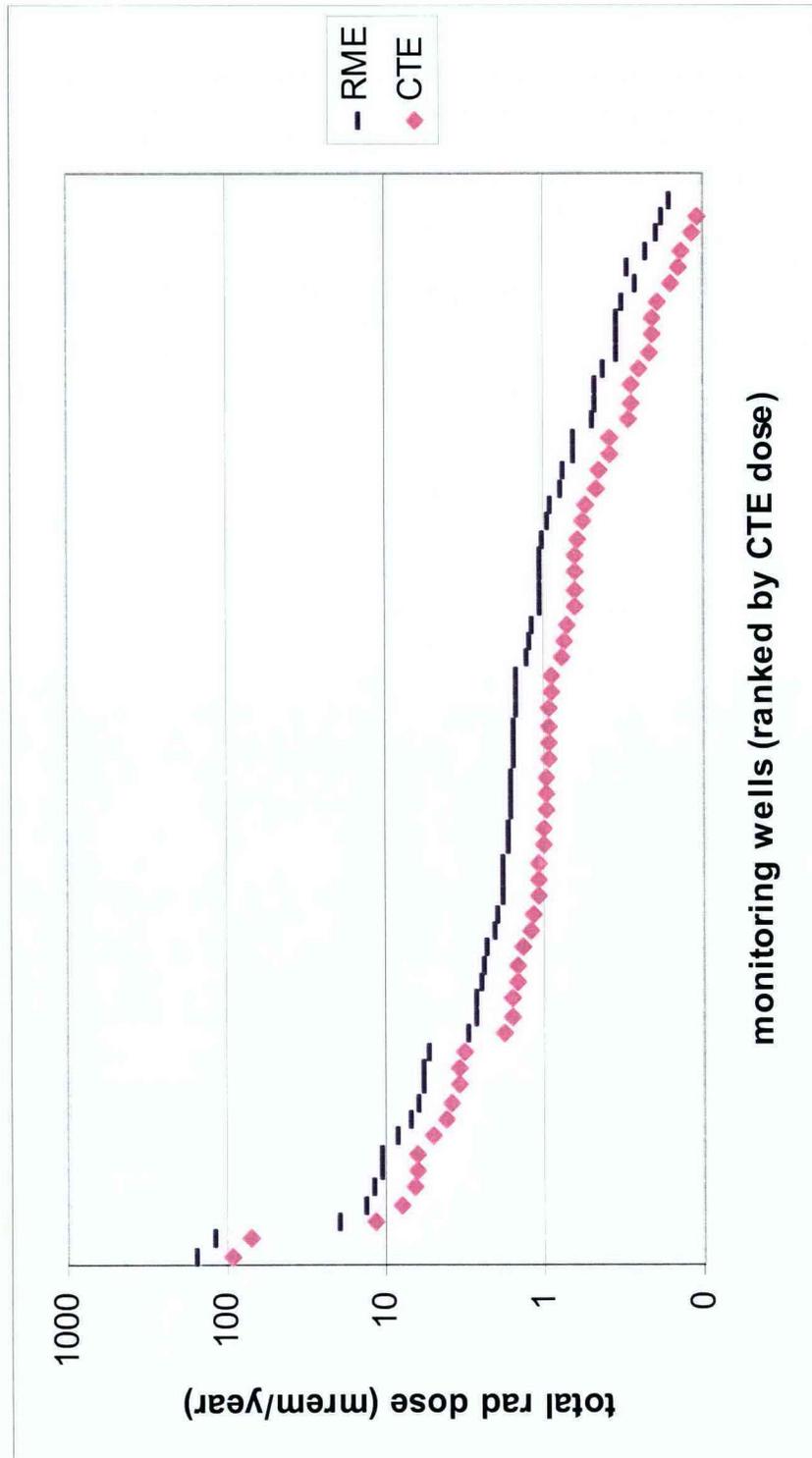
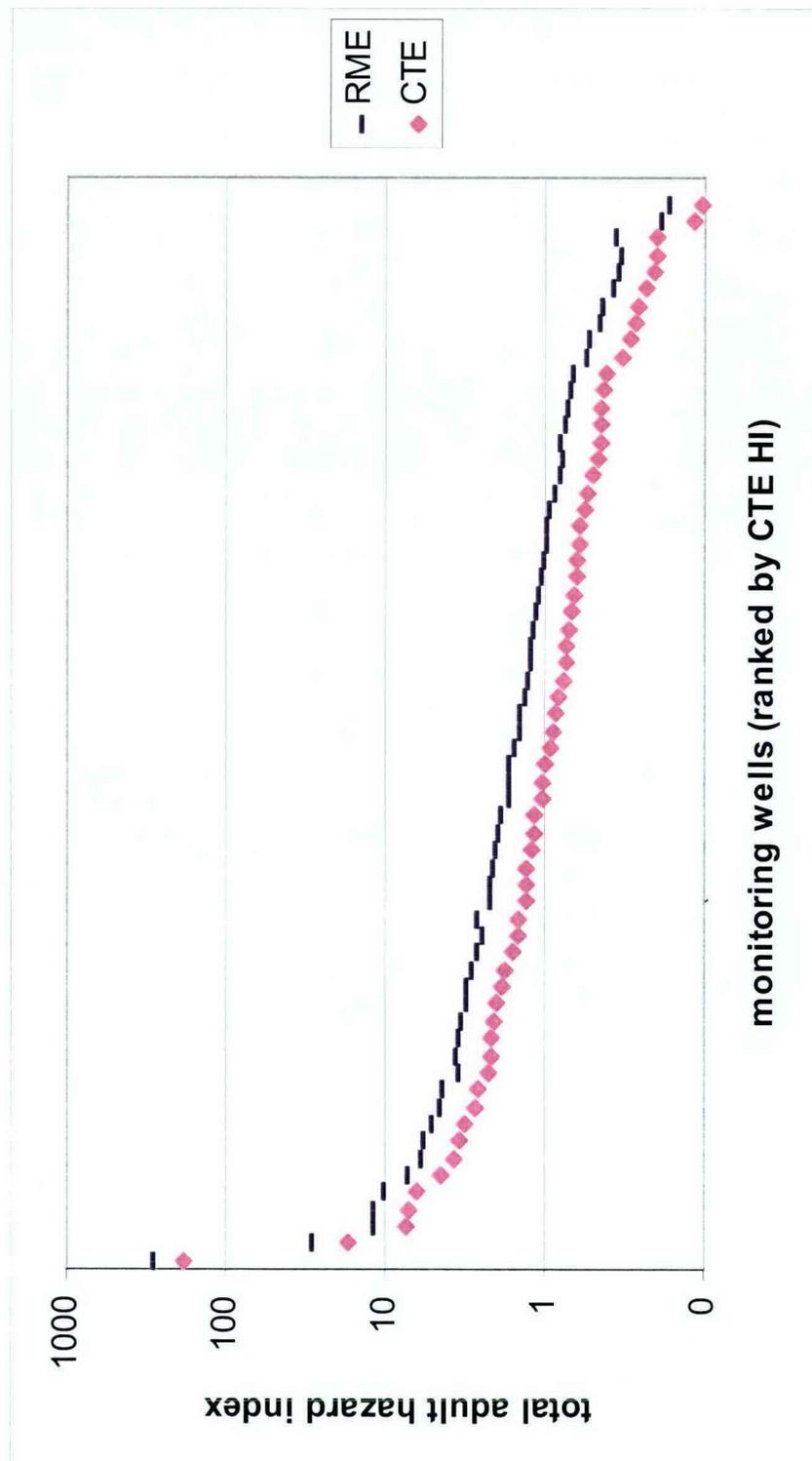


Figure 5-51. Resident Monument Worker Scenario Groundwater Chemical Hazard Index.



Human Health Risk Assessment

Table 5-1. Potentially Complete Exposure Pathways and Associated Media.

Exposure Medium	Exposure Scenario and Receptors: Potentially Complete Pathways				
	Rural Residential (adult; child)	CTUIR (adult; child)	Resident Monument Worker (adult)	Industrial / Commercial (adult)	Recreational (adult; child)
Upland Surface Soil (< 15 ft waste site data; RCBRA data)	Inadvertent Ingestion; Dust Inhalation; Dermal Absorption ¹ ; External Irradiation	Inadvertent Ingestion; Dust Inhalation; Dermal Absorption ¹ ; External Irradiation	Inadvertent Ingestion; Dust Inhalation; Dermal Absorption ¹ ; External Irradiation	Inadvertent Ingestion; Dust Inhalation; Dermal Absorption ¹ ; External Irradiation	Inadvertent Ingestion; Dust Inhalation; Dermal Absorption ¹ ; External Irradiation
Upland Subsurface Soil (> 15 ft waste site data)	Via drill cuttings; mixed with surface soil	Via drill cuttings; mixed with surface soil	Via drill cuttings; mixed with surface soil	<i>incomplete pathway</i>	<i>incomplete pathway</i>
Riparian Soil (RCBRA data, B/C Pilot data, 100-NR-2 data)	<i>incomplete pathway</i>	(Inadvertent Ingestion; Dust Inhalation; Dermal Absorption; External Irradiation) ²	(Inadvertent Ingestion; Dust Inhalation; Dermal Absorption; External Irradiation) ²	<i>incomplete pathway</i>	(Inadvertent Ingestion; Dermal Absorption; External Irradiation) ³ Dust Inhalation
Near-Shore Sediment (RCBRA data, B/C Pilot data, 100-NR-2 data)	<i>incomplete pathway</i>	(Inadvertent Ingestion; Dermal Absorption; External Irradiation) ²	(Inadvertent Ingestion; Dermal Absorption; External Irradiation) ²	<i>incomplete pathway</i>	Inadvertent Ingestion; Dermal Absorption ¹ ; External Irradiation
Garden Produce ⁴	Ingestion	Ingestion	<i>incomplete pathway</i>	<i>incomplete pathway</i>	<i>incomplete pathway</i>
Native Plants	<i>incomplete pathway</i>	Ingestion	<i>incomplete pathway</i>	<i>incomplete pathway</i>	<i>incomplete pathway</i>
Poultry and Eggs ⁴	Ingestion	<i>incomplete pathway</i>	<i>incomplete pathway</i>	<i>incomplete pathway</i>	<i>incomplete pathway</i>
Beef ⁴	Ingestion	Ingestion	<i>incomplete pathway</i>	<i>incomplete pathway</i>	<i>incomplete pathway</i>
Milk ⁴	Ingestion	<i>incomplete pathway</i>	<i>incomplete pathway</i>	<i>incomplete pathway</i>	<i>incomplete pathway</i>
Wild Game ⁴	<i>incomplete pathway</i>	Ingestion	<i>incomplete pathway</i>	<i>incomplete pathway</i>	Ingestion
Food Fish ⁵	Ingestion	Ingestion	<i>incomplete pathway</i>	<i>incomplete pathway</i>	Ingestion

Human Health Risk Assessment

Table 5-1. Potentially Complete Exposure Pathways and Associated Media.

Exposure Medium	Exposure Scenario and Receptors: Potentially Complete Pathways				
	Rural Residential (adult; child)	CTUIR (adult; child)	Resident Monument Worker (adult)	Industrial / Commercial (adult)	Recreational (adult; child)
Groundwater (RCBRA monitoring well data)	Ingestion; Dermal Absorption ¹ ; Indoor Volatiles Inhalation	Ingestion; Dermal Absorption ¹ ; Indoor Volatiles Inhalation; Sweat Lodge Inhalation and Dermal Absorption ¹	Ingestion; Dermal Absorption ¹ ; Indoor Volatiles Inhalation (<i>residential component</i>)	<i>incomplete pathway</i>	<i>incomplete pathway</i>
Seeps and River Water ⁶	<i>incomplete pathway</i>	Sweat Lodge Ingestion; Inhalation and Dermal Absorption ¹	<i>incomplete pathway</i>	<i>incomplete pathway</i>	<i>incomplete pathway</i>

¹ Not evaluated for radionuclides (EPA 1989; Section 10.5.5).

² Because upland RCBRA soil data reflect potential contributions from waste sites, these scenarios are evaluated assuming 100% of exposure time is related to upland soil rather than riparian soil or sediment.

³ Casual User and Avid Hunter recreational scenarios are evaluated assuming 100% of exposure time is related to upland soil rather than riparian soil. Riparian soil is used as the source term for inhalation exposure for the Avid Angler recreational scenario; other Avid Angler exposure pathways pertain to sediments.

⁴ Environmental data for this medium were not collected under the 100 Area and 300 Area Component of the RCBRA Sampling and Analysis Plan (DOE/RL-2005-42), either because this is a hypothetical future exposure medium or because such data were not expected to reflect localized exposures. Contaminant concentrations in these media will be estimated by modeling from concentrations in a sampled medium.

⁵ Environmental data for this medium were not collected under the 100 Area and 300 Area Component of the RCBRA Sampling and Analysis Plan (DOE/RL-2005-42). Data from surrogate media (scuplin) were sampled as part of the 100 Area and 300 Area investigation.

⁶ Risk calculations not performed for these media. Very few surface water analytes with adequate detection limits for conducting background comparisons were elevated relative to Reference Area samples. Adequate quantities of seep water generally unavailable for sweat lodge use. Groundwater is a much protective source term for assessing potential exposures via these pathways than either surface water or seep water.

CTUIR = Confederated Tribes of the Umatilla Indian Reservation

RCBRA = River Corridor Baseline Risk Assessment

Table 5-2. Spatial Scales for Evaluating Soil-Related Exposure Pathways.

Spatial Scale	Exposure Scenario and Receptors: Potentially Complete Pathways				
	Rural Residential (adult; child)	CTUIR (adult; child)	Resident Monument Worker (adult)	Industrial / Commercial (adult)	Recreational (adult; child)
Local Area Related to an individual waste site.	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption ¹ ; External Irradiation; Garden Produce Ingestion; Poultry and Egg Ingestion; Beef and Milk from Penned Cattle	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption ¹ ; External Irradiation; Garden Produce Ingestion ² ; Beef Ingestion from Penned Cattle ³	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption ¹ ; External Irradiation <i>(for the residential component of this scenario)</i>	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption ¹ ; External Irradiation	<i>No local-area exposure pathways.</i>
Broad Area Related to an entire operational area.	Beef and Milk from Free-Range Cattle; Dust Inhalation ⁴	Use of Native Plants; Ingestion of Meat from Wild Game; Dust Inhalation ⁴	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption ¹ ; External Irradiation <i>(for the occupational component of this scenario)</i>	<i>No broad-area exposure pathways.</i>	Inadvertent Soil Ingestion; Dust Inhalation; Dermal Absorption ¹ ; External Irradiation

¹ Not evaluated for radionuclides (EPA 1989; Section 10.5.5).

² Evaluated as a localized surrogate for use of gathered native plants.

³ Evaluated as a localized surrogate for ingestion of meat from wild game.

⁴ Inhalation exposure from the broad area source term is added to "local area" exposure for the combined Local and Broad Areas calculations.

CTUIR = Confederated Tribes of the Umatilla Indian Reservation

Table 5-3. Henry's Constants (unitless). (2 Pages)

Analyte	Value	Analyte	Value
Acenaphthene	6.7E-03	Dibenz[a,h]anthracene	6.3E-07
Acenaphthylene	4.7E-04	Dibenzofuran	8.8E-03
Acetone	1.6E-03	Di-n-butylphthalate	1.9E-04
Aldrin	7.1E-03	1,4-Dichlorobenzene	1.0E-01
Anthracene	2.7E-03	1,2-Dichloroethane	4.1E-02
Aroclor 1016	1.1E-02	cis-1,2-Dichloroethene	1.7E-01
Aroclor 1221 ¹	1.3E-02	Dichloroprop	5.1E-07
Aroclor 1232 ¹	8.4E-03	Dieldrin	6.3E-04
Aroclor 1242 ¹	8.0E-03	Diethyl ether	5.2E-02
Aroclor 1248 ¹	1.2E-02	Diethylphthalate	1.9E-05
Aroclor 1254	1.2E-02	Di-n-octylphthalate	1.9E-05
Aroclor 1260 ¹	1.2E-02	Endosulfan I ³	2.8E-03
Aroclor 1262 ^{1,2}	1.2E-02	Endosulfan II ³	2.8E-03
Aroclor 1268 ^{1,2}	1.2E-02	Endosulfan sulfate	5.0E-10
Benzene	2.4E-01	Endrin	3.2E-04
Benzo(a)anthracene	1.4E-04	Endrin aldehyde	1.8E-04
Benzo(a)pyrene	4.6E-05	Endrin ketone ⁴	3.2E-04
Benzo(b)fluoranthene	4.7E-03	Ethylene glycol	2.5E-06
Benzo(ghi)perylene	1.1E-05	Fluoranthene	6.7E-04
Benzo(k)fluoranthene	3.5E-05	Fluorene	2.7E-03
Alpha-BHC	2.8E-04	Heptachlor	6.3E+01
beta-1,2,3,4,5,6-Hexachlorocyclohexane	1.8E-05	Heptachlor epoxide	4.0E-04
Delta-BHC	1.8E-05	Indeno(1,2,3-cd)pyrene	6.7E-05
Bis(2-ethylhexyl) phthalate	5.5E-06	Isophorone	2.8E-04
1-Butanol	3.7E-04	Methoxychlor	6.7E-04
2-Butanone	2.0E-03	Methyl isobutyl ketone	5.9E-03
2-Butoxyethanol	6.7E-05	Methylenechloride	9.2E-02
Butylbenzylphthalate	2.0E-04	2-Methylnaphthalene	2.2E-02
2-secButyl-4,6-dinitrophenol(DNBP)	2.1E-02	3+4 Methylphenol (cresol, m+p) ¹	3.8E-04
Carbazole	3.6E-06	Naphthalene	2.0E-02
Carbon disulfide	1.3E+00	Pentachlorophenol	1.0E-06
Carbon tetrachloride	1.3E+00	Phenanthrene	9.7E-04
Chlordane	2.1E-03	Phenol	1.7E-05
Chloroform	1.6E-01	Picloram	2.2E-12
Chrysene	4.0E-03	Pyrene	4.6E-04
Dalapon ⁵	1.0E-10	1,1,2,2-Tetrachloroethane	1.0E-01

Table 5-3. Henry's Constants (unitless). (2 Pages)

Analyte	Value	Analyte	Value
Dicamba	8.4E-08	Tetrachloroethene	7.4E-01
Dichlorodiphenyldichloroethane	2.8E-04	Toluene	2.8E-01
Dichlorodiphenyldichloroethylene	1.7E-03	Trichloroethene	4.1E-01
Dichlorodiphenyltrichloroethane	3.5E-04	Trichloromonofluoromethane	4.1E+00
2,4-Dichlorophenoxyacetic acid	3.6E-04	2,4,5-Trichlorophenoxyacetic acid	4.0E-09
4-(2,4-Dichlorophenoxy)butanoic acid	9.6E-08	2-(2,4,5-Trichlorophenoxy)propionic acid ⁵	1.0E-10

¹ Value is the midpoint of the range cited in HSDB (NLM 2007).

² Aroclor 1260 used as a surrogate.

³ Endosulfan used as a surrogate.

⁴ Endrin used as a surrogate.

⁵ This compound is effectively nonvolatile (NLM 2007). A value of 1E-10 is used to represent zero.

Table 5-4. Plant-Soil Concentration Ratios (Kp-s) for Garden Produce. (3 Pages)

Analyte	mg/kg dry plant per mg/kg soil	Analyte	mg/kg dry plant per mg/kg soil
INORGANIC CHEMICALS¹			
Aluminum	2.7E-02	Mercury	2.5E+00
Antimony	6.7E-02	Molybdenum	8.7E-01
Arsenic	5.3E-01	Nickel	3.3E-01
Barium	3.3E-02	Nitrogen in Nitrate ³	0.0E+00
Beryllium	2.7E-02	Nitrogen in Nitrite ³	0.0E+00
Boron ⁴	2.0E+00	Selenium	6.7E-01
Cadmium	2.0E+00	Silver	1.0E+00
Chromium	1.7E-03	Strontium (elemental)	2.0E+00
Hexavalent Chromium	1.7E-03	Thallium	1.3E+00
Cobalt	5.3E-01	Tin	1.7E-02
Copper	8.7E-01	Uranium	1.7E-02
Fluoride	1.3E-01	Vanadium	1.3E-02
Manganese	2.0E+00	Zinc	2.7E+00
ORGANIC CHEMICALS²			
Acenaphthylene	1.7E-01	Dibenzofuran	1.6E-01
Acetaldehyde	6.1E+01	Di-n-butylphthalate	5.7E-02
Acetone	5.3E+01	Dicamba	2.0E+00
Aldrin	2.6E-02	1,4-Dichlorobenzene	3.7E-01
Anthracene	9.7E-02	1,2-Dichloroethane	5.3E+00
Aroclor 1016	2.0E-02	cis-1,2-Dichloroethene	3.3E+00
Aroclor 1221	1.7E-01	Dichloroprop	4.0E-01
Aroclor 1232	9.2E-02	Dieldrin	9.7E-02
Aroclor 1242	1.6E-01	Diethyl ether	1.2E+01
Aroclor 1248	1.0E-02	Diethylphthalate	1.4E+00
Aroclor 1254	6.8E-03	Di-n-octylphthalate	8.1E-04
Aroclor 1260	4.5E-03	Endosulfan I	2.4E-01
Aroclor-1262	4.5E-03	Endosulfan II	2.4E-01
Aroclor-1268	4.5E-03	Endosulfan sulfate	3.0E-01
Benzene	2.4E+00	Endrin	8.5E-02
Benzo(a)anthracene	2.0E-02	Endrin aldehyde	6.5E-02
Benzo(a)pyrene	1.3E-02	Endrin ketone	8.5E-02
Benzo(b)fluoranthene	1.1E-02	Ethylene glycol	2.4E+02
Benzo(ghi)perylene	5.7E-03	Fluoranthene	5.0E-02

Table 5-4. Plant-Soil Concentration Ratios (Kp-s) for Garden Produce. (3 Pages)

Analyte	mg/kg dry plant per mg/kg soil	Analyte	mg/kg dry plant per mg/kg soil
Benzo(k)fluoranthene	1.2E-02	Fluorene	1.4E-01
Alpha-BHC	2.5E-01	Heptachlor	1.3E-01
beta-1,2,3,4,5,6-Hexachlorocyclohexane	2.5E-01	Heptachlor epoxide	2.9E-02
Delta-BHC	1.6E-01	Indeno(1,2,3-cd)pyrene	5.9E-03
Bis(2-ethylhexyl) phthalate	1.6E-03	Isophorone	4.2E+00
1-Butanol	1.2E+01	Methoxychlor	6.5E-02
2-Butanone	2.6E+01	Methyl isobutyl ketone	7.8E+00
2-Butoxyethanol	1.3E+01	Methylenechloride	6.9E+00
Butylbenzylphthalate	5.6E-02	2-Methylnaphthalene	2.2E-01
2-secButyl-4,6-dinitrophenol(DNBP)	3.4E-01	3+4 Methylphenol (cresol, m+p)	1.5E+00
Carbazole	2.7E-01	Naphthalene	4.8E-01
Carbon disulfide	2.1E+00	Pentachlorophenol	4.4E-02
Carbon tetrachloride	9.3E-01	Phenanthrene	9.7E-02
Chlordane	2.6E-02	Phenol	5.3E+00
Chloroform	2.7E+00	Picloram	2.6E+01
Chrysene	2.0E-02	Pyrene	5.7E-02
Dalapon	1.4E+01	1,1,2,2-Tetrachloroethane	6.9E-01
Dichlorodiphenyldichloroethane	1.3E-02	Tetrachloroethene	4.2E-01
Dichlorodiphenyldichloroethylene	6.7E-03	Toluene	1.1E+00
Dichlorodiphenyltrichloroethane	3.9E-03	Trichloroethene	1.2E+00
2,4-Dichlorophenoxyacetic acid	9.2E-01	Trichloromonofluoromethane	1.4E+00
4-(2,4-Dichlorophenoxy)butanoic acid	3.5E-01	2,4,5-Trichlorophenoxyacetic acid	1.9E-01
Dibenz[a,h]anthracene	6.8E-03	2-(2,4,5-Trichlorophenoxy)propionic acid	2.5E-01
RADIONUCLIDES¹			
Americium-241	6.7E-03	Potassium-40	2.0E+00
Barium-133	3.3E-02	Radium-226	2.7E-01
Carbon-14	3.7E+01	Radium-228	2.7E-01
Cesium-137	2.7E-01	Strontium-90	2.0E+00
Cobalt-60	5.3E-01	Technetium-99	3.3E+01
Europium-152	1.7E-02	Thorium-228	6.7E-03
Europium-154	1.7E-02	Thorium-230	6.7E-03
Europium-155	1.7E-02	Thorium-232	6.7E-03
Nickel-63	3.3E-01	Tritium	3.2E+01
Plutonium-238	6.7E-03	Uranium-233/234	1.7E-02

Table 5-4. Plant-Soil Concentration Ratios (Kp-s) for Garden Produce. (3 Pages)

Analyte	mg/kg dry plant per mg/kg soil	Analyte	mg/kg dry plant per mg/kg soil
Plutonium-239/240	6.7E-03	Uranium-235	1.7E-02
Plutonium-241	6.7E-03	Uranium-238	1.7E-02

¹ Produce plant-soil concentration ratios for radionuclides and metals are suggested values from Table 9 of Wang et al (1993) standardized to dry weight using a conversion factor of 0.15.

² Organic chemical plant-soil concentration ratios based on Equation A-2-17 of EPA/530-R-05-006.

³ Nitrate and nitrite in soil is expected to largely be related to fertilizer amendments. Drought, high temperature, nutrient deficiency, plant damage, and immaturity at harvest can correlate with high nitrate levels in plants (<http://muextension.missouri.edu/explore/agguides/agchem/g09804.htm>).

⁴ Boron value from Figure 2-2 of Baes et al (1984).

Table 5-5. Octanol-Water Partition Coefficients (log Kow). (2 Pages)

Analyte	Log Value	Analyte	Log Value
Acenaphthene	3.9	Dibenz[a,h]anthracene	6.5
Acenaphthylene	4.07	Dibenzofuran	4.12
Acetone	-0.24	Di-n-butylphthalate	4.9
Aldrin	5.5	1,4-Dichlorobenzene	3.5
Anthracene	4.5	1,2-Dichloroethane	1.5
Aroclor 1016	5.7	cis-1,2-Dichloroethene	1.86
Aroclor 1221	4.1	Dichloroprop ¹	3.43
Aroclor 1232	4.5	Dieldrin	4.5
Aroclor 1242	4.1	Diethyl ether	0.89
Aroclor 1248	6.2	Diethylphthalate	2.5
Aroclor 1254	6.5	Di-n-octylphthalate	8.1
Aroclor 1260	6.8	Endosulfan I	3.83
Aroclor 1262 ³	6.8	Endosulfan II	3.83
Aroclor 1268 ³	6.8	Endosulfan sulfate	3.66
Benzene	2.1	Endrin	4.6
Benzo(a)anthracene	5.7	Endrin aldehyde	4.8
Benzo(a)pyrene	6	Endrin ketone ⁴	4.6
Benzo(b)fluoranthene	6.124	Ethylene glycol	-1.36
Benzo(ghi)perylene	6.63	Fluoranthene	5
Benzo(k)fluoranthene	6.1	Fluorene	4.2
Alpha-BHC	3.8	Heptachlor	4.3
beta-1,2,3,4,5,6-Hexachlorocyclohexane	3.78	Heptachlor epoxide	5.4
Delta-BHC	4.14	Indeno(1,2,3-cd)pyrene	6.6
Bis(2-ethylhexyl) phthalate	7.6	Isophorone	1.67
1-Butanol	0.88	Methoxychlor	4.8
2-Butanone	0.29	Methyl isobutyl ketone	1.2
2-Butoxyethanol	0.83	Methylenechloride	1.3
Butylbenzylphthalate	4.91	2-Methylnaphthalene	3.87
2-secButyl-4,6-dinitrophenol(DNBP) ¹	3.56	3+4 Methylphenol (cresol, m+p) ²	2.42
Carbazole ¹	3.72	Naphthalene	3.3
Carbon disulfide	2.2	Pentachlorophenol	5.1
Carbon tetrachloride	2.8	Phenanthrene	4.5
Chlordane	5.5	Phenol	1.5
Chloroform	2	Picloram	0.3
Chrysene	5.7	Pyrene	4.9

Table 5-5. Octanol-Water Partition Coefficients (log Kow). (2 Pages)

Analyte	Log Value	Analyte	Log Value
Dalapon	0.78	1,1,2,2-Tetrachloroethane	3.03
Dicamba	2.21	Tetrachloroethene	3.4
Dichlorodiphenyldichloroethane	6.02	Toluene	2.7
Dichlorodiphenyldichloroethylene	6.51	Trichloroethene	2.61
Dichlorodiphenyltrichloroethane	6.91	Trichloromonofluoromethane	2.5
2,4-Dichlorophenoxyacetic acid	2.81	2,4,5-Trichlorophenoxyacetic acid	4
4-(2,4-Dichlorophenoxy)butanoic acid	3.53	2-(2,4,5-Trichlorophenoxy)propionic acid	3.8

¹ Value was obtained from the PHYSical PROPERTIES database maintained by Syracuse Research Corporation (SRC 2007).

² Value is the midpoint of the range cited in HSDB (NLM 2007).

³ Aroclor 1260 used as a surrogate.

⁴ Endrin used as a surrogate.

Table 5-6. Chicken and Egg Biotransfer Factors. (3 Pages)

Analyte	Chicken mg/kg chckn per mg/kg soil	Egg mg/kg egg per mg/kg soil	Analyte	Chicken mg/kg chckn per mg/kg soil	Egg mg/kg egg per mg/kg soil
INORGANIC CHEMICALS¹					
Aluminum	6.0E-03	7.0E-02	Mercury	1.1E-02	2.0E-01
Antimony	8.3E-01	8.0E-01	Molybdenum	1.9E-01	7.8E-01
Arsenic	8.1E-04	1.5E+00	Nickel	1.0E-03	1.0E-01
Barium	4.0E-01	2.0E-02	Nitrogen in Nitrate	0.0E+00	0.0E+00
Beryllium	(a)	(a)	Nitrogen in Nitrite	0.0E+00	0.0E+00
Boron	8.4E-01	1.0E-01	Selenium	8.5E+00	9.3E+00
Cadmium	2.0E-01	8.0E-01	Silver	5.0E-01	5.0E-01
Chromium	2.0E-01	8.0E-01	Strontium (elemental)	3.5E-02	3.0E-01
Hexavalent Chromium	5.0E-01	1.0E-01	Thallium	3.0E-01	8.0E-01
Cobalt	5.1E-01	4.9E-01	Tin	2.0E-01	8.0E-01
Copper	1.0E-02	2.0E+00	Uranium	1.2E+00	9.9E-01
Fluoride	5.0E-02	6.5E-02	Vanadium	(a)	(a)
Manganese	6.0E-03	7.0E-02	Zinc	6.5E+00	2.6E+00
ORGANIC CHEMICALS¹					
Acenaphthylene	2.0E-02	1.1E-02	Dibenzofuran	2.1E-02	1.2E-02
Acetaldehyde	1.6E-05	9.3E-06	Di-n-butylphthalate	2.8E-02	1.6E-02
Acetone	2.1E-05	1.2E-05	Dicamba	2.9E-03	1.7E-03
Aldrin	3.0E-02	1.7E-02	1,4-Dichlorobenzene	1.3E-02	7.5E-03
Anthracene	2.5E-02	1.4E-02	1,2-Dichloroethane	9.3E-04	5.3E-04
Aroclor 1016	2.9E-02	1.7E-02	cis-1,2-Dichloroethene	1.7E-03	9.8E-04
Aroclor 1221	2.0E-02	1.2E-02	Dichloroprop	1.2E-02	7.1E-03
Aroclor 1232	2.5E-02	1.4E-02	Dieldrin	2.5E-02	1.4E-02
Aroclor 1242	2.0E-02	1.2E-02	Diethyl ether	2.9E-04	1.6E-04
Aroclor 1248	2.6E-02	1.5E-02	Diethylphthalate	4.4E-03	2.5E-03
Aroclor 1254	2.3E-02	1.3E-02	Di-n-octylphthalate	5.7E-03	3.3E-03
Aroclor 1260	1.9E-02	1.1E-02	Endosulfan I	1.7E-02	9.7E-03
Aroclor-1262 ³	1.9E-02	1.1E-02	Endosulfan II	1.7E-02	9.7E-03
Aroclor-1268 ³	1.9E-02	1.1E-02	Endosulfan sulfate	1.5E-02	8.6E-03
Benzene	2.5E-03	1.4E-03	Endrin	2.6E-02	1.5E-02
Benzo(a)anthracene	2.9E-02	1.7E-02	Endrin aldehyde	2.8E-02	1.6E-02
Benzo(a)pyrene	2.8E-02	1.6E-02	Endrin ketone	2.6E-02	1.5E-02

Table 5-6. Chicken and Egg Biotransfer Factors. (3 Pages)

Analyte	Chicken mg/kg chckn per mg/kg soil	Egg mg/kg egg per mg/kg soil	Analyte	Chicken mg/kg chckn per mg/kg soil	Egg mg/kg egg per mg/kg soil
Benzo(b)fluoranthene	2.7E-02	1.5E-02	Ethylene glycol	8.9E-07	5.1E-07
Benzo(ghi)perylene	2.1E-02	1.2E-02	Fluoranthene	2.9E-02	1.7E-02
Benzo(k)fluoranthene	2.7E-02	1.5E-02	Fluorene	2.2E-02	1.2E-02
Alpha-BHC	1.7E-02	9.5E-03	Heptachlor	2.3E-02	1.3E-02
beta-1,2,3,4,5,6- Hexachlorocyclohexane	1.6E-02	9.4E-03	Heptachlor epoxide	3.0E-02	1.7E-02
Delta-BHC	2.1E-02	1.2E-02	Indeno(1,2,3-cd)pyrene	2.2E-02	1.2E-02
Bis(2-ethylhexyl) phthalate	1.0E-02	5.7E-03	Isophorone	1.3E-03	7.1E-04
1-Butanol	2.8E-04	1.6E-04	Methoxychlor	2.8E-02	1.6E-02
2-Butanone	7.7E-05	4.4E-05	Methyl isobutyl ketone	5.3E-04	3.1E-04
2-Butoxyethanol	2.5E-04	1.5E-04	Methylenechloride	6.5E-04	3.7E-04
Butylbenzylphthalate	2.8E-02	1.6E-02	2-Methylnaphthalene	1.8E-02	1.0E-02
2-secButyl-4,6- dinitrophenol(DNBP)	1.4E-02	7.9E-03	3+4 Methylphenol (cresol, m+p)	3.9E-03	2.3E-03
Carbazole	1.6E-02	9.0E-03	Naphthalene	1.1E-02	6.3E-03
Carbon disulfide	2.9E-03	1.7E-03	Pentachlorophenol	2.9E-02	1.7E-02
Carbon tetrachloride	6.4E-03	3.7E-03	Phenanthrene	2.5E-02	1.4E-02
Chlordane	3.0E-02	1.7E-02	Phenol	9.3E-04	5.3E-04
Chloroform	2.1E-03	1.2E-03	Picloram	7.9E-05	4.5E-05
Chrysene	2.9E-02	1.7E-02	Pyrene	2.8E-02	1.6E-02
Dalapon	2.3E-04	1.3E-04	1,1,2,2-Tetrachloroethane	8.3E-03	4.7E-03
Dichlorodiphenyldichloroethane	2.8E-02	1.6E-02	Tetrachloroethene	1.2E-02	6.9E-03
Dichlorodiphenyldichloroethylene	2.3E-02	1.3E-02	Toluene	5.7E-03	3.2E-03
Dichlorodiphenyltrichloroethane	1.8E-02	1.0E-02	Trichloroethene	5.1E-03	2.9E-03
2,4-Dichlorophenoxyacetic acid	6.5E-03	3.7E-03	Trichloromonofluoromethane	4.4E-03	2.5E-03
4-(2,4-Dichlorophenoxy)butanoic acid	1.3E-02	7.7E-03	2,4,5-Trichlorophenoxyacetic acid	1.9E-02	1.1E-02
Dibenz[a,h]anthracene	2.3E-02	1.3E-02	2-(2,4,5- Trichlorophenoxy)propionic acid	1.7E-02	9.5E-03
RADIONUCLIDES¹					
Americium-241	2.0E-04	9.0E-03	Potassium-40	4.0E-01	7.0E-01
Barium-133	8.1E-04	1.5E+00	Radium-226	3.0E-02	2.0E-05
Carbon-14	(b)	(b)	Radium-228	3.0E-02	2.0E-05

Table 5-6. Chicken and Egg Biotransfer Factors. (3 Pages)

Analyte	Chicken mg/kg chckn per mg/kg soil	Egg mg/kg egg per mg/kg soil	Analyte	Chicken mg/kg chckn per mg/kg soil	Egg mg/kg egg per mg/kg soil
Cesium-137	4.4E+00	4.9E-01	Strontium-90	3.5E-02	3.0E-01
Cobalt-60	5.0E-01	1.0E-01	Technetium-99	3.0E-02	3.0E+00
Europium-152	4.0E-03	7.0E-03	Thorium-228	4.0E-03	2.0E-03
Europium-154	4.0E-03	7.0E-03	Thorium-230	4.0E-03	2.0E-03
Europium-155	4.0E-03	7.0E-03	Thorium-232	4.0E-03	2.0E-03
Nickel-63	1.0E-03	1.0E-01	Tritium	(b)	(b)
Plutonium-238	1.5E-04	8.0E-03	Uranium-233/234	1.2E+00	9.9E-01
Plutonium-239/240	1.5E-04	8.0E-03	Uranium-235	1.2E+00	9.9E-01
Plutonium-241	1.5E-04	8.0E-03	Uranium-238	1.2E+00	9.9E-01

¹ Chicken and egg transfer factors for metals and radionuclides from Table 6.18 of NUREG/CR-5512-V1.

² Chicken and egg transfer factors for organic chemicals based on Equations A-2-27 and A-2-28 of EPA/530-R-05-006. Fat content of beef and milk assumed to be 19% and 4%, respectively (EPA/530-R-05-006; A2-2.13.1).

³ Aroclor 1260 used as a surrogate.

(a) No poultry transfer factors available.

(b) Poultry are assumed to be fed store-bought feed. Because soil fractions of H and C are negligible, intake via poultry is not quantified for H and C.

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**Table 5-7. Beef and Milk Biotransfer Factors,
and Grass Plant-Soil Concentration Ratios, for Pinned Cattle.¹ (5 Pages)**

Analyte	Fodder/Soil Conc Ratio mg/kg dry plant per mg/kg soil	Beef Transfer Factor mg/kg beef per mg/day	Milk Transfer Factor mg/kg milk per mg/day
INORGANIC CHEMICALS²			
Aluminum	4.0E-03	5.0E-04	2.0E-04
Antimony	1.0E-01	1.0E-03	1.0E-04
Arsenic	2.0E-01	1.5E-03	1.0E-04
Barium	1.0E-01	2.0E-04	5.0E-04
Beryllium	1.0E-01	1.0E-03	2.0E-06
Boron	4.0E+00	8.0E-04	1.5E-03
Cadmium	1.0E+00	4.0E-04	1.0E-03
Chromium	1.0E-01	9.0E-03	2.0E-03
Hexavalent Chromium	1.0E-01	9.0E-03	2.0E-03
Cobalt	4.0E-01	2.0E-02	2.0E-03
Copper	8.0E-01	1.0E-02	2.0E-03
Fluoride	1.0E-01	2.0E-02	7.0E-03
Manganese	9.2E-01	5.0E-04	3.0E-04
Mercury	1.0E+00	1.0E-01	5.0E-04
Molybdenum	4.0E-01	1.0E-03	1.7E-03
Nickel	1.1E-01	5.0E-03	2.0E-02
Nitrogen in Nitrate	0.0E+00	0.0E+00	0.0E+00
Nitrogen in Nitrite	0.0E+00	0.0E+00	0.0E+00
Selenium	5.0E-01	1.0E-01	1.0E-02
Silver	1.0E-01	3.0E-03	2.5E-02
Strontium (elemental)	2.0E+00	8.0E-03	2.0E-03
Thallium	4.0E-03	2.0E-02	3.0E-03
Tin	1.0E+00	1.0E-02	1.0E-03
Uranium	1.0E-01	3.4E-04	6.0E-04
Vanadium	5.5E-03	1.0E-02	5.0E-04
Zinc	5.0E-01	1.0E-01	1.0E-02
ORGANIC CHEMICALS³			
Acenaphthylene	1.7E-01	2.7E-02	5.7E-03
Acetaldehyde	6.1E+01	2.2E-05	4.6E-06
Acetone	5.3E+01	2.9E-05	6.0E-06
Aldrin	2.6E-02	4.1E-02	8.6E-03
Anthracene	9.7E-02	3.4E-02	7.1E-03

Table 5-7. Beef and Milk Biotransfer Factors, and Grass Plant-Soil Concentration Ratios, for Penned Cattle.¹ (5 Pages)

Analyte	Fodder/Soil Conc Ratio mg/kg dry plant per mg/kg soil	Beef Transfer Factor mg/kg beef per mg/day	Milk Transfer Factor mg/kg milk per mg/day
Aroclor 1016	2.0E-02	4.0E-02	8.4E-03
Aroclor 1221	1.7E-01	2.7E-02	5.8E-03
Aroclor 1232	9.2E-02	3.4E-02	7.2E-03
Aroclor 1242	1.6E-01	2.8E-02	5.9E-03
Aroclor 1248	1.0E-02	3.5E-02	7.4E-03
Aroclor 1254	6.8E-03	3.1E-02	6.5E-03
Aroclor 1260	4.5E-03	2.6E-02	5.5E-03
Aroclor 1262 ⁴	4.5E-03	2.6E-02	5.5E-03
Aroclor 1268 ⁴	4.5E-03	2.6E-02	5.5E-03
Benzene	2.4E+00	3.4E-03	7.1E-04
Benzo(a)anthracene	2.0E-02	4.0E-02	8.4E-03
Benzo(a)pyrene	1.3E-02	3.8E-02	7.9E-03
Benzo(b)fluoranthene	1.1E-02	3.6E-02	7.6E-03
Benzo(ghi)perylene	5.7E-03	2.9E-02	6.1E-03
Benzo(k)fluoranthene	1.2E-02	3.6E-02	7.7E-03
Alpha-BHC	2.5E-01	2.3E-02	4.8E-03
beta-1,2,3,4,5,6-Hexachlorocyclohexane	2.5E-01	2.2E-02	4.7E-03
Delta-BHC	1.6E-01	2.8E-02	6.0E-03
Bis(2-ethylhexyl) phthalate	1.6E-03	1.4E-02	2.9E-03
1-Butanol	1.2E+01	3.8E-04	8.1E-05
2-Butanone	2.6E+01	1.0E-04	2.2E-05
2-Butoxyethanol	1.3E+01	3.5E-04	7.3E-05
Butylbenzylphthalate	5.6E-02	3.9E-02	8.1E-03
2-secButyl-4,6-dinitrophenol(DNBP)	3.4E-01	1.9E-02	3.9E-03
Carbazole	2.7E-01	2.1E-02	4.5E-03
Carbon disulfide	2.1E+00	3.9E-03	8.3E-04
Carbon tetrachloride	9.3E-01	8.7E-03	1.8E-03
Chlordane	2.6E-02	4.1E-02	8.6E-03
Chloroform	2.7E+00	2.9E-03	6.1E-04
Chrysene	2.0E-02	4.0E-02	8.4E-03
Dalapon	1.4E+01	3.1E-04	6.6E-05
Dichlorodiphenyldichloroethane	1.3E-02	3.7E-02	7.9E-03
Dichlorodiphenyldichloroethylene	6.7E-03	3.1E-02	6.5E-03
Dichlorodiphenyltrichloroethane	3.9E-03	2.4E-02	5.1E-03

Table 5-7. Beef and Milk Biotransfer Factors, and Grass Plant-Soil Concentration Ratios, for Penned Cattle.¹ (5 Pages)

Analyte	Fodder/Soil Conc Ratio mg/kg dry plant per mg/kg soil	Beef Transfer Factor mg/kg beef per mg/day	Milk Transfer Factor mg/kg milk per mg/day
2,4-Dichlorophenoxyacetic acid	9.2E-01	8.8E-03	1.8E-03
4-(2,4-Dichlorophenoxy)butanoic acid	3.5E-01	1.8E-02	3.9E-03
Dibenz[a,h]anthracene	6.8E-03	3.1E-02	6.5E-03
Dibenzofuran	1.6E-01	2.8E-02	5.9E-03
Di-n-butylphthalate	5.7E-02	3.8E-02	8.1E-03
Dicamba	2.0E+00	4.0E-03	8.4E-04
1,4-Dichlorobenzene	3.7E-01	1.8E-02	3.8E-03
1,2-Dichloroethane	5.3E+00	1.3E-03	2.7E-04
cis-1,2-Dichloroethene	3.3E+00	2.3E-03	4.9E-04
Dichloroprop	4.0E-01	1.7E-02	3.5E-03
Dieldrin	9.7E-02	3.4E-02	7.1E-03
Diethyl ether	1.2E+01	3.9E-04	8.2E-05
Diethylphthalate	1.4E+00	6.0E-03	1.3E-03
Di-n-octylphthalate	8.1E-04	7.8E-03	1.6E-03
Endosulfan I	2.4E-01	2.3E-02	4.9E-03
Endosulfan II	2.4E-01	2.3E-02	4.9E-03
Endosulfan sulfate	3.0E-01	2.0E-02	4.3E-03
Endrin	8.5E-02	3.5E-02	7.4E-03
Endrin aldehyde	6.5E-02	3.7E-02	7.9E-03
Endrin ketone	8.5E-02	3.5E-02	7.4E-03
Ethylene glycol	2.4E+02	1.2E-06	2.5E-07
Fluoranthene	5.0E-02	3.9E-02	8.3E-03
Fluorene	1.4E-01	2.9E-02	6.2E-03
Heptachlor	1.3E-01	3.1E-02	6.5E-03
Heptachlor epoxide	2.9E-02	4.1E-02	8.6E-03
Indeno(1,2,3-cd)pyrene	5.9E-03	2.9E-02	6.2E-03
Isophorone	4.2E+00	1.7E-03	3.6E-04
Methoxychlor	6.5E-02	3.7E-02	7.9E-03
Methyl isobutyl ketone	7.8E+00	7.2E-04	1.5E-04
Methylenechloride	6.9E+00	8.8E-04	1.8E-04
2-Methylnaphthalene	2.2E-01	2.4E-02	5.0E-03
3+4 Methylphenol (cresol, m+p)	1.5E+00	5.4E-03	1.1E-03
Naphthalene	4.8E-01	1.5E-02	3.1E-03

Table 5-7. Beef and Milk Biotransfer Factors, and Grass Plant-Soil Concentration Ratios, for Penned Cattle.¹ (5 Pages)

Analyte	Fodder/Soil Conc Ratio mg/kg dry plant per mg/kg soil	Beef Transfer Factor mg/kg beef per mg/day	Milk Transfer Factor mg/kg milk per mg/day
Pentachlorophenol	4.4E-02	4.0E-02	8.4E-03
Phenanthrene	9.7E-02	3.4E-02	7.1E-03
Phenol	5.3E+00	1.3E-03	2.7E-04
Picloram	2.6E+01	1.1E-04	2.3E-05
Pyrene	5.7E-02	3.8E-02	8.1E-03
1,1,2,2-Tetrachloroethane	6.9E-01	1.1E-02	2.4E-03
Tetrachloroethene	4.2E-01	1.6E-02	3.4E-03
Toluene	1.1E+00	7.7E-03	1.6E-03
Trichloroethene	1.2E+00	6.9E-03	1.4E-03
Trichloromonofluoromethane	1.4E+00	6.0E-03	1.3E-03
2,4,5-Trichlorophenoxyacetic acid	1.9E-01	2.6E-02	5.5E-03
2-(2,4,5-Trichlorophenoxy)propionic acid	2.5E-01	2.3E-02	4.8E-03
RADIONUCLIDES²			
Americium-241	4.0E-03	5.0E-05	2.0E-06
Barium-133	1.0E-01	2.0E-04	5.0E-04
Carbon-14	7.0E-01	3.1E-02	1.2E-02
Cesium-137	2.0E-01	3.0E-02	8.0E-03
Cobalt-60	4.0E-01	2.0E-02	2.0E-03
Europium-152	1.0E-01	2.0E-03	2.0E-05
Europium-154	1.0E-01	2.0E-03	2.0E-05
Europium-155	1.0E-01	2.0E-03	2.0E-05
Nickel-63	1.1E-01	5.0E-03	2.0E-02
Plutonium-238	2.7E-04	1.0E-04	1.0E-06
Plutonium-239/240	2.7E-04	1.0E-04	1.0E-06
Plutonium-241	2.7E-04	1.0E-04	1.0E-06
Potassium-40	3.0E+00	2.0E-02	7.0E-03
Radium-226	2.0E-01	1.0E-03	1.0E-03
Radium-228	2.0E-01	1.0E-03	1.0E-03
Strontium-90	2.0E+00	8.0E-03	2.0E-03
Technetium-99	4.0E+01	1.0E-04	1.0E-03
Thorium-228	9.0E-03	1.0E-04	5.0E-06
Thorium-230	9.0E-03	1.0E-04	5.0E-06
Thorium-232	9.0E-03	1.0E-04	5.0E-06

Table 5-7. Beef and Milk Biotransfer Factors, and Grass Plant-Soil Concentration Ratios, for Penned Cattle.¹ (5 Pages)

Analyte	Fodder/Soil Conc Ratio mg/kg dry plant per mg/kg soil	Beef Transfer Factor mg/kg beef per mg/day	Milk Transfer Factor mg/kg milk per mg/day
Tritium	7.2E-01	1.2E-02	1.0E-02
Uranium-233/234	1.0E-01	3.4E-04	6.0E-04
Uranium-235	1.0E-01	3.4E-04	6.0E-04
Uranium-238	1.0E-01	3.4E-04	6.0E-04

¹ Local Area soil concentrations for penned cattle are averaged over a 2-ha cattle enclosure.

² Fodder-soil concentration ratios for metals and radionuclides are from Table 10 of Wang et al. (1993). Beef transfer factors for metals and radionuclides are suggested values from Table 11 of Wang et al. (1993). Milk transfer factors for metals and radionuclides are suggested values from Table 12 of Wang et al. (1993), standardized to units of mg/kg assuming a density of 1 kg/L.

³ Fodder-soil concentration ratios for metals and radionuclides are from Table 10 of Wang et al. (1993). Beef and milk transfer factors for organic chemicals based on Equations A-2-27 and A-2-28 of EPA/530-R-05-006. Fat content of beef and milk assumed to be 19% and 4%, respectively (EPA/530-R-05-006; A2-2.13.1).

⁴ Aroclor 1260 used as a surrogate.

Table 5-8. RME Exposure Parameter Values. (2 Pages)

Parameter Name ¹	Units	Rural Residential	CTUIR ²	Resident Monument Worker	Industrial / Commercial	Recreational (Avid Hunting)	Recreational (Avid Fishing)	Recreational (Casual Use)
IRs,d_adult	mg/day	100	400	100	100	100	100	100
IRs,d_child, 1-6	mg/day	200	400	na	na	na	na	200
IRs,d_child, 7-12	mg/day	na	na	na	na	200	200	na
InhR_adult	m ³ /hr	0.63	1.25	0.63	1.0	1.6	1.0	1.0
InhR_child, 1-6	m ³ /hr	0.34	0.34	na	na	na	na	1.0
InhR_child, 7-12	m ³ /hr	na	na	na	na	1.2	1.0	na
SAs_adult	cm ²	5700	5700	5700 / 3300	3300	3300	5700	5700
SAs_child, 0-6	cm ²	2800	2800	na	na	na	na	2800
SAs_child, 7-12	cm ²	na	na	na	na	1500	3400	na
SAw_adult	cm ²	18,000	18,000	18,000 / na	na	na	na	na
SAw_child	cm ²	6,600	6,600	na	na	na	na	na
AF_adult	mg/cm ²	0.07	0.1	0.07 / 0.07	0.04	0.07	0.07	0.07
AF_child	mg/cm ²	0.2	0.2	na	na	0.2	0.2	0.2
IRw_adult	L/day	2.3	3 (a)	2.3 / na	na	na	na	na
IRw_child	L/day	0.9	1.5	na	na	na	na	na
IRveg (wet wt)	g/kg-day	0.18	17.5	na	na	na	na	na
IRfruit (wet wt)	g/kg-day	0.21	1.8	na	na	na	na	na
IRchick (wet wt)	g/kg-day	1.3	0.89	na	na	na	na	na
IRegg (wet wt)	g/kg-day	1.05	na	na	na	na	na	na
IRbeef (wet wt)	g/kg-day	2.2	0.89	na	na	na	na	na
IRmilk (wet wt)	g/kg-day	19.5	na	na	na	na	na	na
IRfish (wet wt)	g/kg-day	0.86	8.9	na	na	na	3.3	na
IRgame (wet wt)	g/kg-day	na	1.8	na	na	2.2	na	na
Tsite_child	hr	24	24	na	na	8	8	6
Tsite_adult	hr	21	24	13 / 8	8	8	8	6

Table 5-8. RME Exposure Parameter Values. (2 Pages)

Parameter Name ¹	Units	Rural Residential	CTUIR ²	Resident Monument Worker	Industrial/ Commercial	Recreational (Avid Hunting)	Recreational (Avid Fishing)	Recreational (Casual Use)
Tsite,out_child	hr	3	3	na	na	8	8	6
Tsite,out_adult	hr	3	3	3 / 8	8	8	8	6
Tsleep_child	hr	11.5	11.5	na	na	11.5	11.5	11.5
Tsleep_adult	hr	8	8	8	8	8	8	8
t_event,bath_child	hr/event	1.0	1.0	na	na	na	na	na
t_event,bath_adult	hr/event	0.58	0.58	0.58 / na	na	na	na	na
t_event,sweat	hr/event	na	1	na	na	na	na	na
EF_adult	day/yr	350	365	350 / 250	250	30	60	30
EF_child	day/yr	350	365	na	na	30	60	30
EF_food	day/yr	365	365	na	na	365	365	365
EF_sweat	day/yr	na	365	na	na	na	na	na
EV_f,bath	event/day	1	1	1 / na	na	na	na	na
ED_adult	yr	30	70	25 / 25	25	30	30	30
ED_child	yr	6	6	na	na	6	6	6
ED_sweat	yr	na	68	na	na	na	na	na
BW_adult	kg	70	70	70	70	70	70	70
BW_child, 1-6	kg	17.8	17.8	na	na	na	na	16.6
BW_child, 7-12	kg	na	na	na	na	34.6	34.6	na
AT_carc	yr	70	70	70	70	70	70	70
AT_nc	yr	= ED	= ED	= ED	= ED	= ED	= ED	= ED

¹ Definitions of parameter names are provided in Section 5.4 of the risk assessment.

² CTUIR values were obtained from Harris and Harper (2004). These values are not necessarily consistent with "reasonable" maximum exposure conditions as might be defined by using demographic data for this population.

a. An additional 1 L/day is consumed during sweat lodge use, for a total of 4 L across domestic and sweat lodge exposures.

CTUIR = Confederated Tribes of the Umatilla Indian Reservation

RME = reasonable maximum exposure

Table 5-9. CTE Exposure Parameter Values. (2 Pages)

Parameter Name ¹	Units	Rural Residential	CTUIR ²	Resident Monument Worker	Industrial/ Commercial	Recreational (Avid Hunting)	Recreational (Avid Fishing)	Recreational (Casual Use)
IRs,d_adult	mg/day	50		50	50	50	50	50
IRs,d_child, 1-6	mg/day	100		na	na	na	na	100
IRs,d_child, 7-12	mg/day	na		na	na	100	100	na
InhR_adult	m ³ /hr	0.63		0.63	0.63	1.0	1.0	1.0
InhR_child, 1-6	m ³ /hr	0.34		na	na	na	na	1.0
InhR_child, 7-12	m ³ /hr	na		na	na	1.0	1.0	na
SAs_adult	cm ²	5700		5700 / 3300	3300	3300	5700	5700
SAs_child, 0-6	cm ²	2800		na	na	na	na	2800
SAs_child, 7-12	cm ²	na		na	na	1500	3400	na
SAw_adult	cm ²	18,000		18,000 / na	na	na	na	na
SAw_child	cm ²	6,600		na	na	na	na	na
AF_adult	mg/ cm ²	0.04		0.04 / 0.04	0.01	0.04	0.04	0.04
AF_child	mg/ cm ²	0.04		na	na	0.04	0.04	0.04
IRw_adult	L/day	1.4		1.4 / na	na	na	na	na
IRw_child	L/day	0.3		na	na	na	na	na
IRveg (wet wt)	g/kg-day	0.06		na	na	na	na	na
IRfruit (wet wt)	g/kg-day	0.08		na	na	na	na	na
IRchick (wet)	g/kg-day	0.70		na	na	na	na	na
IRegg (wet)	g/kg-day	0.67		na	na	na	na	na
IRbeef (wet)	g/kg-day	1.2		na	na	na	na	na
IRmilk (wet)	g/kg-day	10.2		na	na	na	na	na
IRfish (wet)	g/kg-day	0.49		na	na	na	0.49	na
IRgame (wet)	g/kg-day	na		na	na	0.57	na	na

Table 5-9. CTE Exposure Parameter Values. (2 Pages)

Parameter Name ¹	Units	Rural Residential	CTUIR ²	Resident Monument Worker	Industrial / Commercial	Recreational (Avid Hunting)	Recreational (Avid Fishing)	Recreational (Casual Use)
Tsite_child	hr	24		na	na	8	5	3
Tsite_adult	hr	21		13 / 8	8	8	5	3
Tsite_out_child	hr	1.7		na	na	8	5	3
Tsite_out_adult	hr	1.5		1.5 / 8	8	8	5	3
Tsleep_child	hr	11.5		na	na	11.5	11.5	11.5
Tsleep_adult	hr	8		8	8	8	8	8
t_event_bath_child	hr/event	0.33		na	na	na	na	na
t_event_bath_adult	hr/event	0.25		0.25 / na	na	na	na	na
t_event_sweat	hr/event	na		na	na	na	na	na
EF_adult	day/yr	350		350 / 250	250	10	30	10
EF_child	day/yr	350		na	na	10	30	10
EF_food	day/yr	365		na	na	365	365	365
EF_sweat	day/yr	na		na	na	na	na	na
EV_f_bath	event/day	1		1 / na	na	na	na	na
ED_adult	yr	9		6.6 / 6.6	6.6	9	9	9
ED_child	yr	6		na	na	6	6	6
ED_sweat	yr	na		na	na	na	na	na
BW_adult	kg	70		70	70	70	70	70
BW_child, 1-6	kg	17.8		na	na	na	na	16.6
BW_child, 7-12	kg	na		na	na	34.6	34.6	na
AT_carc	yr	70		70	70	70	70	70
AT_nc	yr	= ED		= ED	= ED	= ED	= ED	= ED

¹ Definitions of parameter names are provided in Section 5.4 of the risk assessment.

² Only a single set of exposure parameter values is provided in Harris and Harper (2004).

CTE = central tendency exposure

CTUIR = Confederated Tribes of the Umatilla Indian Reservation

Table 5-10. Bioavailability Values (Gastrointestinal Absorption) for Soil Ingestion.

Analyte	Relative Bioavailability
Antimony	15%
Barium	7%
Beryllium	0.7%
Cadmium	2.5%
Chromium III	1.3%
Chromium VI	2.5%
Lead	60%
Manganese	4%
Mercury (salts)	7%
Nickel	4%
Silver	4%
Vanadium	2.6%
All other metals	100%

Gastrointestinal absorption factors from Exhibit 4-1 of RAGS Part E (EPA/540/R/99/005).

Table 5-11. Dermal Absorption Fraction Values for Soil Contact.

Analyte	ABS _d Value
Arsenic	0.03
Cadmium	0.001
Chlordane	0.04
2,4-Dichlorophenoxyacetic acid	0.05
DDT	0.003
Dioxins ¹	0.03 / 0.001
Lindane	0.04
Polyaromatic hydrocarbons	0.13
Polychlorinated biphenyls	0.14
Pentachlorophenol	0.25
All other semivolatile organic compounds	0.1

¹ Second value to be applied if soil organic carbon fraction >10%.

Dermal absorption fractions from Exhibit 3-4 of RAGS Part E (EPA/540/R/99/005).

Table 5-12. Chemical-Specific Values for Calculating Dermal Absorption from Water.
(4 Pages)

Analyte	Permeability Coefficient (Kp) cm/hr	Lag Time (τ) hr/event	Stratum Corneum/Epidermis Permeability (B) unitless	Time to Steady State (t^*) hr	Absorbed Fraction (FA) unitless
INORGANIC CHEMICALS					
Aluminum	0.001				
Antimony	0.001				
Arsenic	0.001				
Barium	0.001				
Beryllium	0.001				
Boron	0.001				
Cadmium	0.001				
Chromium	0.001				
Hexavalent Chromium	0.002				
Cobalt	0.0004				
Copper	0.001				
Fluoride	0.001				
Manganese	0.001				
Mercury	0.001				
Molybdenum	0.001				
Nickel	0.0002				
Nitrogen in Nitrate	0.001				
Nitrogen in Nitrite	0.001				
Selenium	0.001				
Silver	0.0006				
Strontium (elemental)	0.001				
Thallium	0.001				
Uranium	0.001				
Vanadium	0.001				
Zinc	0.0006				
ORGANIC CHEMICALS					
Acenaphthylene	1.1E-01	7.5E-01	5.1E-01	1.8E+00	1
Acetaldehyde	5.4E-04	1.9E-01	1.4E-03	4.4E-01	1
Acetone	5.2E-04	2.2E-01	1.5E-03	5.3E-01	1
Aldrin	6.1E-02	1.2E+01	4.5E-01	2.8E+01	1
Anthracene	1.5E-01	1.0E+00	7.6E-01	4.0E+00	1
Aroclor 1242	1.9E-02	4.5E+00	1.2E-01	1.1E+01	1

**Table 5-12. Chemical-Specific Values for Calculating Dermal Absorption from Water.
(4 Pages)**

Analyte	Permeability Coefficient (Kp) cm/hr	Lag Time (τ) hr/event	Stratum Corneum/Epidermis Permeability (B) unitless	Time to Steady State (t*) hr	Absorbed Fraction (FA) unitless
Aroclor 1248	4.5E-01	4.5E+00	3.0E+00	1.9E+01	1
Aroclor 1254	4.6E-01	7.1E+00	3.2E+00	3.0E+01	1
Aroclor 1260	3.0E-01	1.7E+01	2.3E+00	7.1E+01	1
Aroclor-1262	3.0E-01	1.7E+01	2.3E+00	7.1E+01	1
Benzene	1.4E-02	2.9E-01	4.8E-02	6.9E-01	1
Benzo(a)anthracene	4.8E-01	2.0E+00	2.8E+00	8.4E+00	1
Benzo(a)pyrene	5.6E-01	2.7E+00	3.4E+00	1.2E+01	1
Benzo(b)fluoranthene	6.7E-01	2.7E+00	4.1E+00	1.2E+01	1
Benzo(ghi)perylene	1.1E+00	3.7E+00	6.8E+00	1.6E+01	1
Benzo(k)fluoranthene	6.5E-01	2.7E+00	4.0E+00	1.2E+01	1
Alpha-BHC	1.2E-02	4.5E+00	7.9E-02	1.1E+01	1
beta-1,2,3,4,5,6-Hexachlorocyclohexane	1.2E-02	4.5E+00	7.6E-02	1.1E+01	1
Delta-BHC	2.0E-02	4.5E+00	1.3E-01	1.1E+01	1
Bis(2-ethylhexyl) phthalate	1.1E+00	1.6E+01	8.1E+00	7.3E+01	1
1-Butanol	2.3E-03	2.7E-01	7.7E-03	6.6E-01	1
2-Butanone	9.7E-04	2.7E-01	3.2E-03	6.4E-01	1
2-Butoxyethanol	1.2E-03	4.8E-01	5.1E-03	1.2E+00	1
Butylbenzylphthalate	4.9E-02	5.9E+00	3.3E-01	1.4E+01	1
2-secButyl-4,6-dinitrophenol(DNBP)	1.6E-02	2.3E+00	9.5E-02	5.6E+00	1
Carbazole	5.2E-02	9.1E-01	2.6E-01	2.2E+00	1
Carbon disulfide	1.7E-02	2.8E-01	5.6E-02	6.7E-01	1
Carbon tetrachloride	1.5E-02	7.6E-01	7.3E-02	1.8E+00	1
Chlordane	3.4E-02	2.1E+01	2.7E-01	5.0E+01	1
Chloroform	7.1E-03	4.9E-01	3.0E-02	1.2E+00	1
Chrysene	4.8E-01	2.0E+00	2.8E+00	8.4E+00	1
Dalapon	8.2E-04	6.6E-01	3.8E-03	1.6E+00	1
Dichlorodiphenyldichloroethane	2.4E-01	6.5E+00	1.7E+00	2.6E+01	1
Dichlorodiphenyldichloroethylene	5.2E-01	6.3E+00	3.6E+00	2.7E+01	1
Dichlorodiphenyltrichloroethane	6.0E-01	1.0E+01	4.3E+00	4.4E+01	1
2,4-Dichlorophenoxyacetic acid	6.6E-03	1.8E+00	3.8E-02	4.4E+00	1
4-(2,4-Dichlorophenoxy)butanoic acid	3.8E-03	9.5E+00	2.7E-02	2.3E+01	1
Dibenz[a,h]anthracene	8.5E-01	3.8E+00	5.5E+00	1.7E+01	1
Dibenzofuran	9.5E-02	9.2E-01	4.7E-01	2.2E+00	1

**Table 5-12. Chemical-Specific Values for Calculating Dermal Absorption from Water.
(4 Pages)**

Analyte	Permeability Coefficient (Kp) cm/hr	Lag Time (τ) hr/event	Stratum Corneum/Epidermis Permeability (B) unitless	Time to Steady State (t^*) hr	Absorbed Fraction (FA) unitless
Di-n-butylphthalate	7.5E-02	3.8E+00	4.8E-01	9.1E+00	1
Dicamba	2.6E-03	1.8E+00	1.5E-02	4.4E+00	1
1,4-Dichlorobenzene	4.9E-02	7.0E-01	2.3E-01	1.7E+00	1
1,2-Dichloroethane	4.3E-03	3.8E-01	1.7E-02	9.0E-01	1
cis-1,2-Dichloroethene	7.7E-03	3.7E-01	2.9E-02	8.8E-01	1
Dichloroprop	1.4E-02	2.2E+00	8.3E-02	5.2E+00	1
Dieldrin	1.1E-02	1.4E+01	8.2E-02	3.4E+01	1
Diethyl ether	2.4E-03	2.7E-01	7.8E-03	6.6E-01	1
Diethylphthalate	4.0E-03	1.8E+00	2.3E-02	4.4E+00	1
Di-n-octylphthalate	2.3E+00	1.6E+01	1.7E+01	7.4E+01	1
Endosulfan I	2.8E-03	2.0E+01	2.2E-02	4.8E+01	1
Endosulfan II	2.8E-03	2.0E+01	2.2E-02	4.8E+01	1
Endosulfan sulfate	2.2E-03	2.0E+01	1.7E-02	4.8E+01	1
Endrin	1.3E-02	1.4E+01	9.5E-02	3.4E+01	1
Endrin aldehyde	1.7E-02	1.4E+01	1.3E-01	3.4E+01	1
Endrin ketone	1.3E-02	1.4E+01	9.5E-02	3.4E+01	1
Ethylene glycol	9.0E-05	2.3E-01	2.7E-04	5.6E-01	1
Fluoranthene	2.3E-01	1.4E+00	1.3E+00	5.6E+00	1
Fluorene	1.1E-01	9.0E-01	5.5E-01	2.1E+00	1
Heptachlor	8.9E-03	1.3E+01	6.6E-02	3.1E+01	1
Heptachlor epoxide	3.8E-02	1.6E+01	2.9E-01	3.8E+01	1
Indeno(1,2,3-cd)pyrene	1.0E+00	3.7E+00	6.5E+00	1.6E+01	1
Isophorone	3.4E-03	6.2E-01	1.5E-02	1.5E+00	1
Methoxychlor	2.7E-02	9.1E+00	1.9E-01	2.2E+01	1
Methyl isobutyl ketone	2.7E-03	3.8E-01	1.0E-02	9.2E-01	1
Methylenechloride	3.8E-03	3.1E-01	1.4E-02	7.5E-01	1
2-Methylnaphthalene	9.1E-02	6.6E-01	4.2E-01	1.6E+00	1
3+4 Methylphenol (cresol, m+p)	1.3E-02	5.1E-01	5.5E-02	1.2E+00	1
Naphthalene	4.6E-02	5.5E-01	2.0E-01	1.3E+00	1
Pentachlorophenol	1.2E-01	3.3E+00	7.5E-01	1.3E+01	1
Phenanthrene	1.5E-01	1.0E+00	7.6E-01	4.0E+00	1
Phenol	4.6E-03	3.5E-01	1.7E-02	8.5E-01	1
Picloram	1.1E-04	2.4E+00	6.6E-04	5.7E+00	1

Table 5-12. Chemical-Specific Values for Calculating Dermal Absorption from Water.
(4 Pages)

Analyte	Permeability Coefficient (Kp) cm/hr	Lag Time (τ) hr/event	Stratum Corneum/Epidermis Permeability (B) unitless	Time to Steady State (t^*) hr	Absorbed Fraction (FA) unitless
Pyrene	2.0E-01	1.4E+00	1.1E+00	5.5E+00	1
1,1,1,2-Tetrachloroethane	1.6E-02	9.1E-01	7.8E-02	2.2E+00	1
Tetrachloroethene	3.3E-02	8.9E-01	1.6E-01	2.1E+00	1
Toluene	2.9E-02	3.4E-01	1.1E-01	8.3E-01	1
Trichloroethene	1.5E-02	5.7E-01	6.8E-02	1.4E+00	1
Trichloromonofluoromethane	1.2E-02	6.2E-01	5.4E-02	1.5E+00	1
2,4,5-Trichlorophenoxyacetic acid	2.6E-02	2.8E+00	1.6E-01	6.8E+00	1
2-(2,4,5-Trichlorophenoxy)propionic acid	1.6E-02	3.4E+00	1.0E-01	8.1E+00	1

Kp for inorganic chemicals from Exhibit 3-1 of RAGS, Part E (EPA/540/R/99/005).

Kp for organic chemicals from log Kow following Equation 3.8 in Appendix A of EPA/540/R/99/005.

Lag time (τ), a measure of the rate at which an absorbed chemical in the skin is released to the bloodstream, was calculated according to Equation A.4 in RAGS, Part E (EPA/540/R/99/005).

B, the ratio of the permeability through the stratum corneum and the viable epidermis, was calculated according to Equation A.1 in RAGS, Part E (EPA/540/R/99/005).

t^* , the exposure time required to reach steady-state skin concentrations, was calculated as a function of B according to Equations A.5 - A.8 in RAGS, Part E (EPA/540/R/99/005).

FA, the fraction of chemical dissolved in skin that is absorbed into the bloodstream, was obtained from Appendix B of RAGS, Part E (EPA/540/R/99/005).

Table 5-13. Chronic Oral Reference Doses for Inorganic Chemicals. (2 Pages)

Chemical	Oral RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Aluminum	1.0	mouse	minimal neurotoxicity in offspring	100	1	EPA Region 6 HHMSSLs, 2007	Tier 2 (PPRTV)
Antimony	0.0004	rat	longevity, blood glucose, and cholesterol	1000	1	IRIS, 2007	Tier 1 (IRIS)
Arsenic	0.0003	human	hyperpigmentation, keratosis, and possible vascular complications	3	1	IRIS, 2007	Tier 1 (IRIS)
Barium	0.2	mouse	nephropathy	300		IRIS, 2007	Tier 1 (IRIS)
Beryllium	0.002	dog	small intestinal lesions	300	1	IRIS, 2007	Tier 1 (IRIS)
Boron	0.2	rat	decreased fetal weight (developmental)	66		IRIS, 2007	Tier 1 (IRIS)
Cadmium	0.001	human	significant proteinuria; food exposure	10	1	IRIS, 2007	Tier 1 (IRIS); Tier 1 (IRIS) value of 0.0005 is applied for water ingestion
Chromium	1.5	rat	no effects observed	100	10	IRIS, 2007	Tier 1 (IRIS); trivalent chromium
Hexavalent Chromium	0.003	rat	none reported	300	3	IRIS, 2007	Tier 1 (IRIS)
Cobalt	0.02					EPA Region 6 HHMSSLs, 2007	Tier 2 (PPRTV)
Copper	0.04	human	gastrointestinal system irritation	none	none	EPA Region 6 HHMSSLs, 2007	Tier 3 (HEAST)
Fluoride	0.06	human	objectionable dental fluorosis; cosmetic effect	1	1	IRIS, 2007	Tier 1 (IRIS); soluble fluoride
Lead							See Section 5.5.8
Manganese	0.14	human	CNS effects	1	1	IRIS, 2007	Tier 1 (IRIS)
Mercury	0.0003	rat	autoimmune effects; mercuric chloride (HgCL)	1000	1	IRIS, 2007	Tier 1 (IRIS); HgCl
Molybdenum	0.005	human	increased uric acid levels	30	1	IRIS, 2007	Tier 1 (IRIS)

Table 5-13. Chronic Oral Reference Doses for Inorganic Chemicals. (2 Pages)

Chemical	Oral RfD (mg/kg-day)	Species	Critical Effect	UF	MIF	Source	Notes
Nickel	0.02	rat	decreased body and organ weights	300	1	IRIS, 2007	Tier 1 (IRIS); nickel soluble salts
Nitrogen in Nitrate	1.6	human	early clinical signs of methemoglobinemia in excess of 10% (0-3 months old infants formula)	1	1	IRIS, 2007	Tier 1 (IRIS)
Nitrogen in Nitrite	0.1	human	methemoglobinemia	1	10	IRIS, 2007	Tier 1 (IRIS)
Selenium	0.005	human	clinical selenosis	3	1	IRIS, 2007	Tier 1 (IRIS)
Silver	0.005	human	argyria	3	1	IRIS, 2007	Tier 1 (IRIS)
Strontium (elemental)	0.6	rat	rachitic bone	300	1	IRIS, 2007	Tier 1 (IRIS)
Thallium	0.00008	rat	increased levels of SGOT and LDH	3000	1	IRIS, 2007	Tier 1 (IRIS); carbonate, chloride, or sulfate
Uranium	0.003	rabbit	initial body weight loss; moderate nephrotoxicity	1000	1	IRIS, 2007	Tier 1 (IRIS); uranium soluble salts
Vanadium	0.007	rat	decreased hair cysteine	100	none	IRIS, 2007	Tier 1 (IRIS); vanadium pentoxide
Zinc	0.3	human	decreases in erythrocyte Cu, Zn-superoxide dismutase (ESOD) activity in healthy adult male and female volunteers	3		IRIS, 2007	Tier 1 (IRIS)

Table 5-14. Chronic Oral Reference Doses for Organic Chemicals. (7 Pages)

Chemical	Oral RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
1,1,2,2-Tetrachloroethane							Tier 2 (PPRTV) oral RfD not used because oral route covered by Tier 1 (IRIS) oral SF
1,2-Dichloroethane							Tier 3 (NCEA) oral RfD not used because oral route covered by Tier 1 (IRIS) oral SF
1,4-Dichlorobenzene	2.29E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation
1-Butanol	1.00E-01	rat	hypoactivity and ataxia	1000	1	IRIS, 2007	Tier 1 (IRIS)
2-(2,4,5-Trichlorophenoxy)propionic acid	8.00E-03	dog	histopathological changes in liver	100	1	IRIS, 2007	Tier 1 (IRIS)
2,4,5-Trichlorophenoxyacetic acid	1.00E-02	rat	increased urinary coproporphyrins (other effect: Reduced neonatal survival.)	300	1	IRIS, 2007	Tier 1 (IRIS)
2,4-Dichlorophenoxyacetic acid	1.00E-02	rat	hematologic, hepatic and renal toxicity	100	1	IRIS, 2007	Tier 1 (IRIS)
2-Butanone	6.00E-01	rat	decreased pup body weight	1000	1	IRIS, 2007	Tier 1 (IRIS)
2-Butoxyethanol	5.00E-01	rat and mice	Changes in mean corpuscular volume (MCV)	10	1	IRIS, 2007	Tier 1 (IRIS)
2-Methylnaphthalene	4.00E-03	mouse	pulmonary alveolar proteinosis	1000	1	IRIS, 2007	Tier 1 (IRIS)
2-secButyl-4,6-dinitrophenol(DNBP)	1.00E-03	rat	decreased fetal weight	1000	1	IRIS, 2007	Tier 1 (IRIS)
3+4 Methylphenol (cresol, m+p)	5.00E-02	rat	decreased body weights and neurotoxicity	1000	1	IRIS, 2007	Tier 1 (IRIS) surrogate; 3-methylphenol. Data only available for this isomer in the mixture

Table 5-14. Chronic Oral Reference Doses for Organic Chemicals. (7 Pages)

Chemical	Oral RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
4-(2,4-Dichlorophenoxy)butanoic acid	8.00E-03	dog	internal hemorrhage, mortality	1000	1	IRIS, 2007	Tier 1 (IRIS)
Acenaphthene	6.00E-02	mouse	hepatotoxicity	3000	1	IRIS, 2007	Tier 1 (IRIS)
Acenaphthylene	6.00E-02					IRIS, 2007	Tier 1 (IRIS) surrogate; acenaphthene, structural similarity
Acetone	9.00E-01	rat	nephropathy	1000	1	IRIS, 2007	Tier 1 (IRIS)
Aldrin	3.00E-05	rat	liver toxicity	1000	1	IRIS, 2007	Tier 1 (IRIS)
Alpha-BHC							Tier 3 (NCEA) value not used because route is covered by Tier 1 (IRIS) oral SF.
Anthracene	3.00E-01	mouse	no observed effects	3000	1	IRIS, 2007	Tier 1 (IRIS)
Aroclor 1242	2.00E-05					IRIS, 2007	Tier 1 (IRIS) surrogate; aroclor 1254, structural similarity
Aroclor 1248	2.00E-05					IRIS, 2007	Tier 1 (IRIS) surrogate; aroclor 1254, structural similarity
Aroclor 1254	2.00E-05	monkey	ocular exudates, inflamed and prominent Meibomian glands, distorted growth of finger and toe nails; decreased antibody (IgG and IgM) response to sheep erythrocytes	300	1	IRIS, 2007	Tier 1 (IRIS)
Aroclor 1260	2.00E-05					IRIS, 2007	Tier 1 (IRIS) surrogate; aroclor 1254, structural similarity
Aroclor 1262	2.00E-05					IRIS, 2007	Tier 1 (IRIS) surrogate; aroclor 1254, structural similarity
Benzene	4.00E-03	human	decreased lymphocyte count	300	1	IRIS, 2007	Tier 1 (IRIS)
Benzo(a)anthracene							Route is covered by Tier 3 (NCEA) oral SF.

Table 5-14. Chronic Oral Reference Doses for Organic Chemicals. (7 Pages)

Chemical	Oral RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Benzo(a)pyrene							Route is covered by Tier 1 (IRIS) oral SF.
Benzo(b)fluoranthene							Route is covered by Tier 3 (NCEA) oral SF.
Benzo(ghi)perylene	3.00E-02					IRIS, 2007	Tier 1 (IRIS) surrogate; pyrene, structural similarity
Benzo(k)fluoranthene							Route is covered by Tier 3 (NCEA) oral SF.
beta-1,2,3,4,5,6-Hexachlorocyclohexane							Tier 3 (NCEA) value not used because route is covered by Tier 1 (IRIS) oral SF.
Bis(2-ethylhexyl) phthalate	2.00E-02	Guinea pig	increased relative liver weight	1000	1	IRIS, 2007	Tier 1 (IRIS)
Butylbenzylphthalate	2.00E-01	rat	significantly increased liver-to-body weight and liver-to-brain weight ratios	1000	1	IRIS, 2007	Tier 1 (IRIS)
Carbazole							Route is covered by Tier 3 (HEAST) oral SF.
Carbon disulfide	1.00E-01	rabbit	fetal toxicity/ malformations	100	1	IRIS, 2007	Tier 1 (IRIS)
Carbon tetrachloride	7.00E-04	rat	liver lesions	1000	1	IRIS, 2007	Tier 1 (IRIS)
Chlordane	5.00E-04	mouse	hepatic necrosis; chlordane (technical)	300	1	IRIS, 2007	Tier 1 (IRIS)
Chloroform	1.00E-02	dog	moderate/marked fatty cyst formation in the liver and elevated SGPT	100	1	IRIS, 2007	Tier 1 (IRIS)
Chrysene							Route is covered by Tier 3 (NCEA) oral SF.

Table 5-14. Chronic Oral Reference Doses for Organic Chemicals. (7 Pages)

Chemical	Oral RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
cis-1,2-Dichloroethene	1.00E-02					EPA Region 6 HHMSSLs, 2007	Tier 2 (PPRTV)
Dalapon	3.00E-02	rat	increased kidney body weight ratio	300	1	IRIS, 2007	Tier 1 (IRIS)
Delta-BHC							Tier 3 (NCEA) surrogate alpha-BHC value not used because route is covered by Tier 1 (IRIS) oral SF for surrogate alpha-BHC.
Dibenz[a,h]anthracene							Route is covered by Tier 3 (NCEA) oral SF.
Dibenzofuran	2.00E-03					EPA Region 6 HHMSSLs, 2007	Tier 3 (NCEA)
Dicamba	3.00E-02	rabbit	maternal and fetal toxicity	100	1	IRIS, 2007	Tier 1 (IRIS)
Dichlorodiphenyldichloroethane							Route is covered by Tier 1 (IRIS) oral SF.
Dichlorodiphenyldichloroethylene							Route is covered by Tier 1 (IRIS) oral SF.
Dichlorodiphenyltrichloroethane	5.00E-04	rat	liver lesions	100	1	IRIS, 2007	Tier 1 (IRIS)
Dichloroprop	8.00E-03					IRIS, 2007	Tier 1 (IRIS) surrogate; 4-(2,4- Dichlorophenoxy)butanoic acid, structural similarity
Dieldrin	5.00E-05	rat	liver lesions	100	1	IRIS, 2007	Tier 1 (IRIS)
Diethyl ether	2.00E-01	rat	depressed body weights	3000	1	IRIS, 2007	Tier 1 (IRIS)
Diethylphthalate	8.00E-01	rat	decreased growth rate, food consumption and altered organ weights	1000	1	IRIS, 2007	Tier 1 (IRIS)

Table 5-14. Chronic Oral Reference Doses for Organic Chemicals. (7 Pages)

Chemical	Oral RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Di-n-butylphthalate	1.00E-01	rat	increased mortality	1000	1	IRIS, 2007	Tier 1 (IRIS)
Di-n-octylphthalate	4.00E-02	rat	hepatic effects	1000	1	RAIS, 2007	Tier 2 (PROV); value listed as provisional. Other occurring sources are Superfund Health Risk Technical Support Center or draft IRIS assessments.
Endosulfan I	6.00E-03	rat	reduced body weight gain in males and females; increased incidence of marked progressive glomerulonephrosis and blood vessel aneurysms in males (other effect: Decreased weight gain in males and neurologic findings in both sexes.)	100	1	IRIS, 2007	Tier 1 (IRIS) surrogate; endosulfan, endosulfan I is an isomer of endosulfan
Endosulfan II	6.00E-03						Tier 1 (IRIS) surrogate; endosulfan, endosulfan II is an isomer of endosulfan
Endosulfan sulfate	6.00E-03						Tier 1 (IRIS) surrogate; endosulfan, structural similarity
Endrin	3.00E-04	dog	mild histological lesions in liver, occasional convulsions	100	1	IRIS, 2007	Tier 1 (IRIS)
Endrin aldehyde	3.00E-04						Tier 1 (IRIS) surrogate; endrin, endrin aldehyde is an impurity and breakdown product of endrin
Endrin ketone	3.00E-04						Tier 1 (IRIS) surrogate; endrin, endrin ketone is the primary metabolite of endrin
Ethylene glycol	2.00E+00	rat	kidney toxicity	100	1	IRIS, 2007	Tier 1 (IRIS)

Table 5-14. Chronic Oral Reference Doses for Organic Chemicals. (7 Pages)

Chemical	Oral RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Fluoranthene	4.00E-02	mouse	nephropathy, increased liver weights, hematological alterations, and clinical effects	3000	1	IRIS, 2007	Tier 1 (IRIS)
Fluorene	4.00E-02	mouse	decreased RBC, packed cell volume and hemoglobin	3000	1	IRIS, 2007	Tier 1 (IRIS)
Heptachlor	5.00E-04	rat	liver weight increases in males	300	1	IRIS, 2007	Tier 1 (IRIS)
Heptachlor epoxide	1.30E-05	dog	increased liver-to-body weight ratio in both males and females	1000	1	IRIS, 2007	Tier 1 (IRIS)
Indeno(1,2,3-cd)pyrene							Route is covered by Tier 3 (NCEA) oral SF.
Isophorone	2.00E-01	dog	no observed effects	1000	1	IRIS, 2007	Tier 1 (IRIS)
Methoxychlor	5.00E-03	rabbit	excessive loss of litters	1000	1	IRIS, 2007	Tier 1 (IRIS)
Methyl isobutyl ketone	8.57E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation; Tier 3 (HEAST) oral RfD not used because more confidence in route extrapolation of IRIS (Tier 1) inhalation RfD.
Methylenechloride	6.00E-02	rat	liver toxicity	100	1	IRIS, 2007	Tier 1 (IRIS)
Naphthalene	2.00E-02	rat	decreased mean terminal body weight in males	3000	1	IRIS, 2007	Tier 1 (IRIS)
Pentachlorophenol	3.00E-02	rat	liver and kidney pathology	100	1	IRIS, 2007	Tier 1 (IRIS)
Phenanthrene	3.00E-02					IRIS, 2007	Tier 1 (IRIS) surrogate; pyrene, structural similarity
Phenol	3.00E-01	rat	decreased maternal weight gain	300	1	IRIS, 2007	Tier 1 (IRIS)
Picloram	7.00E-02	dog	increased liver weights	100	1	IRIS, 2007	Tier 1 (IRIS)

Table 5-14. Chronic Oral Reference Doses for Organic Chemicals. (7 Pages)

Chemical	Oral RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Pyrene	3.00E-02	mouse	kidney effects (renal tubular pathology, decreased kidney weights)	3000	1	IRIS, 2007	Tier 1 (IRIS)
Tetrachloroethene	1.00E-02	mouse/ rat	hepatotoxicity in mice, weight gain in rats	1000	1	IRIS, 2007	Tier 1 (IRIS)
Toluene	8.00E-02	rat	increased kidney weight	3000		IRIS, 2007	Tier 1 (IRIS)
Trichloroethene	3.00E-04					EPA Region 6 HHMSSLs	Tier 3 (NCEA)
Trichloromonofluoromethane	3.00E-01	mouse/ rat	survival and histopathology	1000	1	IRIS, 2007	Tier 1 (IRIS)

Table 5-15. Chronic Inhalation Reference Doses for Inorganic Chemicals. (2 Pages)

Chemical	Inhalation RfC (mg/m ³)	Inhalation RID (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Aluminum	5.00E-03	1.43E-03	human	psychomotor and cognitive impairment	300	1	EPA Region 6 HHMSSLs, 2007	Tier 2 (PPRTV)
Antimony								No Data
Arsenic								No Data
Barium	5.00E-04	1.43E-04	rat	fetotoxicity	1000	none	EPA Region 6 HHMSSLs, 2007	Tier 3 (HEAST)
Beryllium	2.00E-05	5.71E-06	human	beryllium sensitization and progression to chronic beryllium disease	10	1	IRIS, 2007	Tier 1 (IRIS)
Boron	2.00E-02	5.71E-03	human	respiratory tract bronchus irritation; bronchitis	100	none	EPA Region 6 HHMSSLs, 2007	Tier 3 (HEAST)
Cadmium								Route covered by Tier 1 (IRIS) inhalation SF
Chromium								No Data
Hexavalent Chromium	1.00E-04	2.86E-05	rat	lactate dehydrogenase in bronchioalveolar lavage fluid; Cr(VI) particulates	300	1	IRIS, 2007	Tier 1 (IRIS); particulate matter
Cobalt	2.00E-05	5.71E-06					EPA Region 6 HHMSSLs, 2007	Tier 2 (PPRTV)
Copper								No Data
Fluoride								No Data
Lead								See Section 5.5.8
Manganese	5.00E-05	1.40E-05	human	impairment of neurobehavioral function	1000	1	IRIS, 2007	Tier 1 (IRIS)
Mercury	3.00E-04	8.57E-05	human	hand tremor; increases in memory disturbances; slight	30	1	IRIS, 2007	Tier 1 (IRIS); elemental Hg

Table 5-15. Chronic Inhalation Reference Doses for Inorganic Chemicals. (2 Pages)

Chemical	Inhalation RfC (mg/m ³)	Inhalation RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Molybdenum				subjective and objective evidence of autonomic dysfunction				No Data
Nickel								No Data
Nitrogen in Nitrate								No Data
Nitrogen in Nitrite								No Data
Selenium								No Data
Silver								No Data
Strontium (elemental)								No Data
Thallium								No Data
Uranium								No Data
Vanadium								No Data
Zinc								No Data

Table 5-16. Chronic Inhalation Reference Doses for Organic Chemicals. (8 Pages)

Chemical	Inhalation RfC (mg/m ³)	Inhalation RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
1,1,2,2-Tetrachloroethane								Tier 2 (PPRTV) oral RfD not used to extrapolate to an inhalation RfD because inhalation route covered by Tier 1 (IRIS) inhalation SF.
1,2-Dichloroethane								Tier 3 (NCEA) inhalation RfD not used because inhalation route covered by Tier 1 (IRIS) inhalation SF.
1,4-Dichlorobenzene	8.00E-01	2.29E-01	rat	increased liver weights in P1 males	100	1	IRIS, 2007	Tier 1 (IRIS)
1-Butanol		1.00E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation
2-(2,4,5-Trichlorophenoxy)propionic acid		8.00E-03					IRIS, 2007	Tier 1 (IRIS) route extrapolation
2,4,5-Trichlorophenoxyacetic acid		1.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation
2,4-Dichlorophenoxyacetic acid		1.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation
2-Butanone	5.00E+00	1.43E+00	mouse	developmental toxicity (skeletal variations)	300	1	IRIS, 2007	Tier 1 (IRIS)
2-Butoxyethanol	1.30E+01	3.71E+00	rat	Changes in red blood cell (RBC) count	30	1	IRIS, 2007	Tier 1 (IRIS)
2-Methylnaphthalene		4.00E-03					IRIS, 2007	Tier 1 (IRIS) route extrapolation
2-secButyl-4,6-dinitrophenol(DNBP)		1.00E-03					IRIS, 2007	Tier 1 (IRIS) route extrapolation

Table 5-16. Chronic Inhalation Reference Doses for Organic Chemicals. (8 Pages)

Chemical	Inhalation RfC (mg/m ³)	Inhalation RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
3+4 Methylphenol (cresol, m+p)		5.00E-02					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation; 3-methylphenol.
4-(2,4-Dichlorophenoxy)butanoic acid		8.00E-03					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Acenaphthene		6.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Acenaphthylene		6.00E-02					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation; acenaphthene, structural similarity
Acetone		9.00E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Aldrin		3.00E-05					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Alpha-BHC								Tier 3 (NCEA) route extrapolation of oral RfD not used because the inhalation route is covered by Tier 1 (IRIS) inhalation SF.
Anthracene		3.00E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Aroclor 1242		2.00E-05					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation
Aroclor 1248		2.00E-05					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation
Aroclor 1254		2.00E-05					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Aroclor 1260		2.00E-05					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation

Table 5-16. Chronic Inhalation Reference Doses for Organic Chemicals. (8 Pages)

Chemical	Inhalation RfC (mg/m ³)	Inhalation RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Aroclor 1262		2.00E-05					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation
Benzene	3.00E-02	8.60E-03	human	decreased lymphocyte count	300	1	IRIS, 2007	Tier 1 (IRIS)
Benzo(a)anthracene								Route is covered by Tier 3 (NCEA) inhalation SF.
Benzo(a)pyrene								Route is covered by Tier 1 (IRIS) inhalation SF.
Benzo(b)fluoranthene								Route is covered by Tier 3 (NCEA) inhalation SF.
Benzo(ghi)perylene		3.00E-02					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation
Benzo(k)fluoranthene								Route is covered by Tier 3 (NCEA) inhalation SF.
beta-1,2,3,4,5,6-Hexachlorocyclohexane								Tier 3 (NCEA) route extrapolation of oral RfD not used for oral RfD because the inhalation route is covered by Tier 1 (IRIS) inhalation SF.
Bis(2-ethylhexyl) phthalate		2.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Butylbenzylphthalate		2.00E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Carbazole								Route is covered by Tier 3 (HEAST) inhalation SF.
Carbon disulfide	7.00E-01	2.00E-01	human	peripheral nervous system dysfunction	30	1	IRIS, 2007	Tier 1 (IRIS)
Carbon tetrachloride		7.00E-04					IRIS, 2007	Tier 1 (IRIS) route extrapolation

Table 5-16. Chronic Inhalation Reference Doses for Organic Chemicals. (8 Pages)

Chemical	Inhalation RfC (mg/m ³)	Inhalation RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Chlordane	7.00E-04	2.00E-04	rat	hepatic effects; chlordane (technical)	1000	1	IRIS, 2007	Tier 1 (IRIS)
Chloroform		1.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation; Tier 3 (NCEA) inhalation RfD not used because more confidence is placed in route extrapolation of Tier 1 (IRIS) oral RfD
Chrysene								Route is covered by Tier 3 (NCEA) inhalation SF.
cis-1,2-Dichloroethene		1.00E-02					EPA Region 6 HHMSSLs, 2007	Tier 2 (PPRTV) route extrapolation
Dalapon		3.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Delta-BHC								Tier 3 (NCEA) route extrapolation of oral RfD for surrogate alpha-BHC not used because the inhalation route is covered by the route extrapolation of a Tier 1 (IRIS) oral SF for surrogate alpha-BHC.
Dibenz[a,h]anthracene								Route is covered by Tier 3 (NCEA) inhalation SF.
Dibenzofuran		2.00E-03					EPA Region 6 HHMSSLs, 2007	Tier 3 (NCEA) route extrapolation

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Table 5-16. Chronic Inhalation Reference Doses for Organic Chemicals. (8 Pages)

Chemical	Inhalation RfC (mg/m ³)	Inhalation RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Dicamba		3.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Dichlorodiphenyldichloroethane								Route is covered by route extrapolation of a Tier 1 (IRIS) oral SF.
Dichlorodiphenyldichloroethylene								Route is covered by route extrapolation of a Tier 1 (IRIS) oral SF.
Dichlorodiphenyltrichloroethane		5.00E-04					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Dichloropropr		8.00E-03						Tier 1 (IRIS) route extrapolation
Dieldrin		5.00E-05					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Diethyl ether		2.00E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Diethylphthalate		8.00E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Di-n-butylphthalate		1.00E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Di-n-octylphthalate		4.00E-02					RAIS, 2007	Tier 2 (PPRTV) Route Extrapolation
Endosulfan I		6.00E-03					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation, endosulfan
Endosulfan II		6.00E-03					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation
Endosulfan sulfate		6.00E-03					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation

Table 5-16. Chronic Inhalation Reference Doses for Organic Chemicals. (8 Pages)

Chemical	Inhalation RfC (mg/m ³)	Inhalation RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Endrin		3.00E-04					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Endrin aldehyde		3.00E-04					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation
Endrin ketone		3.00E-04					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation
Ethylene glycol		2.00E+00					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Fluoranthene		4.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Fluorene		4.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Heptachlor		5.00E-04					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Heptachlor epoxide		1.30E-05					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Indeno(1,2,3-cd)pyrene								Route is covered by Tier 3 (NCEA) inhalation SF.
Isophorone		2.00E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Methoxychlor		5.00E-03					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Methyl isobutyl ketone	3.00E+00	8.57E-01	rat	Reduced fetal body weight, skeletal variations, and increased fetal death in mice, and skeletal variations in rats.	300	1	IRIS, 2007	Tier 1 (IRIS)

Table 5-16. Chronic Inhalation Reference Doses for Organic Chemicals. (8 Pages)

Chemical	Inhalation RfC (mg/m ³)	Inhalation RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Methylenechloride		6.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation; Tier 3 (HEAST) inhalation RfD not used because more confidence is placed in route extrapolation of Tier 1 (IRIS) oral RfD.
Naphthalene	3.00E-03	8.57E-04	mouse	nasal effects: hyperplasia and metaplasia in respiratory and olfactory epithelium, respectively	3000	1	IRIS, 2007	Tier 1 (IRIS)
Pentachlorophenol		3.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Phenanthrene		3.00E-02					IRIS, 2007	Tier 1 (IRIS) surrogate route extrapolation; pyrene, structural similarity
Phenol		3.00E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Picloram		7.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Pyrene		3.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation
Tetrachloroethene		1.00E-02					IRIS, 2007	Tier 1 (IRIS) route extrapolation; Tier 3 (NCEA) inhalation RfD not used because more confidence is placed in route extrapolation of Tier 1 (IRIS) oral RfD.

Table 5-16. Chronic Inhalation Reference Doses for Organic Chemicals. (8 Pages)

Chemical	Inhalation RfC (mg/m ³)	Inhalation RfD (mg/kg-day)	Species	Critical Effect	UF	MF	Source	Notes
Toluene	5.00E+00	1.43E+00	human	neurological effects in occupationally-exposed workers	10		IRIS, 2007	Tier 1 (IRIS)
Trichloroethene		1.10E-02					EPA Region 6 HHMSSLs	Tier 3 (NCEA)
Trichloromonofluoromethane		3.00E-01					IRIS, 2007	Tier 1 (IRIS) route extrapolation; Tier 3 (HEAST) inhalation RfD not used because more confidence is placed in route extrapolation of Tier 1 (IRIS) oral RfD.

Table 5-17. Chronic Oral Slope Factors for Inorganic Chemicals. (2 Pages)

Chemical	Oral CSF (mg/kg-day)	Species	Tumors Observed	Dose- Response Model	W.O.E class	Source	Notes
Aluminum							Route covered by Tier 2 (PPRTV) oral RfD.
Antimony							Route covered by Tier 1 (IRIS) oral RfD
Arsenic	1.5	human	skin cancer	multistage, with time	A	IRIS, 2007	Tier 1 (IRIS)
Barium							Route covered by Tier 1 (IRIS) oral RfD
Beryllium							Route covered by Tier 1 (IRIS) oral RfD
Boron							Route covered by Tier 1 (IRIS) oral RfD
Cadmium							Route covered by Tier 1 (IRIS) oral RfD
Chromium							Route covered by Tier 1 (IRIS) oral RfD
Hexavalent Chromium							Route covered by Tier 1 (IRIS) oral RfD
Cobalt							Route covered by Tier 2 (PPRTV) oral RfD
Copper							Route covered by Tier 3 (HEAST) oral RfD
Fluoride							Route covered by Tier 1 (IRIS) oral RfD
Lead							See Section 5.5.8
Manganese							Route covered by Tier 1 (IRIS) oral RfD
Mercury							Route covered by Tier 1 (IRIS) oral RfD
Molybdenum							Route covered by Tier 1 (IRIS) oral RfD
Nickel							Route covered by Tier 1 (IRIS) oral RfD
Nitrogen in Nitrate							Route covered by Tier 1 (IRIS) oral RfD
Nitrogen in Nitrite							Route covered by Tier 1 (IRIS) oral RfD
Selenium							Route covered by Tier 1 (IRIS) oral RfD
Silver							Route covered by Tier 1 (IRIS) oral RfD
Strontium (elemental)							Route covered by Tier 1 (IRIS) oral RfD
Thallium							Route covered by Tier 1 (IRIS) oral RfD