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6. Baseline Risk Assessment

The purpose of a baseline risk assessment is to estimate the risk of leaving the affected material in place (i.e., no action). This assessment was performed for the CSMRI Site by New Horizons and remains valid as the Baseline Risk Assessment. Based largely on the results of the baseline risk assessment, the 2004 RI/FS concluded that the no further action alternative and leaving the Site in its condition as of 2004 were not acceptable because they were not protective of human health and the environment, and a ROD was issued to document this conclusion. The 2004 RI/FS explained that the subsistence farmer and urban resident would be exposed to excessive risk with the then-current baseline Site conditions and it could have been a continuing problem for the underlying groundwater and Clear Creek. As noted previously, the 2004 remediation was halted because the nature and extent of the contamination was found to have been greater than previously calculated by the investigation in the 2004 RI/FS. Therefore, it was clear as of 2004 that the risks to the subsidence farmer, urban resident, the underlying groundwater, and Clear Creek could only be greater than was previously estimated by the prior risk assessment. Having rejected the no action alternative in the 2004 RI/FS, and knowing the nature and extent of contamination was greater than previously believed, it was safe to assume that a proactive remedy would be necessary. The 2006 investigation method included the excavation and stockpiling of the impacted soils to determine the nature and extent of contamination because it was the most reliable and cost-effective method to determine the nature and extent under these Site circumstances. Excavation of the contaminated soils was also one of the necessary elements of the eligible remaining remedial alternatives that would have resulted in a protective remedy. The investigative excavation of the contaminated soils also altered the physical conditions of the Site by taking the in-situ contamination and transferring it to one of two stockpiles on Site. The results of the additional investigation performed in 2006 - 2007 confirmed that the nature and extent of contamination were greater than that calculated by the 2004 RI/FS. The baseline risk is greater than that previously believed in 2004. Because the risk was great enough to reject the "No Action" alternative in 2004, and the risk is now known to be greater than before, there is no need to perform another baseline risk assessment. The baseline conditions are the in-situ conditions, not the conditions as currently found with the two stockpiles.

Nonetheless, even if the Site conditions as they exist in April 2007 were selected as the new baseline conditions, with all contaminated soils located ex-situ in two stockpiles at the Site, those baseline conditions would still warrant the rejection of the no action alternative as not being protective of human health and the environment. This conclusion is demonstrated by looking at the impact the current Site conditions have on the 2004 RI/FS risk assessment without having to perform another risk assessment from the beginning of the formal risk assessment process. It is more reasonable and cost effective to build off the prior work and add new information developed during the 2006 – 2007 investigation than to perform another risk assessment. The risk assessment discussed below evaluates the alternative scenario that the baseline conditions are the current conditions of having all soil contamination located in two stockpiles on site. This is presented to demonstrate that the no action remedy must be rejected whether baseline conditions are the in-situ conditions or the ex-situ conditions. The main impacts to the following risk assessment caused by the changed Site configuration are the temporary elimination of impacted soil from providing a source for groundwater contamination (the stockpiles are on a *liner) and the locally increased risk resulting from all the Site-impacted soil being placed in stockpiles.*

This risk assessment completed in 2004 examined both carcinogenic risks and health hazards associated with the material in-place on the Site. Near-term land use scenarios could include a recreational area. Foreseeable land use could also include the construction of student housing, research, or academic buildings. For example, the School recently competed for a research project consisting of a facility for a supercomputer, and they considered using a portion of the site for this. However, future land use could include an urban resident or potentially a subsistence farmer considering the persistence of the metals and the longevity of the radionuclides (half-life: Ra-226, 1.6 x 10^3 years; Th-230, 7.6 x 10^4 years). The requirements of 40 CFR §192.02 require that remedies for sites with similar radionuclide contaminants provide up to 1,000 years of protection to human health and the environment (at least 200 years). For a CERCLA NCP baseline risk assessment, the conservative subsistence farmer scenario was used as the baseline. After public comment, the subsistence farmer scenario was replaced with the urban resident scenario in the 2004 ROD. This had a small impact on the DCGLs for the Site, but for practical purposes, it made little difference because field excavation was likely to lead to the excavation of the same volumes of contaminated soil. The change in receptors had not materially altered the remedy selection or remedy performance. To provide an overall picture of relative risk, urban residential and recreational scenarios have been provided for comparison in this assessment.

6.1 Human Health Evaluation

Acceptable exposures, evaluated in the 2004 RI/FS, to known or suspected carcinogens are generally those that represent an excess upper-bound lifetime cancer risk to an individual of between 10^{-4} to 10^{-6}. EPA uses the 10^{-6} risk level as the point of departure for determining remediation goals for the National Priority List (NPL) sites. However, the upper boundary of the risk range is not a discrete line at 1×10^{-6}. A specific risk estimate around 10^{-4} may be considered acceptable if justified based on site-specific conditions (EPA 1991). EPA references site-specific acceptable risks in the range of 3×10^{-4}, but risks may become unacceptable in the range of 6×10^{-4} (EPA 1997a).

Noncarcinogens are evaluated by their systemic effect on target organs or systems. EPA defines acceptable human exposure levels (including sensitive subgroups) as those that do not cause adverse effects during a lifetime or part of a lifetime, incorporating an adequate margin of safety. This acceptable exposure level is best approximated by a hazard index (HI) of 1. If an HI is less than 1, adverse effects usually are not expected. As the HI increases beyond 1, the possibility of adverse health effects also increases.

The HI is calculated by summing the hazard quotients (HQ) for substances that affect the same target organ or organ system (e.g., respiratory system). The HQ is the ratio of potential exposure to the substance and the level at which no adverse health effects are expected. If the HQ is calculated to be less than 1, then no adverse health effects are expected as a result of exposure. If the HQ is greater than 1, then adverse health effects are possible. The HQ cannot be translated to a probability that adverse health effects will occur and is often not proportional to risk.

The approach to human health risk assessment for lead differed from that of other metals and contaminants. Risks from lead exposures typically are estimated from long-term exposures, although elevated blood lead (PbB) concentrations also result from short-term exposures. EPA and the Centers for Disease Control (CDC) have determined that childhood PbB concentrations at or above 10 micrograms of lead per deciliter of blood (µg Pb/dL) present risks to children's health (CDC 1991). Accordingly, EPA seeks to limit the risk that children will have lead concentrations above 10 µg Pb/dL.

Numerous tools were used for the baseline risk assessment in the 2004 RI/FS. Radionuclides risk was modeled using the RESRAD (version 6.2.1) model developed by the Environmental Assessment Division of Argonne National Laboratory for the DOE and the NRC (Yu *et al.* 2001). RESRAD used the current slope factors referenced in the Health Effects Assessment Summary Tables (HEAST). Health hazards were evaluated using the Risk Assessment Information System (RAIS) developed by Bechtel Jacobs Company LLC for the DOE, Office of Environmental Management (http://risk.lsd.ornl.gov/index.shtml). RAIS used the current reference doses and slope factors referenced in the EPA Integrated Risk Information System (IRIS) but for this assessment the information was supplemented by recent publications. The EPA Integrated Exposure Uptake Biokinetic Model for Lead in Children (IEUBK) was used to predict potential PbB concentrations. The model considered several different media through which children can be exposed to lead (EPA 2003b). A preliminary groundwater model was generated using Visual Modflow Pro in combination with Modflow SURFACT (Waterloo Hydrogeologic).

6.2 Exposure Assessment

Currently, the area has increased exposure pathways because the soil is no longer *in situ* but is located in two stockpiles near the access to the Site, but within the perimeter fencing. The soil has had soil tackifier applied to it to prevent short-term dust and erosion. The area is surrounded with a chain-link fence and posted.

Activities at the Site have been limited to monthly and quarterly monitoring and several months of soil segregation activities. External, inhalation, and dermal exposures were minimized through engineering and management controls during these activities. Dosimeters and personal air monitors were determined not to be necessary during soil segregation and maintenance operations. Erosion control measures are in place to limit movement of waterborne particles and airborne particles. No drinking water supply wells are in the immediate vicinity of the Site.

One exposure pathway that has been temporarily eliminated on the Site by the excavation of contaminated soils and placement into the stockpiles is movement of material to the underlying groundwater by particle and solute transport. This has been accomplished by the soil segregation activities eliminating the source for groundwater impacts. Although the groundwater is not used as a drinking water source, it eventually enters the Clear Creek alluvial system. The City of Golden uses Clear Creek as the primary drinking water source, but the surface-water diversion is located about 0.9 mile upstream of the Site. Coors Brewing Company uses alluvial wells located about 0.4 mile downstream from the Site. Additional downstream diversions that supply drinking water include the Agricultural Ditch (0.6 mile) and the Farmers' Ditch (0.7 mile).

For the baseline risk assessment completed for the 2004 RI/FS, the exposure scenarios examined include a subsistence farmer, an urban resident, and a recreational user. Baseline exposure scenarios were examined for a 30-year period and assumed minimal changes to the current topography (depressions left by the demolition of building foundations would remain). Exposure for the subsistence farmer assumed a farmhouse constructed on the existing soil, prior to soil segregation, groundwater as the primary drinking water source (including farm animals), and consumption of crops, meat, and milk produced from the local soil. The urban resident assumed a house similar to neighborhood housing, but drinking water would come from city water mains and minimal consumption of fruits and vegetables raised in a backyard garden. The recreational receptor assumed regular use by a nearby resident who would use the area for a variety of activities. Factors associated with the exposure scenarios were used in the RESRAD and RAIS models. RESRAD and RAIS sample inputs were provided in Appendix J of the 2004 RI/FS.

6.3 Soil Radionuclide Risk Characterization

The risk characterization for the Site completed for the 2004 RI/FS included the risk associated with radionuclides and two of the 11 metals (arsenic and chromium). This section describes the risk calculations performed during the 2004 RI/FS for eight of the identified radionuclides. These risk calculations were modified during preparation of the ROD to focus on the Urban Resident rather than the subsistence farmer. For comparison purposes we have included not only the urban resident but also the subsistence farmer scenarios. Additional radionuclides were identified in the soil samples, but the RESRAD option that only uses radionuclides with half-lives of one year or greater was selected. A half-year option also is available but did not seem to be appropriate for a site of this age. Two radionuclides, K-40 and Cd-109, identified by the sample results were not included in the risk analysis. K-40 is present at concentrations within the range of background values. The Cd-109 analytical results are flagged with an “S”, which indicates possible interference with another element. With a half-life of 464 days, any Cd-109 that may have been present during Site operations should have decayed to the daughter products.

Risk effects of the radionuclides were examined in the 2004 RI/FS, and modified in the ROD using RESRAD 6.21, the DOE and NRC model for site-specific dose assessment of residual radioactivity. Risks associated with the scenarios discussed in Section 6.2 of the 2004 RI/FS were examined using this model. A summary of typical input parameters for the RESRAD model was provided in Appendix I of the original RI/FS. Actual RESRAD runs for each scenario were provided in Appendix J of the original RI/FS. These scenarios include the subsistence farmer scenario which was eliminated during the ROD.

6.3.1 RESRAD Model Description

The RESRAD computer program is a pathway analysis model designed to evaluate the potential radiological dose incurred by an individual who occupies land containing residual radioactive material (Yu *et al*. 2001). Version 6.21 of RESRAD was used in the 2004 RI/FS for this analysis. That version has the capabilities of performing both deterministic and probabilistic dose assessments.

Three primary exposure pathways are considered by the RESRAD model in the 2004 RI/FS including:

1. Direct exposure to external radiation from the contaminated soil,
2. Internal dose from inhalation of airborne radionuclides, including radon progeny, and
3. Internal dose from ingestion of radionuclides, which includes ingestion of:
   - Plant foods grown in the contaminated soil irrigated with contaminated water,
   - Meat and milk from livestock fed with contaminated fodder and water,
   - Drinking water from a contaminated well or pond,
   - Fish from a contaminated pond, and
   - Contaminated soil.

RESRAD has been widely accepted and has a large user base. The models used in the software were designed for and have been successfully applied at sites with relatively complex physical and contamination conditions. In addition, the software has been verified and validated (Yu, 1999; NRC, 1998).

A number of RESRAD capabilities were introduced in the 2004 RI/FS but were not part of the baseline risk assessment. However, these capabilities were important for the evaluation of the selected alternatives.

6.3.2 Critical Population Group

The critical population group, identified during the 2004 RI/FS, represents the potential individuals who would experience the most conservative radiological exposure from the Site now or in the future. The group was modified in the ROD to use the urban resident instead of the subsistence farmer. The intent was to identify exposure scenarios for probable future uses of the Site but not necessarily the worst-case scenario. The worst-case scenario could potentially limit the usefulness of the resulting release criteria without providing significantly increased benefits to the public health, public safety, or the environment. However, radionuclides and metals are problematic for defining the critical population group because of their long-term persistence. Baseline risk assessments typically are made using the subsistence farmer scenario, but the less conservative urban resident was used for this Site.

The definition of the population group or receptor and the site-specific allowable dose was used by RESRAD in the 2004 RI/FS, as modified by the ROD, to determine the DCGL. Although not determined as part of the baseline risk assessment, the DCGLs are used to determine the site-specific cleanup requirements for radionuclides. The allowable dose comes from the release criterion determined by regulatory limits expressed in terms of dose (mrem/yr) (Note: release criteria also are evaluated by cancer incidence of cancer mortality risk). A release criterion is typically based on total or committed effective dose equivalent (TEDE or CEDE) and generally cannot be measured directly. Exposure pathway modeling is used to calculate a radionuclide-specific predicted concentration or surface area concentration of specific nuclides that could result in a dose (TEDE or CEDE) equal to the release criterion. RESRAD uses the term DCGL to describe this concentration. Exposure pathway modeling is an analysis of various exposure pathways and scenarios used to convert dose into concentration. Although regulatory guidance may suggest default DCGLs, site-specific modeling is preferred.

The receptor/site-specific DCGLs for individual radionuclides were calculated during the 2004 RI/FS, as modified by the ROD. The calculated value assumed that only one radionuclide is contributing to the dose established for the release criteria. When multiple radionuclides are present onsite, the combined dose contributed by the radionuclides at their individual DCGL resulted in the release criteria (dose) being exceeded. One method to adjust for the multiple radionuclides was to modify the assumptions made during exposure pathway modeling to account for multiple radionuclides. A second method was to use the sum-of-the-fractions rule to adjust the individual DCGLs. Each radionuclide activity expected at the end of the cleanup is divided by its predicted DCGL for the appropriate receptor. The ratios (fraction) only need to be determined for radionuclides expected to be present at measurable activities after the cleanup. The sum-of-the-fractions, the radionuclide (significant)-specific activities, and DCGLs must be less than or equal to one. As previously mentioned, DCGLs were not determined as part of the baseline risk assessment but were calculated for specific cleanup alternatives discussed in Section 8.2.

Another use of RESRAD, as presented in the 2004 RI/FS, was the determination of area factors for the Site cleanup. Using the approach suggested in MARSSIM, area factors were determined using RESRAD for the small site areas with elevated radionuclide activity. These factors were used to establish DCGLs for elevated measurement comparisons and for the evaluation of scan sensitivities to provide a reasonable level of assurance that any small area of elevated residual activity is not significant. The $DCGL_{emc}$ was established as:

$$DCGL_{emc} = \text{Area Factor} * DCGL$$

During the evaluation of measurement data for each survey unit, any measurement from the unit that is equal to or greater than the DCGL will be investigated by comparison with the DCGLemc using the elevated measurement approach of Section 8.5.1 of MARSSIM to determine if the elevated measurement is acceptable. As with the DCGL, the $DCGL_{emc}$ would be subject to the sum-of-the fractions rule. Again, $DCGL_{emc}$ were not determined for the baseline risk assessment but are included for specific cleanup alternatives (Section 8.2).

Two variations of the baseline scenarios were examined during the 2004 RI/FS to show the importance of the areas with elevated radionuclide activities or metal concentrations. One involved the placement of the receptor only on the area affected by elevated radium-226 activities, also evaluating the co-located metals. The other placed the receptor on areas with lead concentrations above the CDPHE-proposed Tier 2 residential standard (400 mg/kg), again evaluating the risk associated with the co-located metals and radionuclides.

6.3.3 Receptor Dose/Risk Assessment

The 2004 RI/FS, as modified by the ROD, determined the dose for the theoretical receptor (farmer, resident, recreational user) by defining the property where the individual is exposed for 30 years (6 years as a child and 24 years as an adult) in the RESRAD model. The modeled property consisted of an area with site-specific residual radionuclides to an assumed depth. The model incorporated a large number of parameters to numerically simulate the pathways that the radionuclides can use to affect the receptor. A summary of these parameters was provided in Appendix I of the original RI/FS and modified and re-presented in the ROD. For the baseline model, the Site was approximated by a rectangular area with about the same overall surface area as the Site and an average depth of material that was estimated from the RI information gathered during the 2003 RI. Radionuclide activities used for the model were average activities determined from surface soil samples collected during the 2003 RI. Subsurface soil sample activities were not used because the test pits and borings indicated that except for specific areas, the majority of the contamination was located in the upper regions of the soil. (During the 2006 RI, it was learned that significant contamination was located below the upper regions of the soil.) Risk associated with groundwater was determined during the 2004 RI/FS using the RAIS model because of groundwater modeling limitations of the RESRAD model.

Two additional radionuclide activity subsets were determined for the baseline scenario variations mentioned in the previous section. The data sets were generated assuming the receptor was exposed to an area with activities or concentrations above a specified limit. The two subsets include one area with combined radium-226 and -228 activities above 5 pCi/g (radium biased) and a second area with lead concentrations above 400 mg/kg (lead biased). Because of the area selection method - surface soil data were sorted using the mentioned cutoffs rather than using actual adjacent sampling locations - the data sets are biased somewhat higher than actual Site conditions but are representative of a combination of small areas. These subsets were selected to show the variability of the Site and the possible associated risks.

Exposure pathways evaluated by RESRAD, as presented in the 2004 RI/FS and modified by the ROD, included external gamma (gamma radiation from affected material on the property surface), inhalation (dust and soil particles inhaled during normal activities), ingestion (soil, water, and foodstuffs such as meat, milk, fruits, and vegetables), and radon (from diffusion from soil into houses and dissolved in water sources). RESRAD has default values to describe the different pathway parameters, but site-specific data are normally used to refine the model for the actual site and receptor. Some of the factors are more sensitive to change than others, such as the time of exposure to external gamma (fraction of time spent outdoors), permeability/porosity of the contaminated material (for radon), and soil ingestion (children typically ingest more soil). The literature references a wide range of assumptions used for the RESRAD parameters (USACE 2002).

RESRAD includes a diffusion model for estimating radon flow in soil and into habitable structures. Radon is a decay product of radium, and radon gas may migrate into structures constructed on soils containing radium. The RESRAD code estimates the movement of radon through onsite soils and determines possible indoor concentrations. However, indoor radon concentrations are influenced by meteorological conditions, indoor heating and air conditioning practices, local geological characteristics, structural air spaces and airflow conduits, seasonal variances, and other factors that are beyond RESRAD programming. Assumptions made in the 2004 RI/FS concerning RESRAD input parameters such as the contaminated zone density, contaminant zone total porosity, and cover material porosity can significantly affect the predicted radon dose and risk. Heterogeneous soils such as those found at the Site introduce significant uncertainty into radon predictions. The USACE White Paper states that indoor radon concentrations using RESRAD (or another other model) may grossly under estimate or over estimate indoor radon concentrations.

Radon limits and guidelines are based on concentration and not risk. EPA used an indoor concentration limit of 0.02 Working Level (WL), or about 4 pCi/L. This limit has been adopted by the NRC and the DOE and is typically categorically excluded for radiological dose calculations under these agencies. Risks associated with a concentration of 4 pCi/L (assuming residential exposure) is well above the CERCLA target risk range, and even small fractions of the guideline can produce risks on the order of 10^{-4}. While a qualitative evaluation is preferred, the 0.02 WL guideline does exist and, in some cases, must be evaluated in some detail to satisfy regulators and stakeholders. For example, Title 40 CFR Part 192 specifically limits indoor radon levels to 0.02 WL. Although not a risk limit, the regulatory requirement exists and RESRAD can be used to predict indoor radon levels in both WL and pCi/L concentration.

No specific data were collected for onsite radon because of the variability of the Site and the potential for material removal. Because of the lack of site-specific radon information, a limited number of scenarios were evaluated for the 2004 RI/FS for radon to determine parameter sensitivity to potential dose effects. The results of the sensitivity analysis showed significant variation in dose by modifying soil parameters that were possible onsite. Because of the large variation in predicted doses produced by the sensitivity analysis, the actual Site evaluation disregarded the majority of the radon dose/risk contribution (Section 6.3.4). The radon pathway was left on for most scenarios but was minimized by placing the lowest level of the residence below the affected soil (Section 6.3.4).

6.3.4 RESRAD Results

A summary of the RESRAD dose and risk predictions for the various scenarios is provided in Table 6-1. These predictions and Table 6-1 were prepared for the 2004 RI/FS and are re-printed herein. The two area variations are provided for comparison (Section 6.3.2, last paragraph).

Table 6-1
RESRAD Dose and Risk Predictions

Scenario	30-Year Dose (mrem/yr)	30-Year Risk
Current Conditions – Average Soil Activities		
Subsistence Farmer	42	7.4×10^{-4}
Urban Resident	35	6.0×10^{-4}
Recreational User	0.32	7.3×10^{-6}
Current Conditions – Radium Biased Soil Location		
Subsistence Farmer	190	3.4×10^{-3}
Urban Resident	64	1.3×10^{-3}
Recreational User	1.5	3.4×10^{-5}
Current Conditions – Lead Biased Soil Location		
Subsistence Farmer	110	1.9×10^{-3}
Urban Resident	37	8.1×10^{-4}
Recreational User	0.87	2.0×10^{-5}

This summary does not include the risk associated with the onsite metals. Table 6-2 presents the total risk.

Because of the uncertainty of the RESRAD radon calculation, the scenarios were modified in the 2004 RI/FS to minimize the radon prediction. Using a basement with a floor located beneath the affected soil layer effectively minimizes the influence of the radon without turning off the radon pathway completely. For comparison, RESRAD was run for the average soil conditions, assuming slab construction (structure built directly on top of the affected soil). Dose and risk numbers calculated during the 2004 RI/FS were as follows:

Subsistence Farmer	Dose 220 mrem/yr	Risk $3.5x10^{-3}$
Urban Resident	Dose 210 mrem/yr	Risk $3.4x10^{-3}$
Recreational User	Dose 0.46 mrem/yr	Risk $9.8x10^{-6}$

These scenarios assumed the contaminated soils consist of a sandy clay, but by changing the permeability parameter to reflect more of a clayey sand, the dose for the subsistence farmer drops to 92 mrem/yr and the risk decreases to $1.5x10^{-3}$. Adding one meter of clay cover material can further decrease the subsistence farmer dose to 4.8 mrem/yr with an associated risk of $7.5x10^{-5}$.

The decay of Ra-226 to radon and its daughters could be a significant component of the total risk/dose to future Site receptors. Therefore, it seems appropriate to consider this in making a risk management decision for the Site. If the radon pathway is not bypassed (lowest level of residence is placed in the affected soil) dose and risk values (assuming a clayey sand soil) are about five times greater than the same scenario without the influence of radon. The “no-action” alternative is unacceptable whether or not radon emanation effects are considered.

When the alternative baseline conditions of *ex-situ* stockpiles are considered, the impacts are expected to be equal to or greater than those calculated in the 2004 baseline risk assessment. This is because the impacted material remains on Site and, while temporarily prevented from serving as a source of groundwater contamination, if left at the Site indefinitely, will eventually migrate to groundwater. In addition, the location of the stockpiled material would serve as a long-term source of radon into any structure constructed in that location as well as a source of contamination available for uptake by future receptors through the other exposure routes (i.e., dermal absorption, inhalation, and ingestion). As recently modeled, the dose to an urban resident due to radon emanation from the stockpiled material could be as high as 2,087 mrem/year.

6.4 Soil Metals Risk and Toxicity Assessment

This section describes the methods used during the 2004 RI/FS to determine the risks and hazard quotients associated with the 11 metals present onsite. The RAIS model was used in 2004 to determine the toxicity of nine of the metals, but cadmium and lead were determined using other methods. The literature indicates that radionuclides also have toxicity effects but no currently published referenced doses are in IRIS. Additional reference material was consulted, but no agreed-upon reference dose was identified. Typically health effects for radionuclides focus on cancer risks.

IRIS (and other reference material) lists both cadmium and lead as possible human carcinogens but neither has been assigned slope factors because of ongoing debates about sensitive populations and cancer-causing mechanisms. These same debates apply to the associated HQ determination, and currently no reference dose is provided for either metal. Estimation of the toxicity associated with each metal is discussed in the following sections. Risk estimates are provided for specific species of arsenic and chromium. The remaining seven metals evaluated during the RI are not currently considered carcinogenic.

6.4.1 RAIS Model Description

The RAIS is a web-based system used during the 2004 RI/FS to disseminate risk tools and supply information for risk assessment activities. Taking advantage of searchable and executable databases, menu-driven queries, and data downloads using the latest web-based technologies, the RAIS offers essential tools and information for the risk assessment process and can be tailored to meet site-specific needs. RAIS uses current values listed in the EPA IRIS database to generate the risks and hazards associated with each metal. RAIS input parameters were modified to mimic the RESRAD parameters, but RAIS does not have sufficient flexibility to exactly reflect the RESRAD inputs. RAIS is a top-level risk assessment program used to provide general information about the affected material.

6.4.2 Receptor Risk/Hazard Quotient Assessment

The 2004 RI/FS determined the HQ and risk for the theoretical receptor (subsistence farmer, urban resident, recreational user), the RAIS model defines an individual who is exposed for 30 years (6 years as a child and 24 years as an adult). The ROD focused on the urban resident rather than the subsistence farmer as the most appropriate exposure scenario. Exposure pathways included dermal (some metals are absorbed through the skin), ingestion (soil, water, and foodstuffs), and inhalation (dust and soil particles inhaled during onsite activities). Soil and water concentrations were entered into the model along with the exposure parameters. The subsistence farmer scenario included the use of onsite groundwater. Average metal concentrations measured in downgradient wells were used as the baseline values. The food exposure route was not used for this top-level risk assessment because of the uncertainty of using generalized food concentration data. It can be assumed that the overall HQ and risk values determined by the model would be biased somewhat low because of this missing component.

The two data subsets described in Section 6.3.3 also were examined in the 2004 RI/FS for the associated metals. Again, these subsets were selected to show the variability of the Site and assist in the determination of appropriate cleanup levels.

6.4.2.1 Cadmium Assessment

The following cadmium assessment was conducted as part of the 2004 RI/FS. Cadmium can be taken into the body by eating food (and associated soil), drinking water, or breathing air. Gastrointestinal absorption from food or water is the principal source of internally deposited cadmium in the general population. Gastrointestinal absorption is generally quite low, with only about 5 percent of the amount ingested being transferred to the bloodstream. Thirty percent of cadmium that reaches the blood deposits in the liver, another 30 percent deposits in the kidneys, and the remainder distributes throughout other organs and tissues of the body (per simplified models that do not reflect intermediate redistribution). Cadmium clears the body with a biological half-life of about 25 years (HHS 2001). The literature also cites a number of studies that have found that cadmium is a major contributor to autoimmune thyroid disease. Acute exposures have documented effects on the gastrointestinal tract, nervous system, kidneys, liver, and cardiovascular system. Chronic exposures have effects on the kidneys and bone with proteinuria, renal stones, and Itai-itai disease.

Because of cadmium's similarity to zinc (forms similar cations), the RAIS model was modified for this assessment to use zinc as a surrogate for cadmium. Major differences between the two metals include the gastrointestinal absorption factors (20 percent for zinc, 5 percent for cadmium), target organs, and the biological half-life (280 days for zinc and 25 years for cadmium – literature values range from 14 to 208 years). Using the zinc surrogate method, hazard quotients for cadmium were estimated to be in the range of $1x10^{-4}$ and do not appear to be a primary driver for the Site. The cadmium HQ also was estimated by modifying the drinking water pathway to simulate soil ingestion (this method would be considered to be conservative because the soil cadmium would not be as bioavailable as the cadmium dissolved in water). This method produced a similar magnitude HQ of $3.5x10^{-4}$.

6.4.2.2 Lead Assessment

The following lead assessment was conducted as part of the 2004 RI/FS. RAIS does not evaluate the HQ for lead because the IRIS database (and other reference material) does not provide a reference dose or slope factor for the metal. Although there is a strong correlation between exposure to lead-contaminated soils and blood lead concentration, numerous factors make a direct prediction of blood lead concentrations difficult. Soil particle size, lead species, bioavailability, and health of the exposed individual affect the uptake of lead. Alternative exposure paths such as lead paint and lead pipes in older buildings also influence blood lead concentrations. According to the IRIS website, "It appears that some of these effects, particularly changes in the levels of certain blood enzymes and in aspects of children's neurobehavioral development, may occur at blood lead levels so low as to be essentially without a threshold. The Agency's RfD Work Group discussed inorganic lead (and lead compounds) at two meetings (07/08/1985 and 07/22/1985) and considered it inappropriate to develop an RfD for inorganic lead." Often lead is regulated by the use of the soil standards; however, there is significant disagreement about the appropriate concentration. A paper published by the Agency for Toxic Substances and Disease Registry (ATSDR) lists recommended lead soil standards ranging from <100 mg/kg to 1,000 mg/kg (HHS 1992). The current proposed Tier 2 soil standard listed by CDPHE is 400 mg/kg. The Tier 2 table value for lead is based on current EPA guidance (EPA 1994).

The definition of residential properties for lead is somewhat different than for other hazardous materials. Residential properties are defined in the Superfund Lead-Contaminated Residential Sites Handbook (EPA 2003b) as any area with high accessibility to sensitive populations and includes:

- Properties containing single- and multi-family dwellings,
- Apartment complexes,
- Vacant lots in residential areas,
- Schools, day-care centers, and community centers,
- Playgrounds, parks, green ways, and
- Any other areas where children may be exposed to site-related contaminated media.

This document defines sensitive populations as young children (those under 7 years of age who are most vulnerable to lead poisoning) and pregnant women. Focus is placed on children less than 7 years old because blood lead levels typically peak in this age range. This age range is when children are most vulnerable to adverse cognitive effects of lead. Pregnant women are included due to the effects of lead on the fetus (EPA 2003b). This definition of residential property is applicable the evaluation of the Site.

6.4.3 IEUBK Model Description

The following IEUBK model description was conducted as part of the 2004 RI/FS. EPA has developed the IEUBK to predict lead levels in blood (PbB) concentrations in children exposed to lead. The model considers several different media through which children can be exposed to lead (EPA 2003b).

EPA and the CDC have determined that childhood PbB concentrations at or above 10 micrograms of lead per deciliter of blood (µg Pb/dL) present risks to children's health (CDC 1991). Accordingly, EPA seeks to limit the risk that children will have lead concentrations above 10 µg Pb/dL. The IEUBK model predicts the geometric mean PbB for a child exposed to lead in various media (or a group of similarly exposed children). The model also calculates the probability that the child's PbB exceeds 10 µg Pb/dL (P10). Preliminary remediation goals (PRGs) generally are determined with the model by adjusting the soil concentration term until the P10 is below 5 percent. Final cleanup level selection for Superfund sites generally is based on the IEUBK model results and the nine criteria analysis per the NCP (EPA 1997a), which includes an analysis of Applicable or Relevant and Appropriate Requirements (ARARs).

The IEUBK model was used to determine relative risk associated with the onsite lead concentrations. The input parameters do not directly correspond to RESRAD parameters because of the emphasis on a child's initial seven years of life. For this evaluation, the scenario-specific lead concentration was used, but the default values were used for the other model parameters. Sensitivity checks showed that the model was relatively sensitive to variation in soil ingestion (10 percent increase in soil ingestion produced a 7 percent increase in blood concentrations), but less sensitive to lead uptake through food consumption (10 percent increase in lead concentrations in food produced a 0.7 percent increase in blood concentrations).

6.4.4 RAIS and IEUBK Results

A summary of the RAIS risk and hazard index predictions for the various scenarios, as originally presented in the 2004 RI/FS, is provided in Table 6-2 along with the combined metal and radionuclide risk.

Table 6-2
RAIS Risk and Hazard Index Predictions

Scenario	30-Year Risk (RAIS)	Hazard Index	Combined Risk (RAIS & RESRAD)
Current Conditions – Average Soil Activities			
Subsistence Farmer	1.5×10^{-4}	1.8	1.0×10^{-3} (1)
Urban Resident	1.5×10^{-4}	1.8	7.5×10^{-4}
Recreational User	1.4×10^{-6}	0.034	8.7×10^{-6}
Current Conditions – Radium Biased Soil Location			
Subsistence Farmer	2.4×10^{-4}	3.2	3.8×10^{-3} (1)
Urban Resident	2.4×10^{-4}	3.2	1.5×10^{-3}

Table 6-2
RAIS Risk and Hazard Index Predictions

Scenario	30-Year Risk (RAIS)	Hazard Index	Combined Risk (RAIS & RESRAD)
Recreational User	3.2×10^{-6}	0.061	3.7×10^{-5}
Current Conditions – Lead Biased Soil Location			
Subsistence Farmer	2.4×10^{-4}	2.6	2.3×10^{-3} (1)
Urban Resident	2.3×10^{-4}	2.6	1.1×10^{-3}
Recreational User	3.2×10^{-6}	0.035	2.3×10^{-5}

1 Includes the RAIS predicted risk from radionuclides in groundwater (see Section 6.5).

The combined risk associated with the subsistence farmer and urban resident scenarios is in excess of the 1×10^{-4} typically considered to be the upper bound of the acceptable risk range. Hazard quotients for the subsistence farmer and urban resident scenarios were above 1. Again because of the limited time the recreational user spends on the Site, the risk level is less than 1×10^{-4} (but greater than 1×10^{-6}) and the HQ is less than 1.

Estimated PbB concentrations predicted by the IEUBK model in the 2004 RI/FS are provided in Table 6-3.

Table 6-3
Estimated Blood Lead Concentrations

Scenario	Blood Lead Concentration (µg/dL)
Current Conditions – Average Soil Activities	3.4
Current Conditions – Radium Biased Soil Location	5.6
Current Conditions – Lead Biased Soil Location	13

The 2004 RI/FS indicated it is difficult to distinguish between the different receptors for the lead exposure because of the way that residential property is defined. The offsite recreational user could include a neighborhood child that enters the site for a play area. Soil ingestion during play activities could be a significant fraction of an actual onsite resident's exposure. PbB also could be affected by lead concentrations in small areas. The guidance on lead requires small parcels of land be considered during the Site investigation, including areas as small as 100 square meters (smaller areas are to be considered if there are play areas) (EPA 2003a). These small areas could have significantly greater average lead concentrations. Using a number of co-located onsite soil samples generated average lead concentrations as high as 2,200 mg/kg, which produced PbB concentrations as high as 20 µg/dL.

The proposed CDPHE soil standard for lead is 400 mg/kg. Soil concentrations below this level are generally considered to be protective of human health and the environment (including children). An alternative risk-based standard can be used if risk modeling shows the alternative to be protective. However, additional data collection and modeling are often more costly than meeting the Tier 2 standard through remedial techniques.

6.5 Groundwater Hazard Index/Risk Assessment

As presented in the 2004 RI/FS, risk and hazard quotients for the water exposure route (use of onsite groundwater) were estimated in RAIS using metal concentrations measured in the downgradient monitoring wells. The effects of the metals were included in the RAIS results tables (Section 6.4.4). Risks associated with the radionuclides were determined separately using highest activities measured in the downgradient well (CSMRI-04). The predicted metals and radionuclide risk for an onsite receptor from the consumption of groundwater would be about 1.1×10^{-4}. The groundwater risk value was included in the combined risk number presented in Table 6-2. These values were only applicable to then-current Site conditions and require an onsite receptor.

Groundwater recharge can be expected to move the affected material into Clear Creek, but dilution effects would make it difficult to detect in the surface water. However, dilution effects are not as significant during drought years. Without source removal, the Site would be a long-term contributor of radionuclide and metal loads to Clear Creek. Segment 14 of Clear Creek (the Clear Creek reach near the Site) already has specific limits on cadmium loads.

No controls on the movement of affected material to groundwater were assumed for the baseline risk assessment. The full effect of continued exposure to precipitation events is difficult to predict with the limited amount of groundwater information. Without source material control, groundwater concentrations of metals and radionuclides would be expected to increase the longer the source material remains exposed to the weather.

Moving the impacted material to lined stockpiles essentially temporarily eliminates the continuing source of groundwater contamination. Hence, with the temporary segregation of the contamination source, contaminant concentrations in groundwater are expected to decrease over the short term.

6.6 Summary of Findings

The 2004 baseline risk assessment indicated that taking no future action and leaving the Site in its then-current condition is not protective of human health and the environment. The subsistence farmer and urban resident would be exposed to excessive risk with then-current Site conditions. Even without the subsistence farmer scenario, the ROD still rejected the no action alternative. The remedy must account for reasonable future land uses, such as urban residents, over a 1,000-year time period. Although there are minimal direct risks to the recreational user, the Site as of 2004 would be a continuing problem for the underlying groundwater and Clear Creek. Long-term institutional controls would be necessary to protect neighborhood children from exposure. Erosion controls would need to be maintained to minimize the transport of affected sediment to surrounding areas and eventually into Clear Creek. Radionuclides such as radium-226 and thorium-230 are very persistent in the environment, with half-lives of 1.6×10^{3} and 7.5×10^{4} years, respectively.

Table 6-4 summarizes some of the factors used to evaluate the 2004 baseline risk assessment. Overall, sufficient risks and hazards associated with the Site warrant remediation.

Table 6-4
Factors Used to Evaluate Baseline Risk Assessment

Scenario	Risk $<10^{-6}$	Risk 10^{-6} to 10^{-4}	Risk $<10^{-4}$	Dose <15 mrem/yr	Dose <25 mrem/yr	Hazard Index <1	PbB <10 µg/dL	Soil Lead <1200 mg/kg	Soil Lead <400 mg/kg	Protective of Groundwater	Satisfies ALARA
Current Conditions – Average Soil Activities							Y	Y	Y	N	N
Subsistence Farmer	N	N	N	N	N	N					
Urban Resident	N	N	N	N	N	N					
Recreational User	N	Y	Y	Y	Y	Y					
Current Conditions – Ra Biased Soil Activities							Y	Y	Y	N	N
Subsistence Farmer	N	N	N	N	N	N					
Urban Resident	N	N	N	N	N	N					
Recreational User	N	Y	Y	Y	Y	Y					
Current Conditions – Pb Biased Soil Activities							N	Y	N	N	N
Subsistence Farmer	N	N	N	N	N	N					
Urban Resident	N	N	N	N	N	N					
Recreational User	N	Y	Y	Y	Y	Y					

Notes: Y, meets requirement; N, does not meet requirement

The Site no is longer in the configuration described in the previous conclusions. Impacted *in-situ* soils have been excavated and placed into *ex-situ* lined stockpiles that temporarily eliminate a source for groundwater impacts. However, wind erosion is still a significant pathway for contaminant migration from the stockpiles.